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Diet and food availability: implications for foraging and dispersal of Prince of Wales northern flying squirrels across managed landscapes

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Where dispersal is energetically expensive, feeding and food availability can influence dispersal success. The endemic Prince of Wales northern flying squirrel (*Glaucomys sabrinus griseifrons*) inhabits a landscape mosaic of old-growth, 2nd-growth, and clear-cut stands, with the latter 2 representing energetically expensive habitats. We estimated the diet of flying squirrels using stable isotope and fecal analyses, determined whether food availability varies among forest stands, and assessed the likelihood of foraging across a managed landscape given the distribution of foods on Prince of Wales Island (POW), Alaska. Both stable isotope and fecal analyses revealed that conifer seeds, lichens, and fungi were the main dietary items consumed and assimilated by flying squirrels. Similarly, soil macroinvertebrates were consumed by squirrels, whereas berries were not. Nonetheless, although examination of stable isotope data suggested that squirrels assimilated few nutrients from truffles, this food source was among the most frequent diet items in feces, probably because flying squirrels assimilate elements other than nitrogen from fungi. Our surveys showed that conifer seeds, truffles, and lichens were more prevalent in old-growth than 2nd-growth and clear-cut habitats. Thus, our results indicate that diet and availability of food items on POW may influence foraging success and dispersal movements of *G. sabrinus* across fragmented landscapes because of limited availability of food resources in the managed habitats. DOI: 10.1644/09-MAMM-A-014R.1.

Key words: clear-cuts, fecal analyses, fungi, old-growth forests, soil macroinvertebrates, Southeast Alaska, stable isotope analysis, truffles

The endemic Prince of Wales flying squirrel (*Glaucomys sabrinus griseifrons* Howell, 1934) inhabits a habitat altered significantly by broadscale timber harvest; the once near contiguous old-growth Sitka spruce (*Picea sitchensis*)–western hemlock (*Tsuga heterophylla*) forests are now a mosaic of old-growth remnants, younger 2nd-growth stands (<60 years in age), and clear-cuts. Future timber harvest will further fragment the old-growth forest on Prince of Wales Island (POW), Alaska. Current plans to maintain a system of old-growth reserves assume that, despite the increased fragmentation, populations of old-growth obligate species will function as metapopulations through continued dispersal among old-growth isolates (United States Department of Agriculture Forest Service 1997). However, our recent studies demonstrated that flying squirrels gliding into 2nd-growth and clear-cut habitats may experience difficulties orienting toward the nearest forest edge, especially on cloudy, rainy nights (Flaherty et al. 2008). Under such conditions, which are common on POW, travel paths (relative to straight-line distance) of flying squirrels across clear-cut and 2nd-growth stands are considerably longer than in old-growth forests and are characterized by multiple pauses, which translate to greater travel time (Flaherty et al. 2008). In addition, we found that the cost of quadrupedal locomotion, the mode of transport adopted by squirrels in clear-cut and 2nd-growth stands, is higher than expected, especially when compared to other arboreal sciurids (E. A. Flaherty, pers. obs.). Thus, unless flying squirrels are able to replenish their depleted energy stores when dispersing across high-cost managed stands, successful dispersal and viable metapopulations are unlikely (Smith and Person 2007; Smith et al., in press).

Although the diet of *G. sabrinus* has been studied extensively (Smith 2007), limited information exists on the diet of populations in the temperate rain forests of Southeast Alaska (Pyare et al. 2002). Based on the diversity of spores identified in fecal samples in past studies, *G. sabrinus* appears to be primarily a mycophagist, specializing on the fruiting bodies of hypogeous mycorrhizal fungi (hereafter, truffles) in most parts of its range (Currah et al. 2000; Maser et al. 1986; Pyare et al. 2002). However, the reliance of *G. s. griseifrons* on fungi in Southeast Alaska may be lower than elsewhere (Pyare et al. 2002). Truffles have low nutritional value (Claridge et al. 1999; Cork and Kenagy 1989; Dubay et al. 2008), and by consuming a diversity of genera that differ in nutritional contents, squirrels are able to obtain essential nutrients (Claridge et al. 1999; Dubay et al. 2008). Because species diversity of truffles is lower in Southeast Alaska, flying squirrels may be unable to meet their nutritional needs by concentrating on truffles as a food source (Pyare et al. 2002). *G. sabrinus* augments its diet by consuming epigeous fungi (mushrooms); arboreal lichens such as *Bryoria*, *Usnea,* and *Alectoria* spp. (Maser et al. 1985; Rosentreter et al. 1997); berries; conifer seeds; new growth tips and buds from trees; bird eggs and young; animal tissue; and invertebrates (Maser et al. 1985; Thysell et al. 1997; Wells-Gosling and Heaney 1984).

Timber harvest changes the structure and microclimate of old-growth forests (Colgan 1997), removes the energy sources (trees) for fungi (Amaranthus et al. 1994; Colgan 1997), and damages the hyphal mat during logging operations (Carey et al. 2002). Thus, resulting clear-cuts, 2nd-growth, and thinned stands exhibit significantly lower fungal biomass and diversity than old-growth stands (Amaranthus et al. 1994; Carey et al. 2002; Waters et al. 1994), and little is known about the length of time required before fungi will reestablish colonies and begin to produce truffles (Amaranthus et al. 1994). The effects of timber harvest on the availability of other potential diet items of flying squirrels are even more obscure. Nonetheless, these managed habitats may be depleted not only in the preferred diet item, fungal fruiting bodies, but also in alternative foods. Such lower availability of food items may reduce the ability of dispersing flying squirrels to replenish their energy stores. Therefore, the objectives of our study were to estimate the relative importance of fungal fruiting bodies and other potential food items in the diet of northern flying squirrels in Southeast Alaska, determine if the abundance of those diet items varies between old-growth and managed forests, and evaluate whether dispersers would be expected to encounter adequate food resources while traversing managed landscapes of Southeast Alaska.

MATERIALS AND METHODS

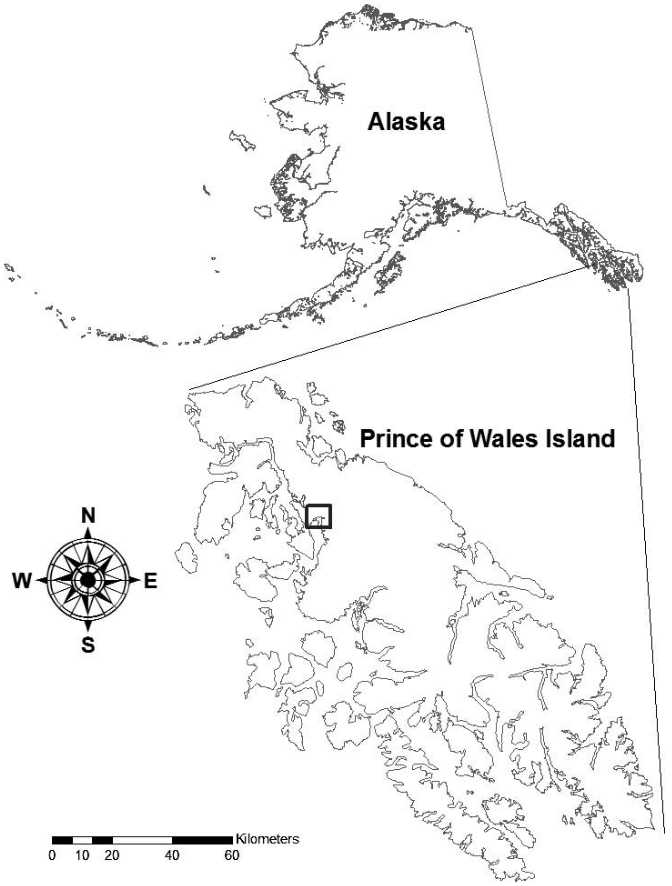

FIG. 1.—Map of core study area on Prince of Wales Island, Southeast Alaska. Trapping area was located inside the box. Surveys of food availability and trapping of northern flying squirrels (*Glaucomys sabrinus griseifrons*) occurred during spring 2003, 2004, and 2005, and autumn 2004 and 2005.

Study area.—Study sites were located on northern POW, Alaska, near the community of Naukati (55°52′N, 133°12′W; Fig. 1). The old-growth habitat is composed of Sitka spruce and western hemlock, with yellow cedar (*Xanthocyparis nootkatensis*) and western red cedar (*Thuja plicata*) occurring in more mesic areas. These old-growth stands contain many down, decaying logs and snags. The understory includes devil's club (*Oplopanax horridus*) and dense areas of *Vaccinium* spp. The 2nd-growth habitat is primarily ''dog-hair'' stands of spruce and hemlock (i.e., densely stocked with small-diameter trees that were on average no more than one-half the diameter of trees in old-growth—Alaback 1982). Second-growth rain forest in Southeast Alaska correspond ecologically to substantially younger 2nd-growth forests at lower latitudes because succession proceeds much slower ($\geq$300 years to develop old-growth forest structure). The remainder of the study area was composed of clear-cuts, which are recently ($<$5 years) disturbed stands with no overstory and a vegetation layer that includes skunk cabbage (*Lysichitum americanum*) and *Vaccinium* spp., with some small pools of standing water.

Livetrapping and sample collection.—Flying squirrels were trapped in the autumn months (August–October) of 2003–2005, a period that overlapped with juvenile dispersal, and the spring months (March–April) of 2004–2005, which corresponded with breeding dispersal. Trapping occurred on 3 different grids situated in old-growth forest stands within an area of approximately 10 km^2. Because radiotelemetry, fine-scale movement, and perceptual range data all indicated that flying squirrels were avoiding clear-cut and 2nd-growth habitats (Flaherty et al. 2008; S. Pyare and W. P. Smith, per. obs.), and because Smith and Person (2007) determined that low-quality old-growth habitats (i.e., peatland–mixed conifer stands) act as population sinks on POW, we did not establish trapping grids in managed stands. Tomahawk No. 201 (13 × 13 × 41-cm) live traps (Tomahawk Live Trap Co., Tomahawk, Wisconsin) were placed approximately 1.5 m above the forest floor on the bole of a tree and baited with a mixture of rolled oats, peanut butter, and molasses (Smith and Nichols 2003). Traps were checked at sunrise, and animals captured for the 1st time were marked with a unique passive integrated transponder tag (Biomark, Boise, Idaho); all captured individuals were weighed and sexed. We collected whole blood samples from each new captured adult for stable isotope analysis. Animals were briefly exposed to halothane, an inhalant anesthetic with a rapid induction rate and recovery (McColl and Boonstra 1999; Menzel et al. 2004), and we collected a small, triangular ear clipping from the medial edge of the right ear using a pair of sharp tissue scissors. We used microcapillary tubes to collect whole blood droplets from the ear. The tubes were stored inside sterile Whirlpacks (Nasco, Fort Atkinson, Wisconsin) and frozen. We collected blood samples only from adults because of the increased risk of mortality from cardiovascular depression associated with halothane exposure (McColl and Boonstra 1999) that could be exacerbated by the physiological stress from dispersal activity in juveniles. We also collected fecal pellets from the anus or from a clean, uncontaminated surface where they fell during handling (Carey et al. 2002; Lehmkuhl et al. 2004; Pyare et al. 2002). We excluded pellets that visibly included bait. Feces were stored in Whirlpacks and frozen. Field methods were approved by the University of Wyoming Institutional Animal Care and Use Committee and followed guidelines approved by the American Society of Mammalogists (Gannon et al. 2007).

Estimating diet with stable isotope and fecal analyses.—We estimated the diet of flying squirrels using both stable isotope and fecal analyses. Stable isotope analysis determines the relative contribution of assimilated diet items, whereas fecal analysis yields information on the diversity of recently digested food (Angerbjörn et al. 1994), in this case fungi, through the identification of cells and spores. Stable isotope analysis provides an index of the relative contribution of each item in the diet (Ben-David and Schell 2001; Phillips and Koch 2002) through measurement of isotopic values of the heavy isotopes of carbon (C) and nitrogen (N) in the tissue of an animal and those in the potential diet items. In this analysis it is important to account for the difference between the isotopic values of the consumer and its diet, or diet-consumer discrimination, which stems from chemical or physiological processes (Gannes et al. 1997). In addition, for reliable estimates of diet contribution, all diet items should have distinctive isotopic signatures, and the appropriate tissues must be used because tissues differ in turnover rates, diet–consumer discrimination, and assimilation of different diet components (Gannes et al. 1997). For example, isotopic values of blood serum correspond with diet during a relatively short period of time, 1–2 weeks prior to sampling, whereas red blood cells reflect diet during the previous 2–3 months (Hilderbrand et al. 1996; Hobson and Clark 1993). Additionally, isotopic discrimination and routing of the different components from each food source (i.e., carbohydrates, lipids, and proteins) to consumer body tissues can complicate the interpretation of isotopic signatures (Gannes et al. 1998). However, the use of stable isotope analysis in conjunction with fecal analyses should provide more reliable estimates of diet composition because fecal analysis may underestimate use of some diet items and thus may not sufficiently describe the importance of some foods (Thysell et al. 1997). Furthermore, unless fecal samples are assigned to individuals, it could provide a biased estimate of the population-level use of a specific resource (Felicetti et al. 2003). The combination of both stable isotope and fecal analyses has been used successfully to estimate the diet of pygmy raccoons (*Procyon pygmaeus*—McFadden et al. 2006), long-nosed bandicoots (*Perameles nasuta*—Thums et al. 2005), mycophagous marsupials (McIlwee and Johnson 1998), and frugivorous bats (Herrera et al. 2001).

For stable isotope analysis we dried samples of potential foods at 60°C for 48 h and then ground the samples into a fine powder using a mixer mill (Retsch MM 200; Glen Mills Inc., Clinton, New Jersey). A subsample was placed into a miniature tin weighing boat (4 × 6 mm) for combustion and sent in duplicate to the University of Wyoming Stable Isotope Facility. Data of $\delta^{13}C$ and $\delta^{15}N$ were generated with a Costech ECS elemental analyzer (Costech Analytical Technologies, Valencia, California) attached to a Finnigan DeltaplusXP mass spectrometer (Thermo Fisher Scientific, Inc., Waltham, Massachusetts) using PeeDee Belemnite (PDB) for the C standard and atmospheric air for N. Sample results were accepted if variance between the 2 subsamples did not exceed 0.15‰ and machine linearity did not deviate from 0.99 (Ben-David et al. 1997). Blood samples collected from flying squirrels were processed similarly. For those samples serum was not separated from blood cells because of the small amount of blood collected in capillary tubes. Consequently, isotopic values for squirrels represent their diet over the 6–8 weeks prior to sampling.

We used multivariate analysis of variance (MANOVA—Zar 1999) and post hoc Scheffé multiple comparisons to test for significant differences in stable isotope values among the various diet items and assessed differences in isotope signatures using a *K*-nearest neighbor randomization test (Rosing et al. 1998). We tested for differences in isotopic values of whole blood between sexes using a Student's *t*-test assuming equal variance (Zar 1999). The isotope data for all distinct diet items and those of flying squirrels were incorporated into a dual-isotope linear mixing model to determine the relative contribution of the various diet items to the overall squirrel diets in spring and autumn. For this analysis, we used the program SISUS (SISUS: Stable Isotope Sourcing Using Sampling—Erhardt 2007; Phillips and Gregg 2003) and analyzed data separately for spring and autumn. We corrected for diet–consumer discrimination by using a change of 1‰ for $\delta^{13}C$ and 3‰ for $\delta^{15}N$ (DeNiro and Epstein 1981; Kelly 2000; McCutchan et al. 2003; Peterson and Fry 1987). We ran the model as concentration-dependent by including data on the composition of tissues of the different diet items because of large differences in C:N ratios (Phillips and Koch 2002) in lichens (37.9:0.7), truffles, (47.2:4.3), conifer seeds (53.5:1.6), and soil macroinvertebrates (48.2:12.4—M. Ben-David, pers. obs.).

We thawed fecal samples and placed 2 small portions from each pellet on a microscope slide. One drop of potassium hydroxide (KOH) was added to 1 portion of the sample and mixed vigorously using a razor blade. The other portion of the sample was mixed similarly with a drop of Melzer's solution. KOH is a standard rehydrating medium for mounting fungal spores, and Melzer's can aid in identification of certain fungal genera by reacting with the spore walls and ornamentation to produce a color reaction (Castellano et al. 1989). We covered the 2 samples with an 18 × 18-mm coverslip and examined the slide using bright-field microscopy at 100×, 400×, and 1,000×. We identified food items in the entire field of view for each coverslip (Mitchell 2001). Fungal spores were identified to genus using a spore key (Castellano et al. 1989). We calculated frequency of occurrence of the fungal taxa as the percentage of occurrence in the total number of fecal samples each season (Mitchell 2001; Pyare et al. 2002). We compared the number of genera per fecal sample to season and sex of squirrels using analysis of variance (ANOVA—Zar 1999).

Surveys of food availability and sampling of potential foods.—We used 20-m line transects, pitfall traps, and 1 × 1-m plots to estimate food availability in the 3 habitat types. We used preliminary data from initial surveys during the 1st field season to determine the number of transects required to detect differences in food availability among habitats with a statistical power of 0.90. Using Cohen's (1988:274) effect size index *F*-test for analysis of variance and covariance (Smith and Harke 2001) and a type I error rate equal to the type II error rate (0.10—Smith and Harke 2001), we established that 15 transects in each habitat would provide sufficient power to detect differences among habitats. We chose to measure 3 transects in each sampled stand to account for within-stand heterogeneity and ensure that each stand was properly represented in our sample. Accordingly, we conducted 135 line-transect surveys (3 surveys per stand × 15 stands per habitat × 3 habitat types). We conducted 90 line-transect surveys during the spring and 45 during the autumn. The inequality in surveys between seasons was due to logistical constraint and was not related to the power analysis.

Transect locations within each habitat and the azimuths of transects were chosen randomly. At each site we established transect lines with a compass. A field tape was used to estimate presence and abundance and frequency of occurrence of *Vaccinium* spp., arboreal lichens, and epigeous fungi by walking along the line and estimating the length of interception for each of the diet items considered. Following the survey and along each transect, 5 pitfall traps were installed to sample nonvolant soil macroinvertebrates. Pitfall traps were 473-ml plastic cups buried in the earth with the lip flush with the surface of the forest floor. A plastic plate was placed over each cup to exclude rainwater and to mimic debris that invertebrates seek for shelter. The traps remained in place for approximately 3 days at which point the contents were emptied into individual plastic bags and frozen until identification and further analysis in the laboratory at the University of Wyoming. A total of 225 traps was collected from each of the 3 habitat types for a total of 675 traps. In the laboratory, macroinvertebrates were identified at least to order using a dissection microscope and guide books (Borror and White 1970; Kaston et al. 1978; White 1983; White and Borror 1998).

At each of the line-transect survey sites we conducted 2 truffle surveys (spring) during 2004 and 4 (spring and autumn) during 2005. At each end of each transect (all seasons) and 10 m from the center of the line on each side (spring and autumn 2005) we established a 1 × 1-m plot for estimating availability of spruce and hemlock seeds and truffles. The placement of the grid was initially selected at random but subsequently modified to avoid trees, rocks, and densely vegetated areas. We attempted to place the grids under logs (10–70 cm in diameter) whenever possible to maximize encounters with truffles. We recorded and removed downed woody debris and counted spruce and hemlock cones on each plot. We used methods similar to those reported by Clarkson and Mills (1994) to estimate truffle biomass and chose sampling periods that coincided with spring and autumn truffle blooms (Colgan 1997). We used hand rakes and slowly raked the soil, from the surface of the duff to the organic–mineral soil interface. Unearthed truffles were identified to genus and weighed for fresh biomass. In all, we surveyed 150 truffle plots. While digging, we also counted all uncovered earthworms. We collected subsamples of the various potential food items (i.e., lichens, truffles, mushrooms, berries, conifer seeds, new conifer growth, and soil macroinvertebrates) and froze them for use later in stable isotope analysis.

To determine whether food availability of all items, except invertebrates, differed among habitat types and between seasons, we used a nested 2-way ANOVA (Zar 1999), where the main effects were stand (clear-cut, 2nd-growth, or old-growth) and transect (nested within stand) and Scheffé multiple comparison tests. To test for differences in the abundance of soil macroinvertebrates among habitats and seasons, we averaged the number of invertebrates in each trap (5 traps/line-transect survey) along each line transect and similarly used a 2-way ANOVA with stand, season (spring or autumn), and transect (nested within stand) as main effects. Because many of the invertebrates we collected were too small to serve as food for flying squirrels, we categorized the invertebrates as small (<2 mm in length excluding append-

Table 1.—Mean (± *SE*) δ^{13}C and δ^{15}N values (‰) and elemental concentrations of potential food items for northern flying squirrels (*Glaucomys sabrinus griseifrons*) on Prince of Wales Island, Alaska. Number of samples included in the global mean for calculating the δ^{13}C and δ^{15}N is given by *n*. Letters represent significant differences ($\alpha = 0.05$) in both δ^{13}C and δ^{15}N as determined from MANOVA followed by Scheffé post hoc multiple comparisons (Zar 1999) and K nearest-neighbors randomization tests (Rosing et al. 1998).

Diet item	*n*	δ^{13}C	δ^{15}N	Significance all items	Significance pooled items	Elemental concentration	
						% δ^{13}C	% δ^{15}N
Epigeous fungi	32	−24.49 ± 0.30	4.37 ± 0.55	ag	a	47.2	1.7
Truffles	35	−27.32 ± 0.18	4.48 ± 0.29	b	b	47.2	1.7
Hemlock seeds	20	−25.26 ± 0.25	−1.70 ± 0.67	c		53.5	1.6
Spruce seeds	14	−26.82 ± 0.30	−1.25 ± 0.57	d		53.5	1.6
Mean seeds	34	−25.90 ± 0.24	−1.51 ± 0.45		c	53.5	1.6
Lichens	23	−21.26 ± 0.28	−3.87 ± 0.25	e	d	37.9	0.7
Berries	8	−31.40 ± 0.45	0.29 ± 0.81	f	e	49.3	1.7
Araneidae	28	−25.45 ± 0.17	5.67 ± 0.24	g		48.2	12.4
Scaphinotus angusticollis	30	−26.73 ± 0.15	3.75 ± 0.33	bg		48.2	12.4
Pterostichus spp.	28	−26.49 ± 0.26	4.76 ± 0.25	g		48.2	12.4
Diplopoda	30	−24.36 ± 0.14	2.78 ± 0.32	a		48.2	12.4
Harpaphe hadeniana	30	−23.69 ± 0.15	3.08 ± 0.42	ag		48.2	12.4
Earthworms	17	−26.61 ± 0.15	4.24 ± 0.22	a		48.2	12.4
Mean invertebrates	133	−25.26 ± 0.12	3.84 ± 0.16		f	48.2	12.4

ages), medium (2–10 mm), and large (>10 mm). We then repeated the analysis (with a 2-way ANOVA) only for large macroinvertebrates.

Results

We captured and processed a total of 36, 45, 30, 39, and 50 individual flying squirrels during spring 2003, 2004, and 2005, and autumn 2004 and 2005, respectively. During autumn 2004 and 2005, we captured 15 and 23 juveniles, respectively. From those individuals, we collected a total of 39 blood samples with enough volume for stable isotope analysis: 15 during spring 2004, 9 during spring 2005, and 15 during autumn 2005. These samples were collected from 12 adult females and 27 adult males. We also collected 23, 17, 11, 20, and 11 fecal samples from unique individuals during spring 2003, 2004, and 2005, and autumn 2004 and 2005, respectively, for a total of 51 spring samples and 31 autumn samples. Fecal samples collected during the spring field seasons were all from adult squirrels. During autumn seasons of 2004 and 2005, respectively, we collected 2 and 4 fecal samples from juveniles. We collected both feces and blood samples from 20 individuals in quantities sufficient for both analyses.

Table 2.—Relative contribution (± *SD*) to the diet of northern flying squirrels (*Glaucomys sabrinus griseifrons*) during spring 2003–2005 and autumn 2004–2005 on Prince of Wales Island, Alaska, for diet items. Proportions of diet item in overall squirrel diet were estimated by a concentration-dependent, dual-isotope linear mixing model. To account for trophic discrimination, we added 1‰ δ^{13}C and 3‰ δ^{15}N to each source value before incorporating into the model (SISUS). We removed berries and invertebrates from the concentration-dependent linear mixing model because berry seeds were not observed in feces.

Diet item	Relative contribution	
	Spring	Autumn
Epigeous fungi	0.04 ± 0.03	0.10 ± 0.07
Truffles	0.04 ± 0.03	0.09 ± 0.06
Conifer seeds	0.43 ± 0.03	0.33 ± 0.05
Lichens	0.47 ± 0.02	0.43 ± 0.04
Invertebrates	0.02 ± 0.01	0.05 ± 0.04

Diet estimates from stable isotopic analyses.—We found significant differences in the δ^{13}C and δ^{15}N of all diet items ($P < 0.05$; Table 1), except among several invertebrates and epigeous fungi ($P > 0.05$; Table 1). Also, we found significant effects of habitat on the isotopic values of the different macroinvertebrates (E. A. Flaherty and M. Ben-David, pers. obs.). To ensure that we did not introduce bias to our diet estimates because flying squirrels on POW rarely venture into clear-cuts and 2nd-growth stands (S. Pyare and W. P. Smith, pers. obs.), in subsequent analyses we used the isotopic signatures of items collected in old-growth habitats only. After pooling large-sized soil macroinvertebrates into a single group, invertebrates and epigeous fungi differed isotopically (Table 1). The isotopic signature of that soil macroinvertebrate group was not significantly different ($P > 0.05$) from that of earthworms (Table 1). Although spruce and hemlock seeds also differed ($P < 0.05$) isotopically (Table 1), we used average δ^{13}C and δ^{15}N values of these 2 items to create a category called conifer seeds to reduce the number of food items relative to sample size of squirrels. Finally, we did not include berries in the model because this food item was never identified in our fecal analysis. Therefore, we introduced the following diet categories into a dual-isotope, concentration-dependent linear mixing model: epigeous fungi, truffles, conifer seeds, lichens, and soil macroinvertebrates (Tables 1 and 2; Fig. 2).

Isotopic values in whole blood of flying squirrels captured during spring were more variable than those captured during autumn (Fig. 2), suggesting greater variation in diet among individuals. These differences were not related to the sex of

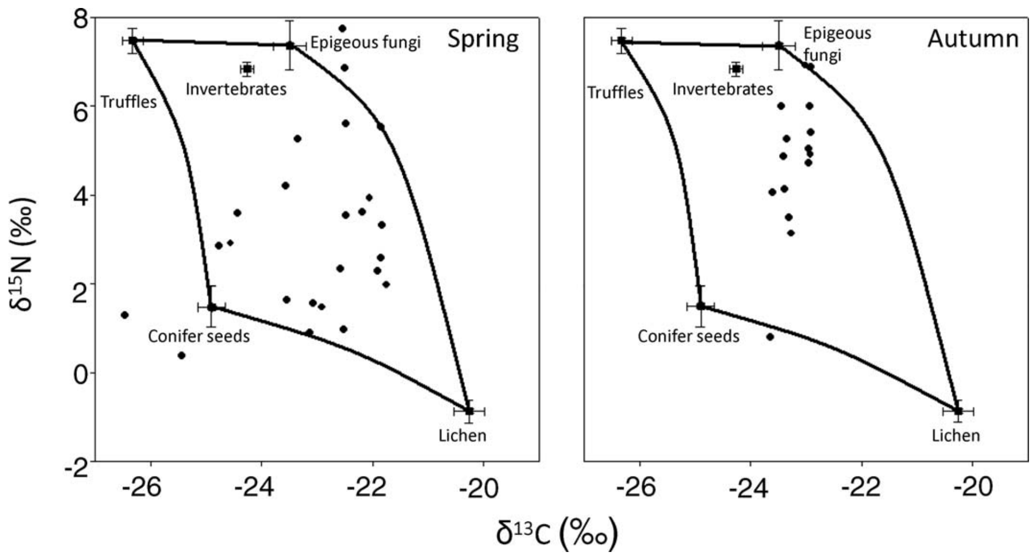

Fig. 2.—Distribution of isotopic values (mean ± *SE*) of potential foods and individual flying squirrels (*Glaucomys sabrinus griseifrons*) in spring and autumn on northern Prince of Wales Island, Alaska, for a concentration-dependent mixing model. The lines connecting potential food sources enclose the mixing space for the dual-isotope linear mixing models used to convert isotopic data to estimates of relative contribution. Although δ^{13}C and δ^{15}N values of several individual squirrels fell out of the mixing space, mean values were well within it in both autumn (δ^{13}C $= -23.20‰ \pm 0.069‰$; δ^{15}N $= 4.77‰ \pm 0.46‰$) and spring (δ^{13}C $= -23.07‰ \pm 0.26‰$; δ^{15}N $= 3.13‰ \pm 0.39‰$). We removed values for berries from the linear mixing model because berry seeds were not observed in feces.

the animals (δ^{13}C: $t_{38} = 2.06$, $P = 0.63$; δ^{15}N: $t_{38} = 2.06$, $P = 0.54$). During spring the δ^{13}C and δ^{15}N values of 4 of 24 individual squirrels fell out of the mixing space (Fig. 2), but only 1 of 15 did so in autumn. Such deviations likely resulted from our use of deterministic discrimination values rather than from pooling diet items (Table 1; Fig. 2). Using the population means, we estimated with the concentration-dependent mixing model that lichens had the largest contribution to the spring diets of northern flying squirrels (47%) followed by conifer seeds (43%), truffles (4%), and epigeous fungi (4%). During autumn, lichens again had the largest overall contribution to the diet (43%), followed by seeds (33%) and epigeous fungi (10%).

Fecal analysis.—We identified 10 different truffle genera and 2 taxa of epigeous fungi, Boletales and *Cortinarius* spp., with the latter more frequently consumed in autumn (Table 3). For truffles, *Elaphomyces* spp. were present in most samples (35–91%) and consumed in both seasons, followed by *Octaviania* (9–85%) and *Gymnomyces* (8–75%), which were largely consumed in autumn (Table 3). Of the other truffle genera, *Hydnotrya* was more prevalent in spring feces, whereas *Rhizopogon* and *Sarcosphaera* were consumed more frequently in autumn (Table 3). No significant difference ($F_{2,83} = 0.115$, $P = 0.736$) existed in the mean number of fungi genera consumed between males and females. Overall, truffles and lichens occurred most often in flying squirrel feces in both spring and autumn; truffle spores were found in 82–96% of the samples, whereas lichen material was found in 100% of the fecal samples during spring and $\geq$90% during autumn (Table 4). The next most common food item was

Table 3.—Percent occurrence of fungal spores in feces of northern flying squirrels (*Glaucomys sabrinus griseifrons*) on northern Prince of Wales Island, Alaska. Feces were collected from trapped individuals during spring 2003–2005 and autumn 2004–2005.

	% occurrence						
Fungus	Spring 2003	Spring 2004	Spring 2005	Spring $\bar{X}$ (*SE*)	Autumn 2004	Autumn 2005	Autumn $\bar{X}$ (*SE*)
Boletales[a]	0	0	0	0.0 (0.0)	50	29	39.5 (10.5)
Cortinarius[a]	0	0	8	2.7 (2.7)	75	24	49.5 (25.5)
Elaphomyces	91	77	83	83.7 (4.1)	35	53	44.0 (9.0)
Gautieria	13	0	0	4.3 (4.3)	0	0	0.0 (0.0)
Gymnomyces	26	0	8	11.3 (7.7)	75	53	64.0 (11.0)
Hydnotria	74	24	8	35.3 (19.9)	0	0	0.0 (0.0)
Hymenogaster	0	0	0	0.0 (0.0)	35	29	32.0 (3.0)
Hysterangium	0	12	0	4.0 (4.0)	5	0	2.5 (2.5)
Octavianina	30	6	17	17.7 (6.9)	85	59	72.0 (13.0)
Rhizopogon	0	6	0	2.0 (2.0)	15	29	22.0 (7.0)
Sarcosphaera	0	0	0	0.0 (0.0)	15	6	10.5 (4.5)
Tubers	9	0	0	3.0 (3.0)	0	0	0.0 (0.0)

[a] Epigeous fungi.

Table 4.—Percent occurrence of food items in feces of northern flying squirrels (*Glaucomys sabrinus griseifrons*) from old-growth Sitka spruce (*Picea sitchensis*)–western hemlock (*Tsuga heterophylla*) stands on northern Prince of Wales Island (POW), Alaska. These data are compared with diet estimates from fecal analysis of flying squirrels from other portions of Prince of Wales Island from Pyare et al. (2002). *n* = sample size of unique squirrels.

		% occurrence						No. truffle genera
	Season	*n*	Truffles	Lichens	Vegetation	Epigeous fungi	Invertebrates	
Northern POW	Spring 2003	23	96	100	91	0	0	7
	Spring 2004	17	82	100	77	0	12	5
	Spring 2005	12	83	100	92	8	0	5
	Spring $\bar{X}$ (*SE*)		87.0 (4.5)	100.0 (0.0)	86.7 (4.8)	2.7 (2.7)	4.0 (4.0)	5.7 (0.7)
	Autumn 2004	20	95	90	70	70	0	8
	Autumn 2005	17	82	94	82	53	12	7
	Autumn $\bar{X}$ (*SE*)		88.5 (6.5)	92.0 (2.0)	76.0 (6.0)	61.5 (8.5)	6.0 (6.0)	7.5 (0.5)
Pyare et al. 2002	Summer–autumn	150	50.4	27.0	55.2	36.1	4.4	3.0

vegetation, which was equally consumed in both seasons, followed by epigeous fungi, which were largely consumed in autumn (Table 4). Soil macroinvertebrates, specifically wing parts, also were present, although infrequently, in feces from both seasons (Table 4).

Food availability.—In general, availability of potential food items for flying squirrels did not differ ($P > 0.05$) seasonally, except for hemlock and spruce seeds and *Vaccinium* spp. (Table 5), which were approximately 2.5 ($\bar{X} \pm SE$, 261 ± 22 cones in spring versus 104 ± 29 cones in autumn) and 2.9 (13 ± 4 cones in spring versus 5 ± 2 cones in autumn) times more available during spring in old-growth for hemlock and spruce cones, respectively, whereas *Vaccinium* was more available in autumn (526.97 ± 75.35 cm in spring versus 714.13 ± 95.20 cm in autumn; Table 5). Transects in old-growth forest stands had 30 times more spruce cones than managed habitats (Table 5). Hemlock cones were 20 times more abundant in old-growth compared with 2nd-growth stands and 9 times more abundant than in clear-cuts (Table 5). Similarly, *Vaccinium* spp. were 2–3 times more common in old-growth (Table 5) than in managed habitats (Table 5). We found no truffles in 2nd-growth stands; truffles were about 2.5 times more abundant in old-growth plots than in clear-cuts ($P = 0.034$). Similarly, there was twice as much lichen in old-growth stands than in clear-cuts and 9 times more than in 2nd-growth stands (Table 5; $P = 0.002$). We found no difference ($P > 0.05$) in the abundance of epigeous fungi among habitats (Table 5). In all surveys of truffles we identified only the genus *Elaphomyces*, which in some of the 1-m^2 plots in old-growth forest reached a total biomass of 32 g.

We collected >3,700 soil macroinvertebrate specimens, which were identified to 13 taxa: Acari, Araneida, Coleoptera, Collembola, Diplopoda, Diptera, Gastropoda, Hymenoptera, Isopoda, Isoptera, Oligochaeta, Opiliones, and Scolopendromorpha. Acarina and Coleoptera were the most abundant orders, comprising >33% and >23% of macroinvertebrates sampled, respectively. Traps along transects in 2nd-growth stands had significantly more soil macroinvertebrates than the other 2 habitats; clear-cuts had the fewest invertebrates ($F_{2,675} = 10.225$, $P = 0.006$; Fig. 3).

After excluding the small (Araneidae, Collembola, Acari, Diptera, and Opiliones) and medium-sized (Buprestidae [Coleoptera], Curculionidae, Isoptera, Gastropoda, and Hymenoptera) invertebrates, analyses revealed that of the large invertebrates (Araneidae [spiders], Chilopoda [centipede], *Scaphinotus angusticollis* and *Pterostichus* spp. [Coleoptera],

Table 5.—Abundance, measured as biomass (g), count (no.), or length of transect intersected (cm) of potential food items from spring (2004–2005) and autumn (2005) surveys on Prince of Wales Island, Southeast Alaska. In each of 15 stands per habitat we conducted 3 surveys for a total of 135 line-transects. We report *P*-values (statistically significant in bold) from a nested ANOVA for comparing availability of food between stands and differences in availability between spring and autumn.

	Habitat ($\bar{X} \pm SE$)							
	Clear-cut		Second-growth		Old-growth		Comparison	
Food item	Spring	Autumn	Spring	Autumn	Spring	Autumn	Between-stands *P*-value	Between-seasons *P*-value
Epigeous fungi (cm)	0.83 ± 0.46	0.33 ± 0.23	0.20 ± 0.20	1.13 ± 0.81	0.23 ± 0.18	2.53 ± 1.43	0.46	0.42
Truffles (g)	0.77 ± 0.57	0.55 ± 0.36	0	0	1.95 ± 0.61	1.03 ± 0.77	**0.00**	0.28
Hemlock cones (no.)	21.55 ± 13.70	11.57 ± 4.32	3.21 ± 1.73	20.55 ± 11.87	261.25 ± 22.07	103.98 ± 28.94	**0.01**	**0.00**
Spruce cones (no.)	0.35 ± 0.13	0.12 ± 0.06	0.27 ± 0.12	0.32 ± 0.12	12.93 ± 3.84	4.47 ± 2.15	**0.03**	**0.046**
Lichens (cm)	1.10 ± 0.68	2.47 ± 1.55	0.50 ± 0.50	0	3.83 ± 1.53	1.13 ± 0.84	**0.03**	0.38
Vaccinium (cm)	198.50 ± 42.60	249.87 ± 59.57	511.50 ± 18.63	72.2 ± 18.63	526.97 ± 75.97	714.13 ± 95.20	**0.00**	**0.04**
New spruce growth (cm)	15.80 ± 8.92	0	7.67 ± 5.46	0	13.33 ± 13.33	0	0.85	0.07

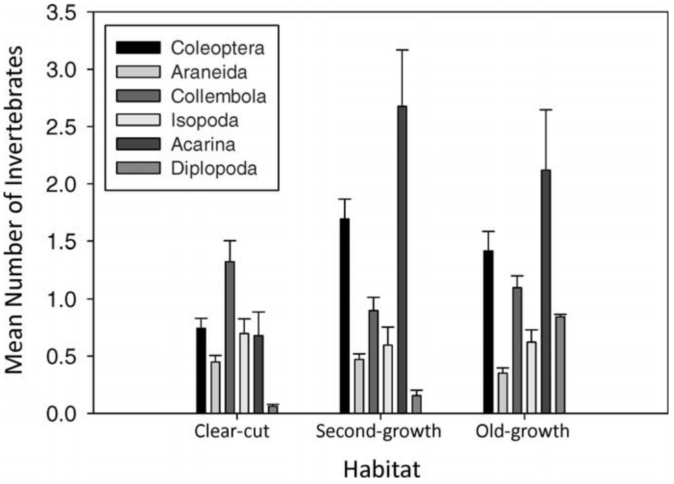

FIG. 3.—Mean (± *SE*) number of soil macroinvertebrates captured in pitfall traps along 135 transects on Prince of Wales Island, Alaska, in the 3 habitat types: clear-cut, 2nd-growth, and old-growth. The 6 major groups of invertebrates (>5 individual invertebrates were sampled) are shown.

Diplopoda [e.g., a small, black millipede and *Harpaphe hadeniana*], and Oligochaeta [earthworms]), only *S. angusticollis* and *Pterostichus* spp. were more abundant in 2nd-growth stands than in other habitats ($F_{2,673} = 9.539$, $P = 0.008$, and $F_{2,673} = 9.446$, $P = 0.008$, respectively; Fig. 4). Oligochaetes were more available in clear-cuts ($F_{2,673} = 9.266$, $P = 0.008$; Fig. 4). No consistent seasonal differences in the number of large soil macroinvertebrates were found among the 3 habitat types (Fig. 4).

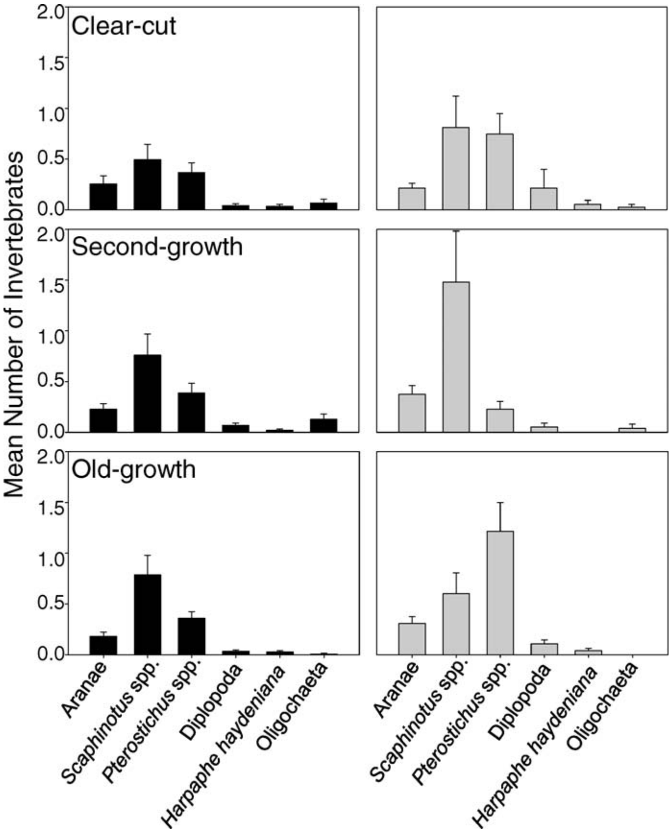

FIG. 4.—Mean (± *SE*) number of 6 large (>10 mm in body length) soil macroinvertebrates sampled in pitfall traps on Prince of Wales Island, Alaska, in the 3 habitat types clear-cut, 2nd-growth, and old-growth, in the spring (black bars) and the autumn (gray bars). These represent the most likely invertebrates consumed by northern flying squirrels (*Glaucomys sabrinus griseifrons*).

DISCUSSION

Both stable isotope and fecal analyses revealed that arboreal lichens, conifer seeds, and fungal sporocarps were the main dietary items consumed and assimilated by *G. sabrinus* on POW. Both analyses also highlighted the increased importance of fungi, especially epigeous fungi, during autumn, and lichens during spring. Similarly, both methods revealed that vegetation, likely in the form of conifer seeds, and soil macroinvertebrates were consumed by squirrels, whereas berries were not. Finally, using both methods we did not detect any differences in diet between the sexes. The main disagreement between the 2 methods was the contribution and importance of truffles. Although examination of stable isotope data suggested that squirrels assimilated few nutrients from truffles (especially N), truffle spores were among the most frequent diet items in feces. It is possible that squirrels frequently consumed truffles but little N was assimilated from this food source.

The discrepancy between results from stable isotopes and fecal analyses potentially could be explained by our use of data only for *Elaphomyces* spp. to represent the isotopic values of all truffles. It is possible that other truffles consumed by *G. s. griseifrons* have different isotopic signatures and that their inclusion would have changed our results. Several animals during both spring and autumn had values that were not included in the isotopic mixing space. Although such misalignment of consumer and diet items could result from effects of habitat use (E. A. Flaherty and M. Ben-David, pers. obs.), diet quality, elemental routing, tissue-turnover rates, and variation in the length of the assimilation period (Karasov and Martinez del Rio 2007), this misalignment most commonly occurs when potential foods are excluded (Newsome et al. 2007). Nonetheless, because relatively few animals were misaligned with the mixing space, and overall the isotopic signatures of both mushrooms and truffles were similar (especially in ^{15}N), it is not likely that our results would have changed dramatically had we sampled other truffles. It is important to note here that we did not analyze other truffle species for stable isotopes because we encountered none during our surveys.

Alternatively, it is possible that although *G. sabrinus* consumes large quantities of truffles, when these squirrels are limited to *Elaphomyces* spp. they may assimilate little of this resource. Past research indicates that movements, population density (Gomez et al. 2005; Pyare and Longland 2002; Waters and Zabel 1995), survival, and recruitment (Lehmkuhl et al. 2006) of *G. sabrinus* are correlated directly with availability of truffles. However, when consumed as the sole food source, *Elaphomyces* has minimal nutritional value for *G. sabrinus* (Cork and Kenagy 1989), other small mammals are unable to maintain body mass when consuming this resource (Cork and Kenagy 1989; Dubay et al. 2008), and in areas and seasons with high species richness, *G. sabrinus* will not consume *Elaphomyces* (Meyer et al. 2005). Thus, the dominance of *Elaphomyces* in our sampling plots and in the feces suggests that flying squirrels on POW likely assimilated few nutrients from the majority of truffles they consumed. Moreover, past work indicates that most of the N found in truffles is indigestible by flying squirrels and other small mammals (Claridge and Cork 1994; Claridge et al. 1999; Cork and Kenagy 1989); other nutrients, such as potassium, phosphorous, and vitamin D, that occur in *Elaphomyces* may explain the preference for this diet item (Dubay et al. 2008). Similarly, although low in N, fiber, lipids, and other important nutrients (Dubay et al. 2008), arboreal lichens are high in calcium (Ca) and have high digestibility (Robbins 1987). It is possible that flying squirrels consume high amounts of lichen to maintain Ca uptake and include other diet item such as conifer seeds, invertebrates, and epigeous fungi to mitigate the low availability of N and other nutrients in truffles and lichens; mixed diets are common among mycophagist mammals (McIlwee and Johnson 1998; Orrock and Pagels 2002).

Our dietary estimates from both fecal and isotope analyses contrast with some findings of previous research in Southeast Alaska (Pyare et al. 2002) and are more similar to diets reported for *G. sabrinus* in other parts of its range (Smith 2007), where typically 100% of fecal samples contained fungal spores (Rosentreter et al. 1997; Wheatley 2007). Pyare et al. (2002) identified truffle spores in 50.4% of their autumn samples compared to ≥82% in this study. Furthermore, our analysis indicated that flying squirrel diets in the northern part of POW contained a greater diversity of truffle genera than reported by Pyare et al. (2002). They identified only 3 genera (mostly *Elaphomyces*, and some *Hymenogaster* and *Sclerogaster*), whereas we identified a minimum of 5–8 genera. Similarly, whereas both stable isotopes and fecal analyses identified lichens as an important food source, Pyare et al. (2002) reported that only 27% of their samples contained lichens. In addition, Pyare et al. (2002) encountered relatively few epigeous fungal spores in their sample of squirrel feces, whereas we estimated this to be an important resource for flying squirrels in autumn. It is possible that the differences in dietary estimates between the 2 studies on POW stem from timing of sampling; Pyare et al. (2002) collected their samples mainly during summer, whereas we sampled squirrels during spring and autumn. Future studies that include all seasons may better elucidate the factors responsible for the divergent dietary estimates of these 2 studies.

The higher variation in diet among individual squirrels during spring corresponded with higher consumption of conifer seeds. In conifers a new crop of cones is produced in summer (Koenig and Knops 2000), and although flying squirrels rarely harvest and cache cones in middens like red squirrels (*Tamiasciurus hudsonicus*—Mowery and Zasada 1984), they likely are able to reach these newly developing cones in autumn. Therefore, it is surprising that conifer seeds were more prominent in spring than autumn diets. We suspect that the higher consumption of conifer seeds during spring is a function of lower availability of truffles, other than *Elaphomyces*, during this time of year. Although we did not encounter any such truffles in our plots in either spring or autumn, the higher occurrence of *Gymnomyces*, *Hymenogaster*, *Octavianina*, *Rhizopogon*, and *Sarcosphaera* in feces during autumn suggests that they were more abundant at that time of year. Given this observation and equal abundance of mushrooms during spring and autumn, it is surprising that squirrels consumed more mushrooms during autumn. Whether autumn mushrooms provide better nutritional value for flying squirrels than those developing in spring is unknown and merits further investigation.

Our dietary data are based on samples collected from animals captured in old-growth stands only. It is possible that we would have drawn different conclusions had we sampled flying squirrels in 2nd-growth stands. Past work has indicated that 2nd-growth habitats can support populations of flying squirrels in other parts of their range (Ransome et al. 2004; Ransome and Sullivan 2003; Wheatley et al. 2005). Nonetheless, Smith (2007) cautioned that population density may not be a reliable indicator of habitat quality. Results from a study on POW comparing flying squirrel use of peatland–mixed conifer to old-growth habitats initially indicated that the number of reproductive females was greater in peatland–mixed conifer stands than in old-growth stands and that recruitment was only slightly lower in the former (Smith and Nichols 2003). However, later population modeling indicated that peatland-mixed conifer stands actually functioned as population sinks (Smith and Person 2007). Moreover, perceptual range and fine-scale movement data (Flaherty et al. 2008), energetics measurements related to the costs of running versus gliding (E. A. Flaherty, pers. obs.), telemetry data, and dispersal modeling (S. Pyare and W. P. Smith, pers. obs.) indicate that northern flying squirrels on POW actively avoid 2nd-growth stands.

Our results suggest low availability of potentially critical food items in managed habitats, which may constrain dispersal of *G. sabrinus* across clear-cut and 2nd-growth habitats. Conifer seeds, truffles, and *Vaccinium* spp. were all significantly more abundant in old-growth habitat. Furthermore, the hemlock and spruce cones we sampled in clear-cuts were likely remnants of the once present old-growth stand and consequently are likely only available for a short time postlogging. Similarly, although we encountered truffles in clear-cut plots, it is unclear how available this resource is in young regenerating stands, because we found truffles only where the roots of tree stumps had not completely died; we recorded no truffles in clear-cuts older than 2–3 years postharvest. Except for 1 sporocarp uncovered while digging a pitfall trap in a >40-year-old stand, we found no truffles in

2nd-growth habitat. Carey et al. (2002) suggested that harvest plans that leave legacy (i.e., old-growth trees) in managed stands will increase the persistence of truffles. This has not been the prescribed management practice in more than 4 decades of logging POW (United States Department of Agriculture Forest Service 1997), nor is it clear if legacy retention will achieve this objective because of the vulnerability of leave trees to windthrow (Concannon 1995). We do acknowledge that our truffle survey technique likely was inadequate to detect the majority of genera because we were only able to uncover 1 of 8 consumed by the squirrels. Nonetheless, the low occurrence of 5 of these genera in the feces of squirrels (especially in spring) suggests that they were rare even in old-growth habitats. Future work should consider using a trained, truffle-detecting dog for line-transect surveys or increasing survey intensity.

Availability of mushrooms and lichens, both important diet items during autumn, was similar in old-growth and clear-cut stands. Nonetheless, both were lower in 2nd-growth habitats that comprise the majority of the managed landscape on POW because of declining frequency of timber harvest in recent years. That lichens were less available in 2nd-growth stands suggests that the lichens found in clear-cuts likely remained from felled trees during the harvest rather than having been blown in from adjacent old-growth stands. Lichens surveyed in clear-cuts were desiccated and appeared older than those surveyed in old growth. Consequently, this resource (like conifer cones) will be available only during a brief period after logging.

Conversely, soil macroinvertebrates, especially those larger than 10 mm, were more abundant in managed habitats than in old-growth stands. Soil moisture is presumably lower in clear-cuts and some 2nd-growth habitats because of the absence of a developed canopy, which affects decomposition, evaporation, and other soil characteristics that influence habitat use by soil invertebrates (Niemelä 1997). Given our relatively high estimates of proportion of soil macroinvertebrates from stable isotope analyses during both spring and autumn, it appears as though flying squirrels dispersing through the managed matrix potentially could replenish depleted energy stores by consuming invertebrates. However, examination of feces indicated that none of the most abundant invertebrates actually were consumed by flying squirrels. Rather, invertebrate remains in feces were small wing parts likely from flies that were consumed coincidentally when flying squirrels fed on mushrooms. Thus, the high abundance of soil macroinvertebrates in managed habitats likely would not improve the foraging success of dispersing flying squirrels.

The high proportion of 2nd-growth stands in the managed matrix on POW, the lack of truffles in those stands and their limited temporal availability in clear-cuts, and the relatively low availability of other alternative foods likely will result in low encounter rates by dispersing squirrels. Low encounter rates with food resources will cause squirrels to commit additional time to foraging in this high-cost environment. The extent to which increased foraging time directly influences dispersal success is unclear, but increasing search time will presumably increase predation risk because flying squirrels could not launch into evasive glides in clear-cut and 2nd-growth stands that lack tall trees while increasing energy expenditure in unfamiliar and structurally deficient habitats (E. A. Flaherty, pers. obs.). Further work is needed to explore the relationship between predation risk and stand age in managed forests.

In conclusion, despite the varied diet of *G. sabrinus* in Southeast Alaska, availability of potential foods is low in managed habitats compared to old-growth forest. Therefore, continued loss of such stands from timber harvest might cause further decline in overall food availability across managed landscapes. Food resources, which were significantly lower in managed stands on POW, are among the most significant factors limiting populations of *G. sabrinus* (Lehmkuhl et al. 2006; Ransome and Sullivan 1997; Smith 2007) and affect reproduction, survival, recruitment, space use, habitat core-use areas, and home-range size (Holloway 2006; Menzel et al. 2004). Because natal and adult breeding dispersal in managed rain forests on POW require that flying squirrels move long distances (Smith et al., in press), these animals likely will encounter clear-cut and 2nd-growth stands. Although these high-cost habitats might not completely eliminate flying squirrel dispersal, the substantially lower permeability of managed stands (Smith et al., in press) could significantly reduce survival and dispersal success (Ransome and Sullivan 2003).

Low use of early seral habitats by flying squirrels could reduce the dissemination of fungal spores into managed stands (Pyare and Longland 2001). Although low soil moisture (Harvey et al. 1979; Luoma et al. 1991) and decreased abundance of coarse woody debris are responsible for the low production of truffles in the managed stands (Amaranthus et al. 1994; Clarkson and Mills 1994), lower inoculation rates of tree roots (Pyare and Longland 2001) could add to the slower rates of fungal establishment in early seral forests. Because the small mammal fauna of Southeast Alaska is depauperate (MacDonald and Cook 1996), few alternative mycophagists are available to serve as spore vectors. Thus, the persistence of flying squirrels in managed landscapes may be necessary to ensure the timely inoculation and reestablishment of colonies of ectomycorrhizal fungi that promote forest development (Carey et al. 1999). Arguably, ensuring the proliferation of mycorrhizal fungi in managed landscapes is important for regenerating timber resources. Nevertheless, without adequate food resources, the ability of flying squirrels to replenish energy stores while dispersing across 2nd-growth and clear-cut habitats may be limited, and populations in managed landscapes are at risk of becoming isolated. Without dispersal, the persistence of *G. sabrinus* in managed landscapes is uncertain (Smith and Person 2007).

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Stand age and habitat influences on salamanders in Appalachian cove hardwood forests

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Abstract

We surveyed cove hardwood stands aged 15, 25, 50, and ≥85 years following clearcutting in the southern Appalachian Mountains of northern Georgia to assess the effects of stand age and stand habitat characteristics on salamander communities using drift-fence array and pitfall methodologies from May 1994 to April 1995. Over a 60,060 pitfall trapnight effort, we collected 3937 salamanders represented by *Desmognathus aeneus*, *Desmognathus monticola*, *Desmognathus ocoee*, *Desmognathus quadramaculatus*, *Eurycea bislineata*, *Gyrinophilus porphyriticus*, *Pseudotriton ruber*, *Plethodon glutinosus*, *Plethodon serratus*, and *Notophthalmus viridescens*. Analysis of covariance with pitfall array to stream distance as the covariate showed that salamander species richness and diversity measures and numbers of *Desmognathus aeneus* and *Desmognathus ocoee* were highest in stands ≥85 years. *Eurycea bislineata* and *Plethodon glutinosus* were more abundant in stands ≤50 years old than in stands ≥85 years. Within cove hardwood stands, species richness and diversity measures and relative abundances of *Desmognathus* spp. and *Gyrinophilus porphyriticus* were negatively correlated with distance to stream. Species richness and diversity were positively correlated to amounts of emergent rock. Species richness, diversity and relative abundances of *Desmognathus* spp. were correlated with basal area within stands and extent of connected mesic, cove hardwood habitat and amount of cove habitat within 1 km radius among stands. *Eurycea bislineata* was negatively correlated with landform index, a measure of surrounding landform sheltering, and *Plethodon glutinosus* was positively correlated with elevation in cove hardwood stands. Our research indicates stand age is an important factor in explaining the abundance and community composition of salamanders in southern Appalachian cove hardwood communities. Because southern Appalachian woodland salamander communities are slow to recover and are substantially changed following disturbances such as clearcutting, populations in small, isolated cove hardwood stands might be more vulnerable to extirpation or may require longer recovery times than those in larger coves. Managers may need to assess habitat features such as cove extent and habitat connectivity to minimize impacts on these taxa by forest management activities in southern Appalachian cove hardwood communities. Published by Elsevier Science B.V.

Keywords: Clearcutting; Cove hardwoods; Habitat connectivity; Salamanders; Southern Appalachians

1. Introduction

Salamander communities constitute an important ecological component of Appalachian ecosystems, often exceeding the combined biomass of other terrestrial vertebrates (Burton and Likens, 1975;

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Hairston, 1987). In the southeastern United States, species diversity of woodland salamanders (family: Plethodontidae) reaches its zenith in the southern Appalachians (Martof et al., 1980, Wilson, 1995). Accordingly, recent attention has focused on the negative impact of clearcutting on woodland salamanders within the region (Ash, 1988, 1997; Petranka et al., 1993, 1994; Harpole and Haas, 1999). In the short-term, clearcutting in the southern Appalachians produces habitat conditions unfavorable for salamanders by increasing forest floor temperatures (Johnson et al., 1985) and reducing leaf litter depth and microsite moisture (Ash, 1995). Recovery times for salamander communities following clearcutting may vary depending upon salamander community composition prior to timber harvest (Ash, 1988) and upon site characteristics such as elevation, aspect, and plant community (Diller and Wallace, 1994; Ford et al., 1999; Harper and Guynn, 1999). Petranka et al. (1993, 1994) suggest that salamander communities require 50–80 years for full recovery to pre-harvest conditions following clearcutting in the southern Appalachians. Conversely, others believe that recovery times might be much less in the southern Appalachians (Ash, 1997; Harper and Guynn, 1999). Central to the issue of salamander recovery following timber harvest are both the short-term and long-term fate of on-site salamanders, and if salamanders on-site or in nearby surrounding forests provide the recolonizing source as clearcuts mature.

Questions regarding sampling methodologies and study site selection have plagued researchers studying salamander response to timber harvest in the southern Appalachians (Ash and Bruce, 1994; DeMaynadier and Hunter, 1995; Ash and Pollock, 1999; Petranka, 1999). For example, proximity to water is an important factor explaining salamander community assemblages in the southern Appalachians (Bruce, 1996) and may serve as a strong confounding factor in examining stand age to salamander community relationships, particularly for those in the genus *Desmognathus*. Also, timing of sampling effort throughout the year can have profound influences on study results, as above-ground terrestrial activity patterns vary from species to species (Petranka, 1998).

To better assess the relationship of clearcutting to salamanders in the southern Appalachians, we initiated a study of salamander abundance among a well-matched, chrono-sequence of cove hardwood stands that were similar in most characteristics except stand age. Additionally, we sought to examine micro- and stand-level habitat factors both dependent and independent of stand age that might be important to salamanders in the southern Appalachians.

2. Methods

In the winter of 1994, we selected 13 cove hardwood stands for study in the Brasstown Ranger District (BRD) of the Chattahoochee National Forest. Stands were located in the Cooper Creek watershed (seven stands), the Miller Cove watershed (two stands) and the Wolf Creek watershed (three stands) in Union County, Georgia and the Hiwassee River headwaters (one stand) in Towns County, Georgia using US Forest Service Continuous Inventory and Stand Condition (CISC) data followed by site visits and assessments. To control for variables other than stand age that might influence salamander communities, criteria used for site selection included: forest cover type recorded as yellow-poplar *(Liriodendron tulipifera* L.) or yellow poplar-northern red oak (*Quercus rubra* L.)–white oak (*Q. alba* L.), elevations between 700 and 1200 m, northerly site exposure, presence of a concave cove landform, and absence of a dense ericaceous shrub layer. We also chose to wholly contain our study within the BRD for logistic reasons and to minimize potential differences in woodland salamander assemblages that can occur within small geographic areas in the southern Appalachians. Actual study site elevations ranged from 730 to 1030 m.

In addition to yellow-poplar and northern red oak, other common overstory species were yellow buckeye (*Aesculus octandra* Marshall), basswood (*Tilia americana* L.), and white ash (*Fraxinus americana* L.). A rich herbaceous understory was present at each site (Ford et al., 2000). Soils at each site were deep, well drained stony loams with a deep humus layer in the Porter and Saunook Series (Wharton, 1989; Cabe, 1995).

We constructed five drift-fence arrays with pitfall traps in three each of these stands aged approximately 15, 25, and 50 years and in the four stands $\geq$85 years old. Stands $\leq$25 years were regenerated following clearcutting under US Forest Service stewardship.

Exact history of older stands is unknown, but the dominance of yellow-poplar, a shade-intolerant species, in an even-aged structure with little or no large downed woody debris, suggests the 50-year-old stands originated following clearcutting or heavy selection harvests. Stands ≥85 years originated after the widespread logging of the southern Appalachians that occurred at the turn of the century (Smith and Linnartz, 1980; Johnson et al., 1993). Portions of two of the older stands in our study, on the Cooper Creek Scenic Area and the Sosebee Cove Scenic area were never logged (Wentworth, personal communication; Duffy and Meier, 1992). Although cove hardwood communities are abundant in the southern Appalachians and the BRD, it is important to note that three factors limited our cove hardwood stand choices. Potential cove hardwood study areas identified in CISC data that were within designated wilderness areas were administratively excluded from this study. Site visits and assessments often revealed errors in CISC overstory type coding or the presence of dense ericaceous shrub growth usually indicating a lower slope position and a more riparian-influenced woodland. Lastly, 50-year-old cove hardwood stands that originated in the 1940s were relatively uncommon on the BRD, as were cove hardwood stands ≤15 years of age.

In each of 13 stands, we installed five drift-fence arrays. Drift-fence arrays with pitfalls are effective methodologies for sampling herpetofauna (Gibbons and Semlitsch, 1982; Mitchell et al., 1997) and shrews (Kirkland and Sheppard, 1994), another facet of this study reported elsewhere (Ford et al., 1997). Array 1 was placed in the center or ravine of the cove, with arrays 2, 3, 4, and 5 installed 50 m away in SE, SW, NW, and NE directions, respectively, from the center point of array 1 (Fig. 1). Because the distance between arrays was large relative to the limited movements and small home ranges of many woodland salamanders (Madison, 1969; Holomuzki, 1982; Mathis, 1991; Petranka et al., 1993, 1994; Petranka, 1998), we considered each array to be an independent sampling unit. Individual arrays consisted of four, 3 m long × 61 cm high aluminum flashing arms arranged in an X (Fig. 1). The bottom of the flashing was buried approximately 20 cm below the soil surface. One pitfall was placed on either side of the flashing near each arm end, and one in each of the four intersections at the center of the array (Fig. 1). Pitfall traps were plastic 946 cm^3 drink cups buried flush with the ground and against the sides of the flashing. Each pitfall cup was filled one-third its volume with 10% formalin to preserve specimens. Trapping was conducted for 7 days at the beginning of each monthly lunar cycle from May 1994 to April 1995, except February 1995 when harsh winter weather uncharacteristic of north Georgia precluded access to the study sites. All collection periods experienced at least one precipitation event over their duration. Following each collection period, salamanders were identified to species and reposited at the University of Georgia Museum of Natural History. Nomenclature follows Petranka (1998).

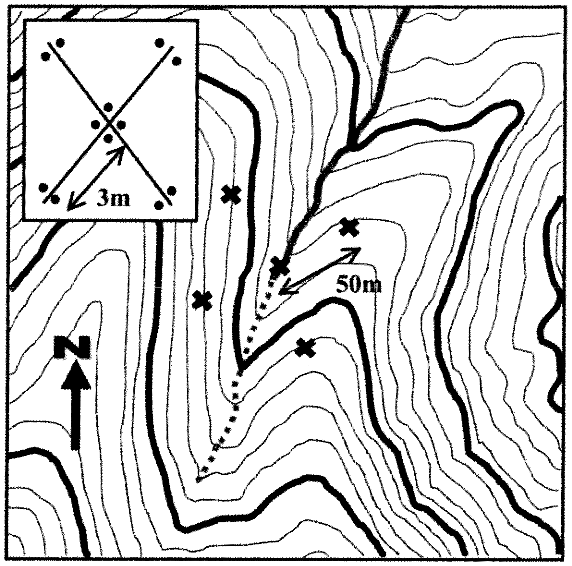

Fig. 1. Typical pitfall drift-fence array design for salamander collection in cove hardwood stands on the Chattahoochee National Forest, Georgia, 1994–1995. Inset depicts pitfall placement at individual drift-fence array (see text for complete description).

Because factors that influence salamander distribution and abundance vary among stands irrespective of overstory age, we measured nine micro-habitat variables at drift-fence arrays and calculated seven stand-level variables and topographic metrics for each study site. Within 10 m circular plots around each drift-fence array, we measured litter depth (LITTER), loose soil depth (SOIL), percent canopy cover (CANOPY), amount and types of coarse woody debris, and amount of emergent rock (ROCK). LITTER and SOIL were measured at one randomly chosen point in each quadrant of each circular plot. LITTER was recorded by brushing away leaves in a small area and measuring the height of the litter to a discernible O layer. SOIL was measured as the distance a piece of construction rebar could be driven in the soil until rock was encountered. CANOPY was calculated using a spherical densiometer. Over each plot, total coarse woody debris area (CWD) was calculated by tallying length and prostrate height at midpoint of all down woody debris ≥ 10 cm diameter. Coarse woody debris ≥ 60 cm prostrate height was considered large and a separate tally of that class was made as well (LGCWD). Each piece of coarse woody debris counted was assessed for decay class (DECAY) following criteria set by Hardt (1993) for the southern Appalachians. Area of emergent rock (ROCK) per plot was calculated by measuring emergent rock length and width above the soil surface. The distance from each array to the nearest seep, spring, creek, or other water source (DISTANCE) was measured to the nearest meter.

Parameters measured across each stand included elevation (ELEV), landform index (LFI), plot surface shape (PSS), aspect (ASPECT), cove area surrounding each study site (COVEHA), total cove area within 1 km of each study site (COVEKM), and stand basal area (BASAL). With the exception of BASAL, these variables were computed from digital terrain models using analytical procedures in ARC/INFO GRID® geographic information software. LFI indicates the degree to which a site is sheltered by surrounding landforms; lower values (≤ 0.15) reflect a less-sheltered position (McNab, 1993). The PSS characterizes the degree of convexity or concavity by comparing the elevation at the center of each study site to elevations of the immediate surrounding landscape (approximately 1 ha). Values typically range from -1.0 (highly concave) to 1.0 (highly convex). Aspect values were linearized using the formula $(1 - \text{cosine (aspect in degrees)}) + (1 - \text{sine (aspect in degrees)})$ so that northeasterly aspects had the lowest values and southwesterly aspects the highest. COVEHA was defined as the area with an $\text{LFI} > 0.24$ (highly sheltered sites), or $0.25 > \text{LFI} > 0.15$ and $\text{PSS} < -0.025$ (less sheltered, but concave sites) at each study stand. Threshold values for LFI and PSS were determined based on prior experience of one of the authors (Odom, 1996) and others (McNab, 1993, 1996) in using these indices to quantify topography in the southern Appalachians. To compute COVEKM, the total area defined as "cove" within 1 km of each study site was summed using proximity and overlay functions in ArcView® Spatial Analyst. BASAL was calculated using a 10-factor handheld prism ($\text{m}^2\,\text{ha}^{-1}$) at each drift-fence array and averaged across each stand (Grosenbaugh, 1952), as it is not a particularly meaningful fine-scaled, micro-habitat variable.

Salamander collections were summed by species across all months. Species richness and Shannon's diversity (H' [$\log_e$]) was calculated for each drift-fence array (Pielou, 1966). Although Shannon's diversity calculations often are biased conservatively with actual H' less than observed H' (Kempton, 1979), these measures are used widely in terrestrial ecology research (Magurran, 1988). To test for differences among stand age and to account for the influence of the proximity of water to drift-fence arrays, we analyzed salamander collection data by species richness and diversity, and by species abundance values using analysis of covariance (ANCOVA) (Steel and Torrie, 1980). The experiment-wide error rate was set at $P = 0.05$ and, using Bonferroni's correction method, $P = 0.003$ was used for individual tests (Sokal and Rohlf, 1995). Mean values of species richness and diversity and species abundance values were reported using treatment means adjusted for the covariate DISTANCE (SAS, 1991). To address questions of salamander recovery following timber harvest, we used orthogonal contrasts of treatment means to test for differences between young-aged stands (15- and 25-year-old stands) that originated following modern clearcutting versus the oldest stands (≥ 85 years), between young-aged stands and 50-year-old stands, and between 50- and ≥ 85-year-old stands (Montgomery, 1991; SAS, 1991). To assess the importance of micro-habitat variables on salamanders, Spearman's rank correlations were performed between richness and diversity values and species collection values with drift-fence micro-habitat variables. To assess the importance of stand-level landscape variables on salamander relative abundance, Spearman's rank correlations were performed between richness, diversity, and species collection values averaged across each stand with topographic metrics.

3. Results

During the 60,060 pitfall trapnight effort, we collected 3937 salamanders. This included 184 *Desmognathus aeneus* (Brown and Bishop), 59 *Desmognathus monticola* (Dunn), 392 *Desmognathus ocoee* (Nicholls), 21 *Desmognathus quadramaculatus* (Holbrook), 626 *Eurycea bislineata* (Green), 45 *Gyrinophilus porphyriticus* (Green), 17 *Pseudotriton ruber* (Latreille), 2,556 *Plethodon glutinosus* (Green), 30 *Plethodon serratus* Grobman, and 7 *Notophthalmus viridescens* (Raf.). Stand age effects were significant and distance from drift-fence arrays to water was a significant covariate for species richness and diversity measures (Table 1). Species richness and diversity measures were greater in ≥85-year-old stands than in young-aged stands or 50-year-old stands (Table 2). *Desmognathus quadramaculatus*, *Pseudotriton ruber*, *Plethodon serratus*, and *Notophthalmus viridescens* collections were excluded from individual ANCOVA tests due to low overall collection numbers. Stand age effects were significant for all other species except

Table 1
Analysis of covariance table for salamander species richness, Shannon's diversity, and species relative abundance among cove hardwood stands 15, 25, 50, and ≥85 years of age for 60,060 pitfall trapnights on the Chattahoochee National Forest, Georgia, 1994–1995[a]

Source	d.f.	SS	MS	*F*	*P*
Species richness					
AGE	3	15.42	5.14	5.34	0.0025
DISTANCE	1	25.28	25.28	26.25	0.0001
Error	60	57.78	0.96		
Shannon's diversity					
AGE	3	1.92	0.65	11.75	0.0001
DISTANCE	1	1.46	1.46	26.14	0.0001
Error	60	3.36	0.06		
Desmognathus aeneus					
AGE	3	16.10	5.37	8.16	0.0001
DISTANCE	1	2.98	2.98	4.55	0.0371
Error	60	39.46	0.66		
Desmognathus monticola					
AGE	3	0.37	0.122	0.17	0.9155
DISTANCE	1	4.99	4.99	6.99	0.0104
Error	60	42.85	0.714		
Desmognathus ocoee					
AGE	3	60.32	20.11	7.81	0.0002
DISTANCE	1	38.93	38.93	18.05	0.0001
Error	60	129.46	2.15		
Eurycea bislineata					
AGE	3	50.47	16.82	14.82	0.0001
DISTANCE	1	7.02	7.02	6.19	0.0157
Error	60	68.09	1.13		
Gyrinophilus porphyriticus					
AGE	3	2.27	0.76	1.90	0.1398
DISTANCE	1	4.15	4.15	10.36	0.0021
Error	60	24.03	0.40		
Plethodon glutinosus					
AGE	3	51.63	17.21	5.57	0.0019
DISTANCE	1	0.12	0.12	0.04	0.8448
Error	60	185.31	3.08		

[a] Stand age = AGE and distance from drift-fence array to water = DISTANCE.

Table 2
Mean salamander species richness, Shannon's diversity, and species relative abundance, and contrasts adjusted for distance from drift-fence array to water covariate among cove hardwood stands 15, 25, 50, and ≥85 years of age for 60,060 pitfall trapnights on the Chattahoochee National Forest, Georgia, 1994–1995[a]

	Stand age (years)			
	15	25	50	≥85
Species richness[b]				
Mean	3.77	4.06	4.14	5.01
S.E.	0.25	0.27	0.26	0.22
Shannon's diversity[c]				
Mean	0.55	0.54	0.45	0.88
S.E.	0.06	0.06	0.06	0.05
Desmognathus aeneus[d]				
Mean	1.20	1.28	0.76	2.09
S.E.	0.21	0.22	0.22	0.18
Desmognathus monticola[e]				
Mean	0.38	0.30	0.33	0.49
S.E.	0.22	0.23	0.23	0.18
Desmognathus ocoee[f]				
Mean	1.11	1.39	0.07	2.71
S.E.	0.38	0.40	0.39	0.33
Desmognathus quadramaculatus[g]				
Mean	0.02	0.12	0.10	0.49
S.E.	0.12	0.13	0.13	0.11
Eurycea bislineata[h]				
Mean	2.36	4.53	2.04	2.37
S.E.	0.28	0.29	0.29	0.24
Gyrinophilus porphyriticus[i]				
Mean	0.58	0.19	0.30	0.64
S.E.	0.16	0.17	0.17	0.14
Pseudotriton ruber[g]				
Mean	0.12	0.56	0.12	0.05
S.E.	0.12	0.12	0.12	0.10
Plethodon glutinosus[j]				
Mean	5.18	7.37	6.39	5.18
S.E.	0.45	0.48	0.47	0.39
Plethodon serratus[g]				
Mean	0	0	0.92	0
S.E.	0	0	0.14	0
Notophthalmus viridescens[g]				
Mean	0.15	0	0.13	0.15
S.E.	0.07	0	0.08	0.07

[a] Means are reported by total trapnights per individual array (924 pitfall trapnights) across individual stand ages ($n = 15$ arrays each in stands aged 15, 25, and 50 years and $n = 20$). For contrasts, YG represents stands aged 15 and 25 years.

[b] Contrasts: YG < 85 ($P = 0.0003$), YG = 50 ($P = 0.4892$), 50 < 85 ($P = 0.0149$).

[c] Contrasts: YG < 85 ($P = 0.0001$), YG = 50 ($P = 0.2387$), 50 < 85 ($P = 0.0001$).

[d] Contrasts: YG < 85 ($P = 0.007$), YG = 50 ($P = 0.0798$), 50 < 85 ($P = 0.0001$).

[e] Contrasts: YG = 85 ($P = 0.5513$), YG = 50 ($P = 0.9622$), 50 = 85 ($P = 0.5894$).

[f] Contrasts: YG < 85 ($P = 0.0011$), YG > 50 ($P = 0.0196$), 50 < 85 ($P = 0.0001$).

[g] Excluded from ANCOVA (see text).

[h] Contrasts: YG > 85 ($P = 0.001$), YG > 50 ($P = 0.0002$), 50 = 85 ($P = 0.3797$).

[i] Contrasts: YG = 85 ($P = 0.1769$), YG = 50 ($P = 0.6902$), 50 = 85 ($P = 0.1344$).

[j] Contrasts: YG > 85 ($P = 0.038$), YG < 50 ($P = 0.0001$), 50 = 85 ($P = 0.0551$).

Desmognathus monticola and *Gyrinophilus porphyriticus* (Table 1). Distance from drift-fence arrays to water was a significant covariate for all species tested except *Plethodon glutinosus* (Table 1). Relative abundances of *Desmognathus aeneus* and *Desmognathus ocoee* were greater in ≥85-year-old stands than in either young-aged or 50-year-old stands (Table 2). Relative abundances of *Eurycea bislineata* and *Plethodon glutinosus* were greater in young-aged stands than in ≥85-year-old stands (Table 2).

Species richness, species diversity, and the relative abundances of salamanders were not significantly correlated with most micro-habitat variables measured, notably LITTER, SOIL, CWD, LGCWD, and DECAY. Abundances of *Plethodon glutinosus* and *Plethodon serratus* were negatively correlated with CANOPY (Table 3). Species richness, species diversity and abundance of *Desmognathus ocoee* were positively correlated with ROCK (Table 3). Species richness and diversity measures and collections of *Desmognathus monticola* and *Gyrinophilus porphyriticus* were negatively correlated with DISTANCE (Table 3).

Among stand-level habitat variables, ASPECT and PSS were not correlated with species richness and diversity or with the relative abundances of any individual species. *Plethodon glutinosus* was positively correlated with ELEV (Table 4). *Desmognathus quadramaculatus* was positively correlated with LFI and *Eurycea bislineata* was negatively correlated with LFI (Table 4). Species richness, species diversity, and abundances of *Desmognathus ocoee, Desmognathus quadramaculatus*, and *Gyrinophilus porphyriticus* were positively correlated with COVEHA (Table 4). *Eurycea bislineata* was negatively correlated with COVEHA (Table 4). Species richness, species diversity and abundances of *Desmognathus monticola, Desmognathus ocoee, Desmognathus quadramaculatus,* and *Gyrinophilus porphyriticus* were positively correlated with COVEKM (Table 4). Species richness, species diversity, and collections of *Desmognathus ocoee* and *Desmognathus quadramaculatus* were positively correlated with BASAL (Table 4). *Eurycea bislineata* was negatively correlated with BASAL (Table 4).

Table 3
Spearman's correlation coefficients between micro-habitat variables and species richness, Shannon's diversity, and total abundances of salamanders per pitfall drift-fence array ($n = 65$), in cove hardwood stands 15, 25, 50, and ≥85 years of age on the Chattahoochee National Forest, Georgia, 1994–1995 (see text for variable descriptions)

	r_s	P
CANOPY		
Plethodon glutinosus	−0.3149	0.0106
Plethodon serratus	−0.3395	0.0057
DISTANCE		
Species richness	−0.5228	0.0001
Shannon's diversity	−0.5016	0.0001
Desmognathus monticola	−0.5267	0.0001
Gyrinophilus porphyriticus	−0.4931	0.0001
ROCK		
Species richness	0.4628	0.0001
Shannon's diversity	0.4643	0.0001
Desmognathus ocoee	0.4091	0.0001

Table 4
Spearman's correlation coefficients between stand-level variables and species richness, Shannon's diversity, and total abundances of salamanders in cove hardwood stands ($n = 13$), 15, 25, 50, and ≥85 years of age on the Chattahoochee National Forest, Georgia, 1994–1995 (see text for variable descriptions)

	r_s	P
BASAL		
Species richness	0.6384	0.0189
Shannon's diversity	0.5714	0.0413
Desmognathus ocoee	0.6443	0.0175
Desmognathus quadramaculatus	0.6241	0.0225
Eurycea bislineata	−0.5659	0.0438
COVEHA		
Species richness	0.5703	0.0419
Shannon's diversity	0.6056	0.0283
Desmognathus ocoee	0.6344	0.0199
Desmognathus quadramaculatus	0.5892	0.0341
Eurycea bislineata	−0.5556	0.0487
Gyrinophilus porphyriticus	0.5892	0.0341
COVEKM		
Species richness	0.6143	0.0255
Shannon's diversity	0.6217	0.0233
Desmognathus monticola	0.6008	0.0299
Desmognathus ocoee	0.5773	0.0388
Desmognathus quadramaculatus	0.7661	0.0023
Gyrinophilus porphyriticus	0.5849	0.0357
ELEV		
Plethodon glutinosus	0.7307	0.0045
LFI		
Desmognathus quadramaculatus	0.6545	0.0152
Eurycea bislineata	−0.5580	0.0475

4. Discussion

Our findings that salamander species richness and diversity measures and the relative abundance of salamander species, notably *Desmognathus* spp., were lower in young-aged cove hardwood stands relative to those ≥85 years-old is consistent with other research in the southern Appalachians (Ash, 1988; Petranka et al., 1993, 1994; Harpole and Haas, 1999). Similar trends have been reported in other environments (Blymer and McGinnes, 1977; Enge and Marion, 1986; Pough et al., 1987; Dupuis et al., 1995). Although stand age impacts to *Desmognathus* spp. could be a result of stream degradation and siltation rather than changed overstory conditions (Petranka et al., 1994), we assume that stream siltation in the 15 and 25-year-old stands we studied following timber harvest under US Forest Service stewardship was minimal. With some precautions, sediment load increases in aquatic systems from logging are generally short-lived in the Appalachians (Kochenderfer et al., 1997). Furthermore, species such as *Desmognathus aeneus* and *Desmognathus ocoee* often occur far from water. We believe that our year-round sampling effort and closely matched study areas provide strong evidence that clearcutting in southern Appalachians cove hardwood forests similar to those we surveyed on the BRD does negatively impact several salamander species. Assuming our older stands represent intact salamander communities in terms of richness and abundance, it appears from our analysis that recovery following clearcutting exceeds 50 years in these cove hardwood stands. We agree with the assertions of Petranka et al. (1994) that lower salamander numbers found in the initial years following timber harvest might not represent a threat in terms of overall species viability. However, the duration of these lowered numbers or altered community assemblages could represent a loss of biodiversity and or diminished ecosystem function. Accordingly, this would be in conflict with the charge of biodiversity maintenance and preservation of ecological processes that has been given to public land managers in the Southeast (Sharitz et al., 1992).

The relative abundances of some species were either unaffected by stand age or showed a negative relationship between increasing stand age and relative abundance. *Desmognathus monticola*, *Gyrinophilus porphyriticus*, and *Pseudotriton ruber*, collected in small numbers, were closely tied to seeps and streams found on our study sites, rather than in the surrounding forest, so stand age may have had little impact. All seven *Notophthalmus viridescens* collected were the terrestrial juvenile red eft stage. *Plethodon glutinosus* and *Eurycea bislineata* were more common in young-aged stands than in older cove hardwood stands in our study. Petranka et al. (1993) detected no difference in numbers of *Plethodon glutinosus* in high elevation forests in the southern Appalachians of western North Carolina between mature forest stands and stands ≥10 years of age. Although more xeric compared to the cove hardwood stands we studied, Ash (1988) noted that *Plethodon glutinosus* disappeared in recently clearcut oak-hickory (*Carya* sp.) stands in western North Carolina. Although *Eurycea bislineata* is aquatic in its breeding ecology, the species is distributed throughout mesic forests in the southern Appalachians (Wilson, 1995; Petranka, 1998). *Plethodon serratus* occurred in only two of the 50-year-old stands we surveyed. We selected study sites within a confined geographic area to reduce the possibility of sampling areas with dissimilar salamander communities, however, these two sites with *Plethodon serratus* occur at the species' easternmost distributional limit in north Georgia. All *Plethodon serratus* were collected in the fall and would not have been detected had we limited sampling to the spring or summer months.

Correlations of richness and diversity measures and relative salamander abundances with habitat variable correlates such as CANOPY or BASAL are not unusual. These variables are related to stand age that our data indicate is highly related to salamander abundance. The correlation between LFI and the relative abundance of *Desmognathus quadramaculatus* probably reflects the association with high LFI values and the presence of larger streams in the most concave cove sites. Species richness and diversity measures and abundance of *Desmognathus ocoee* correlates with ROCK may show the significance that emergent rock has for microsite moisture retention, refugia and feeding substrate important to woodland salamanders (DeMaynadier and Hunter, 1995). Indeed, emergent rock may have served as one of the primary long-term refugia and colonization sources for Plethodontids throughout the central and southern Appalachians following the widespread and destructive timber harvests in the early 1900's (T. Pauley, personal communication). Conversely, other variables such as LITTER, SOIL, CWD, LGCWD, and DECAY that varied across stands (see Ford et al., 1997, 2000), may have been well within upper or lower thresholds that would show significant correlations with salamander abundance. Although leaf litter does decline following clearcutting (Ash, 1995) and may take many years to return to pre-harvest depths (Likens et al., 1978), tree growth and stand development on cove hardwood sites in the southern Appalachians are rapid (Beck and Hooper, 1986) such that litter inputs and leaf litter cover may be relatively high within a few years following regeneration. Stands ≥85 years old contained the largest amounts of CWD and LGCWD, however, all of the young-aged stands we surveyed still contained considerable amounts of residual debris from the previous stand, much of which was at an advanced level of DECAY.

Our data clearly demonstrate that cove hardwood habitat area and the amount of nearby cove hardwood habitat as measured in the variables COVEHA and COVEKM are important factors influencing salamander abundance and species richness and diversity. Salamander abundance might decline due to changes in micro-habitat following clearcutting or from the effects of changed intra-specific or inter-specific competition or increased predation (Kramer et al., 1993; Petranka et al., 1993; Ash and Bruce, 1994). Regardless, the surrounding forest and connecting riparian areas might serve as the most important "source" for salamander recolonization (Dupuis et al., 1995). The salamander communities in our cove hardwood study sites that were imbedded in larger matrices of connected cove habitat or nearby cove habitat had the ability to recover more quickly from clearcutting than those that are more isolated from similar habitat (Fig. 2). In addition to stand-level management considerations for salamanders, such as leaving uncut patches within harvested stands, riparian area protection, and coarse woody debris retention throughout (Dupuis et al., 1995), landscape connectivity should be considered because it potentially aids recolonization of altered or fragmented habitats (DeMaynadier and Hunter, 1995) and helps maintain genetic integrity (Gibbs, 1998). Maintaining landscape connectivity and minimizing habitat fragmentation is a well-accepted tenet of conservation biology (Burkey, 1995) that has direct application in the conservation of the salamander communities we studied. Although current trends show a drastic decline in timber harvest on public lands in the southern Appalachians (Ford et al., 1997), forest managers may need to address salamander conservation at landscape scales, rather than solely at the stand-level. Additional experimental research linking landscape connectivity, forest management, and salamander conservation in the southern Appalachians in cove hardwood and other forest types seems warranted.

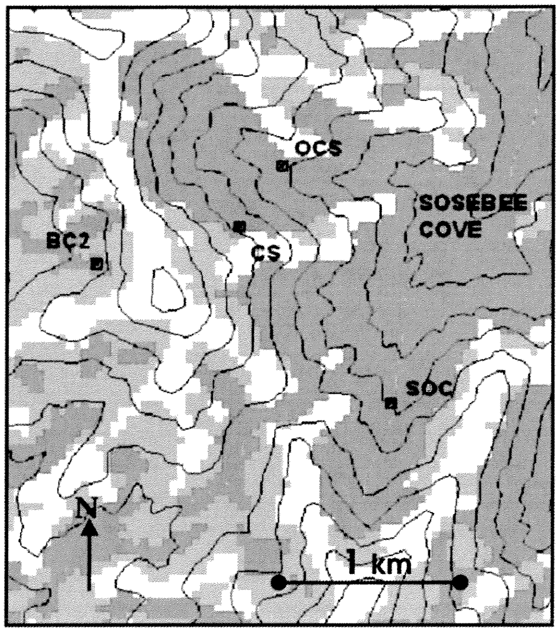

Fig. 2. Illustration of cove hardwood patch sizes surrounding pitfall drift-fence arrays in the Wolf Creek Watershed (CS, OCS, SOC) and the Miller Cove Watershed (BC2) on the Chattahoochee National Forest, Georgia, 1994–1995. Dark gray areas signify mesic, cove hardwood habitat, light gray areas signify drier sideslopes, and white areas signify xeric ridges as determined by GIS analysis. Salamander species richness and diversity is positively correlated to area of connected cove hardwood habitat (see text for complete discussion).

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Cavity trees and coarse woody debris in old-growth and managed northern hardwood forests in Wisconsin and Michigan

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Cavity trees and coarse woody debris in old-growth and managed northern hardwood forests in Wisconsin and Michigan

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Abstract: The effects of uneven-aged management on the availability of coarse woody debris habitat were examined in northern hardwood forests (with and without a hemlock component) in north-central Wisconsin and adjacent western Upper Michigan. Snags, cavity trees, fallen wood, and recent tip-up mounds in 15 managed uneven-aged (selection) stands were compared with levels in 10 old-growth stands and six unmanaged even-aged second-growth stands. Amounts of coarse woody debris in selection stands were generally intermediate between old-growth and even-aged stands. Density of snags >30 cm DBH in northern hardwood selection stands averaged 12/ha, approximately double that found in even-aged northern hardwoods, but only 54% of the level in old-growth northern hardwoods. Highest densities of snags >30 cm DBH occurred in old-growth hemlock–hardwood stands, averaging over 40 snags/ha. For combined forest types, the volume of fallen wood (>10 cm in diameter) was significantly lower in selection stands (60 m^3/ha) and even-aged stands (25 m^3/ha) than in old-growth stands (99 m^3/ha). Volume differences were even more pronounced for large-diameter debris (>40 cm). Cavity tree density in selection stands averaged 11 trees/ha, 65% of the mean number in old-growth stands. Densities of snags (>30 cm DBH) and large-diameter cavity trees (>45 cm) present in selection stands exceeded current guidelines for wildlife tree retention on public forests.

Résumé : Les effets d'un aménagement inéquienne sur la disponibilité des habitats que procurent les gros débris ligneux ont été étudiés dans les forêts de feuillus nordiques, accompagnés ou non de pruche, du Centre-Nord du Wisconsin et de l'Ouest de la péninsule Nord du Michigan qui est adjacente. Les quantités de chicots, d'arbres avec des cavités, de débris ligneux au sol et de tas récents de houppiers présents dans 15 peuplements aménagés de façon inéquienne ont été comparées à celles qui étaient présentes dans 10 vieux peuplements et six peuplements équiennes de seconde venue et non aménagés. La quantité de débris ligneux grossiers dans les peuplements inéquiennes se situait généralement à mi-chemin entre les quantités retrouvées dans les vieux peuplements et dans les peuplements équiennes. La densité des chicots de plus de 30 cm au DHP atteignait en moyenne 12/ha dans les peuplements inéquiennes de feuillus nordiques, soit approximativement le double de ce qu'on retrouve dans les peuplements équiennes de feuillus nordiques, mais seulement 54% de la quantité présente dans les vieilles forêts de feuillus nordiques. Les plus fortes densités de chicots de plus de 30 cm au DHP ont été observées dans les peuplements de feuillus et de pruche, atteignant en moyenne plus de 40 chicots/ha. Pour tous les types de forêts combinés, le volume de débris ligneux au sol (>10 cm de diamètre) était significativement plus faible dans les peuplements inéquiennes (60 m^3/ha) et dans les peuplements équiennes (25 m^3/ha) que dans les vieux peuplements (99 m^3/ha). Les différences de volume étaient encore plus prononcées pour les débris de fort diamètre (>40 cm). La densité des arbres avec des cavités atteignait en moyenne 11 arbres/ha dans les peuplements inéquiennes, soit 65% du nombre moyen dans les vieux peuplements. La densité des chicots (>30 cm au DHP) et des arbres de fort diamètre (>45 cm) avec des cavités présents dans les peuplements inéquiennes excédait les normes actuelles concernant le maintien d'arbres pour la faune sur les terres publiques.

[Traduit par la Rédaction]

Introduction

Coarse woody debris, including snags and fallen wood, provides important habitat elements for a wide array of biota, in addition to its role in nutrient cycling, carbon storage, and other ecosystem functions. Organisms that use coarse woody debris for food and cover range from arthropods, herptiles, birds, and small mammals to a host of fungi and microorganisms (Jaeger 1980; Harmon et al. 1986). In the northern hardwood forest, over 40 species of birds and mammals use cavities in snags and dead portions of live trees for nest sites, dens, escape cover, and winter shelter (Evans and Conner 1979; DeGraaf and Shigo 1985). Fallen trees often create tip-up mounds, which can enhance establishment of some plants, provide nesting sites for certain bird species, and offer moist refuge for amphibians in pits during dry periods (Heatwole 1962; Beatty 1984; Peterson et al. 1990). In unmanaged old-growth forests, the death of large trees and subsequent gap formation result in structurally complex habitat (Franklin et al. 1981; Hunter 1990). In managed forests, efficient harvest and removal of tree boles from the stand can limit development of such features and possibly reduce populations of organisms that depend upon these structures (Haapanen 1965; Cline et al. 1980).

Wildlife population studies suggest that large size and advanced wood decay are two key attributes in vertebrate

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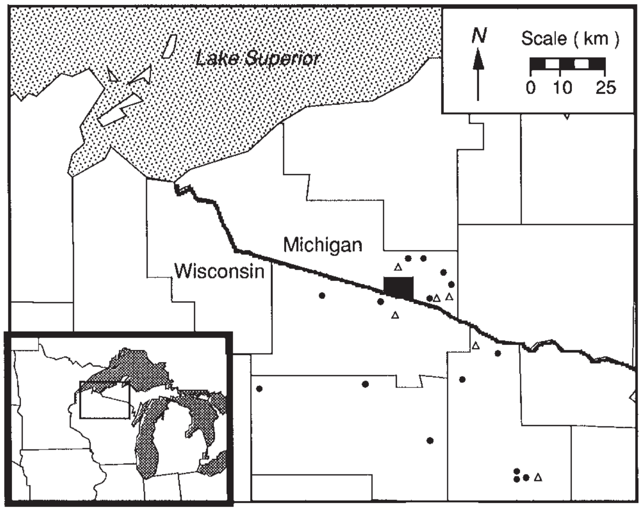
Fig. 1. Map of the study area. All old-growth stands were located within the Sylvania Wilderness Area, Ottawa National Forest (represented by the solid polygon). Locations of individual even-aged and selection stands are indicated with open triangles and solid circles, respectively.



preferences for coarse woody debris (Raphael and White 1984; Swallow et al. 1986). While small woody debris has habitat value for some organisms, large snags and fallen boles persist for a longer time before fragmenting, provide greater forage area, and meet minimum size requirements of a wider range of potential vertebrate users than do smaller elements (Conner et al. 1975; Cline et al. 1980; DeGraaf and Shigo 1985). For certain taxa such as plethodontid salamanders, the availability of cool, moist microclimate under loose bark and within the interior of fallen boles with advanced decay is even more critical to habitat suitability than bole diameter (Aubry et al. 1988; Petranka et al. 1994).

Studies in the eastern United States comparing habitat structure of old-growth forests and younger stands have been limited by the scarcity of old-growth stands. Available studies of coarse woody debris in eastern deciduous forests have focused primarily on second-growth stands originating after heavy logging in the early twentieth century, but that have received little or no silvicultural treatment since the time of stand initiation (e.g., Tritton 1980; Carey 1983; McCarthy and Bailey 1994). Less is known regarding the effects of various management strategies such as intermediate thinning, timber stand improvement, or uneven-aged management on the distribution of cavity trees and coarse woody debris.

Uneven-aged management is the predominant silvicultural system for managing mature northern hardwoods in the upper Midwest on both public and private lands (Jacobs 1987). On public lands, harvests removing 20–30% of the trees in each size class are made at 12- to 15-year intervals. Normally the maximum tree size retained in the stands is 60 cm diameter at breast height (DBH) (Arbogast 1957; Tubbs 1977). It has been suggested that by discriminating against large-diameter, low-vigor, and defective trees, uneven-aged management could restrict the development of cavity trees and coarse woody debris (Zeedyk and Evans 1975; McComb and Noble 1980). Partially in response to these concerns, management guidelines were introduced on some Wisconsin and Michigan national forests in the early 1980s calling for the retention of all active cavity trees and approximately 5–10 snags (>30 cm DBH)/ha (U.S. Department of Agriculture 1980). However, information regarding the actual levels of snags, cavity trees, and fallen wood present in managed uneven-aged northern hardwood forests appears to be limited to two stands in Connecticut and New Hampshire examined by McComb and Noble (1980) and by Gore and Patterson (1986).

The present study is part of a larger interdisciplinary research project investigating differences in forest structure, ecosystem processes, and biological diversity between old-growth and managed hemlock–hardwood forests. Our objective was to examine the effects of uneven-aged management on forest habitat structure in northern hardwood stands (with and without a hemlock component). Snags, cavity trees, fallen wood, and tree-fall mounds in managed uneven-aged stands (hereafter termed selection stands) were compared with base-line levels in unmanaged uneven-aged old-growth stands and in unmanaged even-aged second-growth stands. Unmanaged even-aged second-growth stands (hereafter referred to as even-aged) were included in the study because of their current prevalence on Wisconsin's public lands and because questions regarding their future management remain unresolved (Alverson et al. 1994).

Methods

Study areas

This study was conducted in mesic northern hardwood and hemlock–hardwood stands in north-central Wisconsin and adjacent western upper Michigan (Fig. 1). Most research sites are located on the Winegar terminal moraine complex, within sub-subsection IX.3.2 in Albert's (1995) regional landscape classification. The region is characterized by a thick layer of sandy glacial drift underlain by iron-rich Precambrian bedrock. Rolling irregular topography formed by disintegrating glacial ice includes many kettle lakes and steep sandy ridges (Albert 1995). Elevations range from approximately 500 to 550 m. The upland soils are predominantly sandy loams and loamy sands classified as either well-drained coarse–loamy Typic Haplorthods or moderately well-drained Alfic Fragiorthods. In the Fragiorthods a moderately developed fragipan is found at a depth of 50–100 cm (Hole 1976; Spies and Barnes 1985). The remainder of research sites (those outside the Winegar moraine) were located on similar upland loam and sandy loam spodosols that formed in ground moraines and areas of pitted outwash (i.e., within sub-subsections IX.3.1, IX.3.3, and IX.5; Albert 1995). The climate of the entire region is continental with heavy snows, extremely cold winters, and a frost-free period of less than 100 days (Albert et al. 1986). Mean monthly temperatures range from –12.2°C in January to 18.6°C in July. Annual precipitation averages 850 mm, with 60% of that amount falling between May and September (Lac Vieux Desert weather station, National Climatic Data Center, Asheville, N.C., 1993).

All old-growth stands were located in the Sylvania Wilderness Area on the Ottawa National Forest, Mich. This tract has had only localized past cutting and includes over 6000 ha of old-growth forest (U.S. Department of Agriculture 1964). Trees range in age up to a maximum of about 350 years (Dahir 1994). Ten old-growth stands larger than 20 ha in size were selected, including six stands dominated by sugar maple (*Acer saccharum* Marsh.) and four dominated by a mixture of eastern hemlock (*Tsuga canadensis* (L.) Carrière), sugar maple, and yellow birch (*Betula alleghaniensis* Britt.). Hemlock–hardwood stands were defined as those in which hemlock composed >30% of the basal area for trees ≥2 cm DBH.

Ecological classification systems developed for the Sylvania

Wilderness Area (Spies and Barnes 1985), the Ottawa National Forest, and for the forest habitat types in Michigan (Coffman et al. 1983) and Wisconsin (Kotar et al. 1988) were used to select old-growth, selection, and even-aged stands with similar soil and site characteristics. All northern hardwood and hemlock–hardwood study stands had been classified as forest habitat type *Acer–Tsuga–Dryopteris* (ATD) (Coffman et al. 1983; Kotar et al. 1988), although a few plots were transitional between the ATD habitat type and the *Acer–Viola–Osmorhiza* (AViO) or the *Acer–Tsuga–Maianthemum* (ATM) type.

The primary criteria for managed uneven-aged (selection) study sites were previous management by the selection system on a cutting cycle of 8–15 years, a minimum residual basal area of 16.1 m^2/ha (70 ft^2/acre), and a maximum residual tree diameter >45 cm DBH. These selection stands typically have a range of tree ages, with some individuals more than 200 years old (Cole and Lorimer 1994). Although stands were managed under wildlife tree retention guidelines, they were not managed using a prescription to restore old-growth characteristics, a more recent treatment being introduced on some public forests (e.g., Rominske and Busch 1991).

We also located study areas in six even-aged second-growth northern hardwood stands with a predominant age of 65–75 years. These stands had not been previously thinned and were essentially unmanaged since stand initiation. They contain only scattered larger trees from the former stand and no significant biological legacy of snags and fallen wood. Even-aged hemlock–hardwood stands were not included in the comparison because they are uncommon in the region. The final 31 stands selected included 22 northern hardwood stands (six old-growth, 10 selection, and six even-aged) and nine hemlock–hardwood stands (four old-growth, five selection).

Field procedures

A single large 30 × 100 m rectangular plot was established in each stand. This design accommodated our measurement of forest structural features as well as the sampling needs of various research team members using the same sites for related studies (e.g., soils, fungi, invertebrates, birds, small mammals). Plot center was randomly located within the stand after allowing a 200-m buffer from stand boundaries. The long axis of the plot was oriented east–west and divided into three contiguous 10 × 100 m transects.

The species, DBH, and total height were recorded for all dead trees ≥10 cm DBH and >1.5 m tall within the 30 × 100 m plot. Heights were measured with a clinometer or telescoping height pole. Snags were examined for presence of loose bark plates larger than 25 × 25 cm (i.e., bark cavities), which might serve as bat roosts or brown creeper (*Certhia americana*) nests (Evans and Conner 1979; Brady 1983). Additional information was collected on decay class (see classification for fallen wood below), fragmentation status (1, crown intact; 2, only large branches remaining; 3, bole only; 4, broken bole), and the percentage of bark remaining on snag. Snag size diversity (Raphael and White 1984) was calculated with Shannon's diversity index (H') using four diameter classes (10–30, 30–45, 45–60, >60 cm) and four height classes (<6, 6–12, 12–18, >18 m).

Within a 20 × 100 m area (the center transect plus one randomly selected outer transect), all trees >10 cm DBH were searched for dens and nesting cavities using 8× binoculars (Healy et al. 1989). Following the criteria of Carey (1983), a nesting cavity was considered a hole in any live or dead tree more than 1 m above the ground that provides overhead shelter from precipitation and has no cracks or openings except the entrance. For all cavity trees, data were collected on tree species, live–dead status, and DBH, along with cavity height, location (bole, branch, dead top), opening size (2–5, 5–10, 10–20, >20 cm), and cavity origin (bird excavated or natural wound). Cavities were checked during the winter for better visibility, but no attempt was made to monitor nesting species or verify use during the survey year. Holes were examined for tooth marks, nesting material, rubbing marks, and debris near the entrance to judge the certainty of use. Cavity use certainty was tallied as (*i*) definite, (*ii*) fairly certain, or (*iii*) uncertain. To avoid counting cavities of questionable value to wildlife, only cavities classified as "definite" or "fairly certain" were included in calculations of cavity tree density. Hollow trees and cavities with a second or overhead opening were not considered as nesting cavities, but were recorded as dens or escape structures.

Fallen wood (fallen boles, branches, natural and cut stumps, or harvest tops) was measured within the center 10 × 100 m transect, which was divided into ten 10 × 10 m quadrats. The smallest size class (10.0–19.9 cm in diameter) was subsampled only in the northwest 5 × 5 m quarter of each 10 × 10 m quadrat. For each piece, species, decay class, and origin were recorded. Decay status was recorded using a system of five classes based upon bark slippage and degree of decay into the sapwood and heartwood, modified from Sollins (1982) and Lambert et al. (1980), and similar to that used by Muller and Liu (1991) for deciduous forests in Kentucky. Extent of decay was inspected with a 0.5-cm-diameter pointed metal rod. Decay classes were defined as class I (tight bark and no visual decay), class II (some bark slippage with incipient decay in the sapwood), class III (decay obvious in the outer layers, pointed metal rod penetrates more than half the radius), class IV (some of the outer xylem layers missing, decay extending well towards the core, metal rod penetrates clear through the bole), or class V (organic debris collapsed to ground level and mixing with soil, little structural integrity).

Volume of each piece of fallen wood was calculated from piece length and the cross-sectional area of each end with Smalian's formula for cubic volume (Wenger 1984). Using metal tree calipers, stem diameter was measured at each end, or where the piece either extended beyond the transect boundary or dropped below the minimum measurable diameter. Diameter measurements at additional points along the axis were taken if total length exceeded 5 m. The volumes of any hollow portions of logs or stumps were similarly calculated and subtracted from the total. "Large" fallen wood included all pieces within the sample area that had a midpoint diameter >40 cm, calculated as the quadratic mean of the end diameters. The volume of coarse woody debris in snags was calculated using species-specific regression equations from Hahn (1984), which allow computation of cubic volume from our measures of snag DBH and height.

Sample wood disks (n = 167) were collected for estimates of coarse woody debris mass. Disks were cut into rectangular cubes with a band saw or knife and the dimensions of the subsample measured with vernier calipers. Subsamples were oven-dried to a constant mass at 70°C and then weighed. Wood density for the various species and decay combinations was determined from mass and volume of subsamples, in grams of dry mass per cubic centimetre of "green" volume. Density for decay class I debris was obtained from published tables (U.S. Department of Agriculture Forest Products Laboratory 1976). Stand-level estimates for the biomass of coarse woody debris (>10 cm in diameter) were calculated from these density estimates and from the debris volume in each species and decay class combination sampled in each plot.

Because of the great longevity of tree-fall microtopography and the difficulty in distinguishing mounds from mechanical disturbances in managed landscapes, only recent tip-ups were considered in which woody debris of at least decay class V was still evident from the fallen tree. For all such tip-up mounds and pits within the 10 × 100 m center strip, perpendicular length and width of both pit and mound were measured to the nearest decimetre with a fiberglass tape and treated as ellipse diameters in area calculations.

Statistical analyses

The single large plot in each stand was considered to be the experimental unit. The five stand types included two different forest types (northern hardwood and hemlock–hardwood) and three management histories (even-aged, selection, and old-growth). Of primary interest, a priori, were four comparisons among these five forest type – management history combinations. Two of these involved comparing

Table 1. Summary of two-way and one-way ANOVA tests for the effects of management history and forest type on coarse woody debris characteristics.

	Two-way ANOVA for old-growth and managed uneven-aged stands				One-way ANOVA for northern hardwood stands			
Source of variation	df	Mean squares	*F* value	*P* >*F*	df	Mean squares	*F* value	*P* >*F*
Cavity tree density								
Management history	1	3.568	3.44	0.079	2	2.062	1.23	0.318
Forest type	1	2.387	2.30	0.146				
Management history × forest type	1	0.118	0.11	0.740				
Error	19	1.038			17	1.679		
Density of snags >10 cm DBH								
Management history	1	19.826	10.77	0.004*	2	23.633	8.47	0.002*
Forest type	1	2.902	1.58	0.223				
Management history × forest type	1	14.920	8.10	0.010*				
Error	21	1.841			19	2.791		
Density of snags >30 cm DBH								
Management history	1	35.697	37.72	0.001*	2	11.247	7.60	0.004*
Forest type	1	2.261	2.39	0.137				
Management history × forest type	1	7.275	7.69	0.011*				
Error	21	0.947			19	1.479		
Density of snags >45 cm DBH								
Management history	1	49.118	29.98	0.001*	2	17.746	11.34	0.001*
Forest type	1	1.295	0.79	0.384				
Management history × forest type	1	1.571	0.96	0.339				
Error	21	1.638			19	1.564		
Density of snags with loose bark plates								
Management history	1	2.045	0.87	0.362	2	0.472	0.19	0.827
Forest type	1	1.943	0.82	0.374				
Management history × forest type	1	5.553	2.36	0.140				
Error	21	2.358			19	2.501		
Snag coarse woody debris volume								
Management history	1	64.587	32.78	0.001*	2	4.483	2.16	0.142
Forest type	1	3.333	1.69	0.208				
Management history × forest type	1	19.469	9.88	0.005*				
Error	21	1.970			19	2.071		
Fallen coarse woody debris volume								
Management history	1	29.240	23.03	0.001*	2	40.731	34.4	0.001*
Forest type	1	0.807	0.64	0.434				
Management history × forest type	1	0.021	0.02	0.900				
Error	21	1.270			19	1.184		
Large fallen coarse woody debris volume (>40 cm in diameter)								
Management history	1	18.627	5.47	0.029*	2	38.474	16.63	0.001*
Forest type	1	0.164	0.05	0.829				
Management history × forest type	1	1.010	0.30	0.592				
Error	21	3.404			19	2.313		
Total coarse woody debris volume (snag plus fallen)								
Management history	1	72.326	36.03	0.001*	2	38.153	19.09	0.001*
Forest type	1	0.127	0.06	0.804				
Management history × forest type	1	4.330	2.16	0.157				
Error	21	2.007			19	1.999		

Table 1 (*concluded*).

	Two-way ANOVA for old-growth and managed uneven-aged stands				One-way ANOVA for northern hardwood stands			
Source of variation	df	Mean squares	*F* value	*P* >*F*	df	Mean squares	*F* value	*P* >*F*
Area covered by recent tip-up (pit plus mound)								
Management history	1	0.052	0.01	0.935	2	21.719	2.98	0.075
Forest type	1	15.428	2.01	0.171				
Management history × forest type	1	4.931	0.64	0.432				
Error	21	7.679			19	7.283		

*Statistically significant difference ($P < 0.05$).

old-growth stands and selection stands within each forest type. The other two comparisons involved even-aged versus selection northern hardwood stands and even-aged versus old-growth northern hardwood stands. Because there were no even-aged hemlock–hardwood stands, the four selection and old-growth combinations were compared in a two-way analysis of variance (ANOVA) testing for the effects of both forest type and management history (fixed effects model). This was followed by a comparison of the three northern hardwood management histories in an one-way ANOVA (Table 1). The four planned pairwise comparisons between means of different stand types were made only if the overall ANOVA indicated significant differences ($P < 0.05$). The individual pairwise comparisons were tested using Fisher's least significant difference method with a common estimate of experimental error (MSE). Statistical analyses were performed using the general linear models procedure in SAS (SAS Institute Inc. 1990). Some variables were transformed prior to ANOVA testing to correct for unequal variance among types and nonnormality. Density and volume values were square root transformed, while values for the percent area in pit and mound microtopography were arcsine square root transformed.

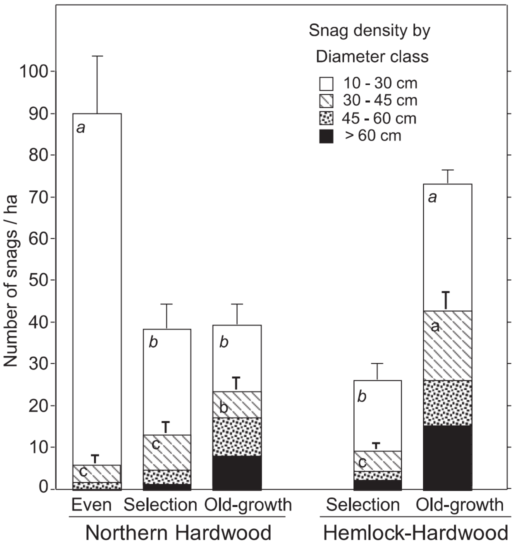
Prior to testing differences in means among the five forest type – management history combinations, two-way ANOVA was used to determine if the means are influenced by two site characteristics found to vary between study stands on the same forest habitat type: (*i*) fragipan (presence or absence) and (*ii*) location on the Winegar moraine (yes or no). Fragipan presence–absence was evenly split for all treatments except selection hemlock–hardwoods, in which no plots had a fragipan. Neither of these factors nor factor × stand type interactions were found to have a significant effect. For all coarse woody debris volume and density variables, the effect of blocking on these two factors resulted in *P* values that exceeded 0.44 in all cases except one (i.e., for fragipan effect on large fallen wood, $P = 0.13$). Consequently, all stands were grouped by the five forest type – management history combinations (i.e., no further blocking on fragipan or location) and analysis conducted as described above.

Results

Snag density and diameter distribution

For northern hardwoods, total snag density (>10 cm DBH) in selection stands was similar to the density measured in old-growth (Fig. 2). Snag density was significantly higher in even-aged stands, but composed primarily of small stems. Nearly three quarters of the snags in even-aged stands were from size classes smaller than the mean live tree diameter. In selection stands, snags were rather evenly distributed above and below the mean live tree diameter (Table 2). Snags in old-growth northern hardwoods had a wide diameter range, but 42% of the dead trees were >45 cm DBH, well above the mean live tree diameter. Combined medium (30–45 cm DBH) and large (>45 cm DBH) snags were significantly more numerous in old-growth than in selection or even-aged northern hardwood stands (Table 1). Selection stands averaged 12 medium and large snags/ha, just over half the level found in old-growth and approximately twice the number in even-aged stands (Fig. 2).

Fig. 2. Density of snags by size class. Top error bars show 1 SE for all snags (>10 cm) and bottom error bars are for snags >30 cm DBH. Treatment means with the same letter are not significantly different at $P = 0.05$.



For hemlock–hardwoods, snag densities in old-growth were significantly higher than in selection stands for all size comparisons shown in Fig. 2. The density of large snags was more than five times greater in old-growth than in selection stands. Total snag density in old-growth hemlock–hardwood was higher than in old-growth northern hardwood, although the proportions of medium and large snags were similar in both. Yellow birch composed 65% of the large snags, nearly half of which were in recent decay classes (I and II). A majority of the large yellow birch trees present in the old-growth hemlock–hardwood stands were snags, possibly casualties of the 1988 drought (cf. Prey et al. 1988).

In old-growth stands of both forest types, dead trees >60 cm DBH constituted 20% of snag density. Snags of this size were absent from all even-aged plots and had low densities in the selection stands (Table 2). In 10 of the 15 selection stands sampled across both forest types, no snags >60 cm DBH

Table 2. Structural characteristics of live trees and snags by forest type and management history.

	Northern hardwood			Hemlock–hardwood	
Characteristic*	Even-aged	Selection	Old-growth	Selection	Old-growth
Total no. of snags sampled	162	115	71	39	88
No. of stands	6	10	6	5	4
Mean DBH (cm)					
Snags[†]	17.6 (12.3, 22.4)	26.8 (16.5, 32.0)	38.9 (18.7, 55.0)	26.0 (12.3, 36.8)	38.8 (21.9, 55.0)
Live trees	20.7	24.2	31.4	24.3	30.8
Density (stems/ha)					
Snags	90	38	39	26	73
Live trees	828	446	313	469	388
% dead trees of total	9.8	7.9	11.2	5.2	15.9
Large snags					
>45 cm DBH	1.7	4.3	16.7	4.0	25.8
>60 cm DBH	0	1.0	7.8	2.0	15.0
Basal area (m^2/ha)					
Snags	2.8	2.8	6.0	2.0	10.9
Live trees	32.3	25.5	34.4	28.0	37.5
% snag basal area of total	7.8	9.9	14.9	6.7	22.5
% of total snag density in each stage					
Snag fragmentation stage					
1. Crown intact	12	6	6	3	8
2. Large branches and bole	37	15	10	21	28
3. Bole only	23	28	27	23	14
4. Broken bole	28	51	58	54	50
% of total snag density in each decay class					
Snag decay class					
I. Sound	17	6	4	5	8
II. Slight decay	55	37	34	36	36
III. Moderate decay	25	52	58	56	53
IV. Advanced decay	2	4	4	3	2
Snags with loose bark plates					
Loose-barked snags/ha[‡]	3.3 (±2.1)	5.3 (±2.4)	3.3 (±1.7)	2.7 (±1.2)	10.0 (±5.6)
% snags with bark plates	3.7	13.9	8.5	10.3	13.6

*Includes trees >10 cm DBH only for both live tree and dead tree characteristics.

[†] First and third quartiles of the snag diameter range presented in parentheses.

[‡] Mean value followed by SE in parentheses.

occurred in the 0.3-ha study plot whereas at least one was present in all of the old-growth plots.

Snag height distribution and size diversity

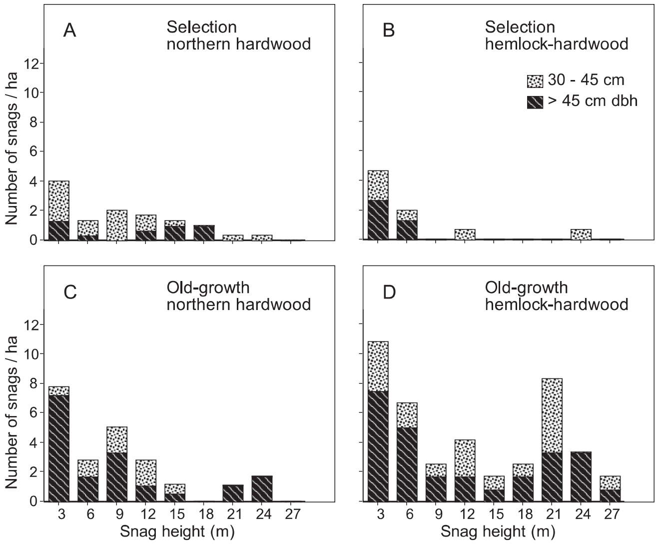
Snag heights are reduced over time by progressive fragmentation (Tyrrell and Crow 1994*b*), altering the suitability of snags for nesting sites and perches. More than half of all snags in both selection and old-growth northern hardwoods had broken boles (fragmentation stage 4), skewing the height distribution toward the shorter height classes. Medium and large snags in these stands were likewise concentrated in shorter height classes (Fig. 3). While absolute densities of medium and large snags were significantly different in selection and old-growth northern hardwoods, these stands had similar proportions of shorter snags (1.5–4.5 m height class), 33 and 35%, respectively, and similar proportions throughout the height profile (Fig. 3). The frequency distributions of snags >30 cm among four 6-m height classes were not significantly different for selection and old-growth stands ($\chi^2 = 0.862$, $P = 0.834$, df = 3). In even-aged northern hardwoods, medium and large snags were more evenly distributed through the height profile, but no snags reached above the 18-m class. Snag size diversity (H'), based on four diameter classes and four height classes, was highest in old-growth ($H' = 1.049$), intermediate in managed uneven-aged ($H' = 0.830$), and lowest in even-aged ($H' = 0.704$).

In hemlock–hardwood selection stands, most snags were relatively short (<7.6 m tall; Fig. 3B). Height distribution in old-growth hemlock–hardwood was distinctly bimodal. Besides a concentration in the two shortest height classes, there was a second grouping in the 19.5- to 22.5-m height class. One third of all large snags, mostly yellow birch in early fragmentation stages, were >18 m tall. Snag size diversity was 30% lower in selection stands ($H' = 0.749$) than in old-growth ($H' = 1.071$). Index values were similar to those for snags in corresponding treatments in northern hardwoods.

Snag fragmentation stage and decay class distributions

Fragmentation and decay class categories were not independent (e.g., almost all fragmentation stage 1 snags were decay class I). The large proportion of broken bole snags (fragmentation stage 4) in both selection and old-growth

Fig. 3. Height distribution of medium and large snags for uneven-aged and old-growth stands. Numbers on the *y*-axis refer to the midpoint of 3-m height classes.



stands resulted primarily from advancing decay in older snags, as opposed to wind or mechanical breakage of recently formed snags. In every stand type, >75% of these broken bole snags were in advanced decay class III or IV.

In northern hardwood selection stands, approximately 43% of all snags were recent (i.e., decay classes I or II), while 56% of snags were in advanced decay class III or IV. Similar decay class distributions were found in old-growth northern hardwoods, as well as in both selection and old-growth hemlock–hardwoods (Table 2). Even-aged stands, in contrast, had a considerably greater percentage of snags (73%) in early decay classes I and II.

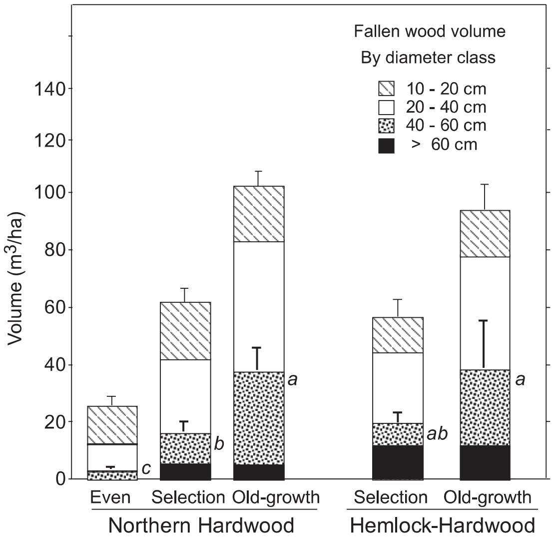
Mean density of snags with loose bark plates ranged from 3 to 10/ha for the five forest type – management history combinations, with highest values in old-growth hemlock–hardwood (Table 2). The proportion of snags with loose bark plates was <15% in all stand types. No significant effects were detected for either management history or forest type (Table 1). Density of loose-barked snags appeared related to tree size and species composition in that they were more common on medium and large snags (22%) than on small snags (4%). Thirty-eight percent of yellow birch snags >30 cm had loose bark plates, and these accounted for over half the total loose-barked snags of this size.

Cavity tree density

Of the 2791 live trees and snags (>10 cm DBH) examined in all stands, 67 contained cavities. Although only 15% of these cavities were in snags, the proportion of stems with cavities was nearly twice as high for snags (3.9%) as for live trees (2.2%). That is, cavities were more common in live trees only because live trees themselves are so much more abundant than snags. More than 70% of the bird-excavated cavities in northern hardwood stands were located in snags or dead portions of live trees. Cavities resulting from natural wounds were more common than those excavated by birds (Table 3).

Estimated cavity tree density in selection stands of both forest types was 50–70% of the density in old-growth (Table 3). These differences were not statistically significant, however, because of great variability among plots (Table 1). Large trees (>45 cm DBH) containing cavities were present at a mean density of 11/ha in old-growth stands (both forest types combined) compared with 5/ha in selection stands (Table 3).

Fig. 4. Size distribution of fallen coarse woody debris volume showing a positive trend for both total volume of fallen debris and volume of large-diameter (>40 cm) fallen debris. Error bars show 1 SE for mean volume of large-diameter debris. Treatment means with the same letter are not significantly different at $P = 0.05$.



Cavity presence was clearly related to tree size. Mean diameters for live cavity trees were 73–104% larger than the overall mean live diameter for the stand type. The proportion of live trees with cavities was consistently greater for larger trees across all five forest type – management history combinations (Table 3). Old-growth stands had the highest proportion of all live trees (>10 cm) with cavities, but in large-diameter live trees the proportion with cavities was similar among selection and old-growth stands.

Snag volume

Mean snag volume in northern hardwood selection stands was very similar to levels in even-aged stands, but just over half the snag volume in old-growth stands (Table 4). More than 75% of standing snag volume in even-aged northern hardwoods was concentrated in trees <30 cm DBH. In contrast, over half of the snag volume in selection stands and over three quarters of the snag volume in old growth were from snags >45 cm DBH, which will eventually contribute to the pool of large-diameter fallen wood.

In old-growth hemlock–hardwoods, mean snag volume was greater than twice the level in northern hardwood old-growth, and standing snags accounted for 38% of total coarse woody debris volume (Table 4). Most of the snag volume in old-growth was yellow birch in early decay classes.

Volume and size distribution of fallen wood

Mean total volume of fallen wood (i.e., fallen boles, branches, stumps, or tops) in northern hardwood selection stands was about double the volume measured in even-aged, but only 60% of volume in old-growth (Fig. 4). These differences were

Table 3. Density and characteristics of trees with nesting cavities in stands of different forest type and management history.

	Northern hardwood			Hemlock–hardwood	
Characteristic	Even-aged	Selection	Old-growth	Selection	Old-growth
No. of trees searched	633	800	512	467	379
No. of cavities found	7	20	22	7	11
Cavity tree density (no./ha)					
Total (>10 cm DBH)*	10.8 (±3.3)	12.5 (±2.5)	18.1 (±4.9)	7.0 (±1.2)	13.8 (±4.3)
>45 cm DBH	0.0	5.2	11.4	5.0	8.8
>60 cm DBH	0.0	0.6	3.1	1.0	2.5
Mean DBH (cm)					
Live cavity trees	27.6	44.8	54.3	49.5	56.2
Snag cavity trees	23.1	50.7	31.2	25.0	36.2
% of total trees searched containing cavities					
Cavities in live trees					
Live >10 cm DBH	0.9	2.1	4.5	1.3	3.0
Live >30 cm DBH	3.6	5.5	9.0	4.8	6.7
Live >45 cm DBH	0.0	13.0	12.3	14.7	11.5
Cavities in dead snags					
Snags >10 cm DBH	3.8	6.5	3.4	3.9	2.1
Snags >30 cm DBH	0.0	16.7	3.1	0.0	4.4
Distribution of located cavities (% of total cavities)					
Live–dead status					
Live trees	71	79	91	86	91
Dead snags	29	21	9	14	9
Cavity opening size class					
0–5 cm	71	70	50	43	46
5–10 cm	29	30	36	43	46
10–25 cm	0	0	14	14	9
Origin of cavity opening					
Natural wound	57	70	77	86	64
Bird excavated	43	30	23	14	36
Cavity location in tree					
Lower bole	43	65	36	29	45
Upper bole	14	5	36	29	45
Live branch	0	5	9	29	0
Dead branch	43	15	18	14	9
Dead top	0	10	0	0	0
Additional habitat features					
Density (no./ha)					
Escape cavities†	24.3	15.6	23.6	8.0	18.7
Den trees (>45 cm DBH)	0.0	1.3	5.0	0.0	3.8

*Mean value for treatment followed by SE in parentheses.

†Includes cavities not meeting the criteria for nesting cavities as outlined in the Methods section.

statistically significant (Table 1). Large-diameter fragments (>40 cm) made up approximately 25% of total fallen wood volume in selection stands, averaging 16 m^3/ha. The volume of large-diameter material was significantly higher in old-growth northern hardwoods and contributed over 35% of the total fallen wood volume in these stands (Fig. 4). In the even-aged stands, large-diameter debris made up only 5% of the total fallen wood volume. Similar trends in total and large-diameter fallen wood volumes were found in hemlock–hardwood stands.

Decay class distribution of fallen wood

In addition to greater total volumes of fallen wood, old-growth stands generally had greater volumes across all decay classes relative to selection stands. In northern hardwood selection stands, volume of large-diameter (>40 cm), advanced decay (classes III–V) fallen wood was 11 m^3/ha, half the volume present in old-growth stands, but substantially higher than the 2 m^3/ha in even-aged stands. In hemlock–hardwoods, volume of large, advanced decay fallen wood in selection stands was 13 m^3/ha compared to 29 m^3/ha in old-growth.

Despite large volume differences among northern hardwood stand types, distributions of fallen wood among decay classes were similar (Table 4). Using three separate ANOVA tests, no significant differences were detected among stand types in the proportion of volume in each of three decay class groupings (I and II: $P = 0.57$, MSE = 179.9, df = 4, 26; III: $P = 0.83$, MSE = 59.8, df = 4, 26; IV and V: $P = 0.63$, MSE = 147.1, df = 4, 26).

Table 4. Distribution of fallen wood volume by size class, decay class, and origin for different stand types, and contribution of fallen wood and snags to the total coarse woody debris.

	Northern hardwood			Hemlock–hardwood	
	Even-aged	Selection	Old-growth	Selection	Old-growth
Decay class of fallen wood (% of volume)					
I	1	2	7	0.4	1
II	19	17	20	24	10
III	52	56	55	49	54
IV	25	21	16	24	19
V	3	4	2	3	16
Origin of fallen wood (% of volume)					
Fallen logs	82	41	81	22	73
Stumps (natural)	3	2	2	1	10
Stumps (cut)	3	11	0	28	0
Fallen branches	9	8	17	4	17
Harvest tops	3	38	0	45	0
Fallen wood volume (m^3/ha)*					
Mean	24.6*c*	61.3*b*	102.2*a*	56.0*b*	93.9*a*
SE	±4.1	±6.3	±6.4	±7.1	±10.6
Snag volume (m^3/ha)*					
Mean	14.6*bc*	13.1*bc*	25.4*b*	5.2*c*	57.5*a*
SE	±3.0	±3.7	±7.1	±1.1	±11.5
Total coarse woody debris volume (m^3/ha)*					
Mean	39.2*c*	74.4*b*	126.9*a*	61.1*b*	151.4*a*
SE	±5.8	±8.6	±12.3	±7.2	±18.1
Mass of coarse woody debris (Mg/ha)					
Fallen wood mass	6.0	14.9	28.7	13.9	20.3
Snag mass	5.5	3.8	7.7	1.5	19.8
Total mass coarse woody debris	11.5	18.7	36.4	15.4	40.1
No. of stands sampled	6	10	6	5	4

*Treatment means followed by the same letter are not significantly different at $P = 0.05$ (protected least significant difference).

Origin of fallen wood

Harvest tops and discarded (unmerchantable) portions of boles contributed much of the fallen wood volume in northern hardwood selection stands, with additional amounts from fallen trees and cut stumps (Table 4). In old-growth and even-aged northern hardwood stands, fallen boles accounted for more than 80% of the fallen volume.

Cut stumps constituted a larger proportion of the total fallen debris volume in hemlock–hardwood selection stands compared with northern hardwood selection stands. This difference was even more dramatic for large-diameter fallen wood, which in northern hardwood selection stands came primarily from logs. In hemlock–hardwood selection stands, 72% of large-diameter fallen wood volume was from cut stumps, nearly half of which was hemlock.

Recent tip-up mounds and pits

There were no statistically significant differences detected among treatments for percentage of stand area in recent pit and mound microtopography. Values were <1% of plot area in all five forest type – management history combinations. Among northern hardwood stand types, the percent area in pits and mounds was highest in selection stands, averaging 0.8% area from a density of 28 tree falls/ha. Some of the tip-ups in selection stands were caused by the felling of neighboring trees. Even-aged stands had the least area in pits and mounds (0.1%) from an average of 7 tree falls/ha. Recent tip-up mounds and pits in old-growth covered 0.6% of the area in northern hardwood and 0.4% in hemlock–hardwood stands.

Discussion

Since the early 1980s, efforts have been made to incorporate retention of snags and cavity trees into silvicultural treatments on many public forests. Selection (i.e., managed uneven-aged) stands in the present study exceeded the minimum target density (5–10/ha) for snags >30 cm recommended in local U.S. Forest Service guidelines (Evans and Conner 1979; DeGraaf and Shigo 1985). The mean levels of large snags, fallen wood, and cavity trees in these selection stands were generally intermediate between those measured in unmanaged even-aged and in old-growth stands. Snag densities (>10 cm DBH) in our selection stands were more than double those found by McComb and Noble (1980) in a managed uneven-aged hemlock–hardwood stand in Connecticut. Likewise, large-diameter fallen wood (>40 cm) was much more abundant in our selection stands than in the managed uneven-aged northern hardwood stand in New Hampshire sampled by Gore and Patterson (1986), which had no fallen stems >38 cm. The volume of debris >40 cm in diameter constituted over 28% of total fallen wood volume in our selection stands, owing to input from discarded portions of the lower bole, fallen snags, and stumps. Total and large-diameter fallen wood volumes in selection stands were significantly greater than those accumulated in the unmanaged second-growth stands of this study, and also exceeded levels reported for unmanaged mature (65–89 years old) and old (>100 years old) mixed mesophytic stands in Maryland (McCarthy and Bailey 1994).

Our estimates of fallen wood volume and mass for even-aged and old-growth northern hardwoods were within the range reported elsewhere for northern hardwood stands of comparable age (Tritton 1980; Gore and Patterson 1986; McCarthy and Bailey 1994). Likewise, the total coarse woody debris volume (i.e., snags and fallen wood) measured in our old-growth hemlock–hardwood stands approximated levels reported by Tyrrell and Crow (1994*a*) for their older hemlock stands containing trees >300 years old. Differences in coarse woody debris attributable to increasing stand age in our unmanaged stands matched trends reported for other eastern forests, including an increase in mean snag diameter and large snag density (Rosenberg et al. 1988; Tyrrell and Crow 1994*a*), an increase in the diameter and total amount of fallen debris (Tritton 1980; Gore and Patterson 1986; McCarthy and Bailey 1994), and a decrease in total snag density (McComb and Muller 1983; Carey 1983; Rosenberg et al. 1988). And as has been reported in other eastern deciduous forests, we found that nesting cavities were positively correlated with increasing tree diameter and that a large percentage of all cavities found were in live trees (Carey 1983; Healy et al. 1989; Welsh and Capen 1992).

Differences in coarse woody debris levels between selection and old-growth stands were most pronounced for large stems. Large-diameter snags and fallen boles have received particular attention in wildlife management guidelines (Thomas et al. 1979; DeGraaf and Shigo 1985; Tubbs et al. 1987) because they are preferentially selected by large-bodied species such as raccoon (*Procyon lotor*), fisher (*Martes pennanti*), marten (*Martes americana*), and pileated woodpecker (*Dryocopus pileatus*). For instance, all marten dens (natal and maternal) observed by Wynne and Sherburne (1984) in northwestern Maine were in large logs or trees >40 cm DBH, and the average size of nest trees selected by pileated woodpeckers is approximately 55 cm DBH (Evans and Conner 1979). Recommended densities of large-diameter (>45 cm DBH) cavity or den trees required to meet the needs of all cavity nesting species range from 0.35 to 2.5 trees/ha (summarized in Table 2 of Tubbs et al. 1987). Densities of large-diameter snags and cavity trees both exceeded 4/ha in our managed uneven-aged stands, although these values were only 21 and 50%, respectively, of the large snag and cavity tree densities that we measured in old-growth.

While low amounts of coarse woody debris can be limiting to wildlife populations (Haapanen 1965; Newton 1994; Carey and Johnson 1995), it is presently unclear whether population densities continue to increase along with coarse woody debris abundance beyond some moderate level of coarse woody debris availability (Raphael and White 1984). Certainly, other factors besides available habitat structure can influence population densities (e.g., territory size requirements, winter habitat, predator population levels). Raphael and White (1984) found that the density of all cavity nesting birds in the Sierra Nevada increased with the density of large snags (>38 cm DBH) until reaching a snag density of about 7.5/ha. Above this snag density level, bird densities were evidently limited by other factors. Maximum density of large snags reached similar levels of 6.7 snags/ha (>38 cm DBH) in our selection stands. However, bird survey data collected by other research project members in our study sites suggest that at these snag levels in the Lake States, population densities of cavity nesting birds may still be limited by available coarse woody debris habitat (Howe and Mossman 1996). Pileated woodpeckers and chimney swifts (*Chaetura pelagica*), two species that prefer snags >50 cm DBH (Evans and Conner 1979), were both significantly more abundant in old-growth than in selection stands (R.W. Howe and M. Mossman, Avian Productivity Study Progress Report, 1997). For eight other bird species generally associated with coarse woody debris (including woodpeckers, brown creeper, and others), trends in the average number of breeding pairs observed appear to be positively correlated with coarse woody debris availability. For instance, the average number of winter wren pairs (*Troglodytes troglodytes*) in old-growth northern hardwood stands was over two times greater than in selection stands and eight times greater than in even-aged stands (Howe and Mossman 1996). For six of the eight species, the number of breeding pairs was at least 30% higher in old-growth northern hardwood than in selection stands. For five of these species, breeding pair numbers were at least 25% higher in selection than in even-aged stands (>85% higher for four of eight species).

The reason for fewer large snags in selection stands compared with old-growth is not simply that trees are not allowed to reach large size. Dahir and Lorimer (1996) found that the average size of canopy trees at the time of death in old-growth northern hardwoods was 51 cm DBH, which is somewhat smaller than the maximum tree size of 60 cm DBH retained in most managed uneven-aged stands. However, active management generally attempts to reduce senescence-related mortality of large canopy trees through the selective retention of vigorous trees. Short intervals between harvests enable efficient salvage of any low-vigor trees from a variety of size classes before or shortly after death. In addition, mortality of 30–50 cm DBH trees is lower in unmanaged mature stands than for the same size classes in old-growth stands (Dahir 1994), partly because of less competition from larger trees. Thus, among large trees, the rate of snag formation appears to be lower in mature stands even without the intervention of management practices.

Because of the short time interval between cutting cycles in uneven-aged stands, managers have the opportunity to reassess and modify the dead wood structure each time the stand is marked for selection harvest. It appears that efforts to identify and retain wildlife trees during timber marking in managed uneven-aged stands may lead to snag and cavity tree abundances above the target levels currently recommended by regional U.S. Forest Service biologists. If it becomes a management objective to increase the density of large-diameter coarse woody debris in uneven-aged stands, this could be similarly accomplished by allowing designated reserve trees to live out their natural life-spans. This approach would provide additional cavity trees, future snags, and subsequent large fallen logs without requiring longer rotation ages for all trees in the stand.

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Forest Roads: A Synthesis of Scientific Information

Editors

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Forest Roads: A Synthesis of Scientific Information

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Abstract

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Effects of roads in forested ecosystems span direct physical and ecological ones (such as geomorphic and hydrologic effects), indirect and landscape level ones (such as effects on aquatic habitat, terrestrial vertebrates, and biodiversity conservation), and socioeconomic ones (such as passive-use value, economic effects on development and range management). Road effects take place in the contexts of environmental settings, their history, and the state of engineering practices, and must be evaluated in those contexts for best management approaches.

Keywords: Roads, roadless areas, forest ecosystems, geomorphology, hydrology, habitat fragmentation, biodiversity, nonmarket values, heritage values, economic development, grazing, mineral resources, fire.

Summary

Roads are a vital component of civilization. They provide access for people to study, enjoy, and commune with forested wildlands and to extract an array of resources from natural and modified ecosystems. Roads have well-documented, short- and long-term effects on the environment that have become highly controversial, because of the value society now places on unroaded wildlands and because of wilderness conflicts with resource extraction.

The approach taken in this report is to identify known and hypothesized road-related issues and to summarize the scientific information available about them. The report identifies links among processes and effects that suggest both potential compatible uses and potential problems and risks. Generalizations are made where appropriate, but roads issues and road science usually cannot be effectively separated from the specific ecologic, economic, social, and public lands management contexts in which roads exist or are proposed.

General Consideration of Road Networks at Intermediate and Large Scales

Across a forest or river basin, the access needs, economic dependencies, landscape sensitivities, downstream beneficial uses of water, and so on can be reasonably well defined, but these relations tend to differ greatly from place to place. An effective synthesis of road issues draws local experts together to thoroughly evaluate road and access benefits, problems and risks, and to inform managers about what roads may be needed, for how long, for what purposes, and at what benefits and costs to the agency and society.

Road effects and uses may be somewhat arbitrarily divided into beneficial and detrimental. The largest group of beneficial variables relates to access. We identified access-related benefits as harvest of timber and special forest products, grazing, mining, recreation, fire control, land management, research and monitoring, access to private inholdings, restoration, local community critical needs, subsistence, and the cultural value of the roads themselves. Nonaccess-related benefits include edge habitat, fire breaks, absence of economic alternatives for land management, and jobs associated with building and maintaining the roads.

Undesirable consequences include adverse effects on hydrology and geomorphic features (such as debris slides and sedimentation), habitat fragmentation, predation, road kill, invasion by exotic species, dispersal of pathogens, degraded water quality and chemical contamination, degraded aquatic habitat, use conflicts, destructive human actions (for example, trash dumping, illegal hunting, fires), lost solitude, depressed local economies, loss of soil productivity, and decline in biodiversity.

For each variable, we sought expert assistance from scientists actively engaged in research related to roads and asked them for information, with emphases on results and conciseness rather than exhaustive descriptions, in the following categories: issues relevant to their topic; science findings; an assessment of reliability, confidence, and limitations inherent in the data; the degree to which the information could be generalized to larger geographic scales than in the original research; secondary links for each topic to other topics; and the ability of the existing knowledge to address the issues raised.

Road development histories crucial to understanding their effects—All roads were not created equal and do not behave the same. Road networks differ greatly in how they developed through time and how they were laid out over terrain; they carry this history into their present performance. The geographic patterns of roads in forest landscapes differ substantially from place to place, with commensurate differences in environmental effects. For example, ridgetop, midslope, and valley floor roads all behave differently, based on the topography they cross, the degree and type of interaction with stream networks, their stability in and response to storms, and their effects on wildfire, wildlife, and vegetation. Distinguishing among the effects of building, maintaining, using, decommissioning, or abandoning roads is crucial because each of these actions affects the environment in many ways.

Knowledge of the state of road systems in national forests is inadequate—We currently lack sufficient information to develop a comprehensive history of the building and maintaining of national forest roads or their current condition. The inventories of the roads differ widely, in both content and status, and frequently lack sufficient information to define benefits, problems, and risks.

Roads create interfaces and ecotones—Roads are long, which creates large amounts of interface within the landscapes traversed. The strength of the interactions at these interfaces differs with time and space; it is controlled by the contrast between adjacent resource patches or ecological units. These interfaces may regulate the flow of energy and materials between adjacent systems. Such sites are sensitive. They have relatively high biodiversity, affect critical habitat for rare and endangered species, and serve as refuges and source areas for pests and predators.

Road management involves important tradeoffs—Almost all roads present benefits, problems, and risks, though these effects differ greatly in degree. Roads provide motorized access, which creates a broad spectrum of options for management but forecloses other options, such as nonmotorized recreation or wildlife refugia. Even a well-designed road system inevitably creates a set of changes to the local landscape, and some values are lost as others are gained; for example, road density and fish populations correlate negatively over a large area in the interior Columbia basin. The basin's environmental assessment shows that subbasins with the highest forest-integrity index were largely unroaded, and subbasins with the lowest integrity had relatively high proportions of moderate or greater road density. In general, greater short- and long-term watershed and ecological risks are associated with building roads into unroaded areas than with upgrading, maintaining, closing, or obliterating existing roads.

Confounding variables are difficult to separate from road-related ones—Changes in the habitat of terrestrial vertebrates, frequency of road kill, and transmission of forest diseases result from road use, not from the presence of the road itself. Separating effects of roads from other landscape and ecological modifications that result from changes in land use that roads enable is often impossible.

Direct Physical and Ecological Effects

Geomorphic effects of roads range from chronic and long-term contributions of fine sediment into streams to catastrophic mass failures of road cuts and fills during large storms. Roads may alter channel morphology directly or may modify channel flow and extend the drainage network into previously unchanneled portions of the hillslope. The magnitude of road-related geomorphic effects differs with climate, geology, road age, construction practices, and storm history. Improvements in designing, constructing, and maintaining roads can reduce road-related erosion at the scale of individual road segments, but few studies have evaluated long-term and watershed-scale changes to sediment yields when roads are abandoned or obliterated.

Roads have three primary effects on **hydrologic processes**: (1) they intercept rainfall directly on the road surface and road cutbanks and affect subsurface water moving down the hillslope; (2) they concentrate flow, either on the surface or in an adjacent ditch or channel; and (3) they divert or reroute water from paths it otherwise would take were the road not present. Problems of road drainage and transport of water and debris—especially during floods—are primary reasons roads fail, often with major structural, ecologic, economic, or other social consequences. The effect of roads on peak streamflow depends strongly on the size of the watershed; for example, capture and rerouting of water can remove water from one small stream while causing major channel adjustments in another stream receiving the additional water. In large watersheds, roads constitute a small proportion of the land surface and have relatively insignificant effects on peak flow. Roads do not seem to change annual water yields, and no studies have evaluated their effect on low flows.

Forest roads can significantly affect **site productivity** by removing and displacing topsoil, altering soil properties, changing microclimate, and accelerating erosion. The direct effect of roads on soil productivity is estimated to range from 1 to 30 percent of the landscape area in managed forest lands. Losses of productivity associated with road-caused accelerated erosion are site specific and highly variable in extent.

Natural populations of animal species are affected by **habitat fragmentation** caused by the presence of roads and by avoidance of areas near roads by some species and attractiveness to those areas by other species. Fragmented populations can produce increased demographic fluctuation, inbreeding, loss of genetic variability, and local extinctions. Roads fragment habitat by changing landscape structure, dissecting vegetation patches, increasing the amount of edge, decreasing interior area, and increasing the uniformity of patch characteristics. Road-avoidance behavior is characteristic of large mammals such as elk, bighorn sheep, grizzly bear, caribou, and wolf. Some studies have shown that the existence of a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates.

On the other hand, roads and their adjacent environment qualify as a distinct **habitat** and result in changes at the species, population, and landscape scales. Some species are associated with edges, including those that use roads as corridors to find food. Roads facilitate **biological invasion** in that disturbed roadside habitats are invaded by exotic (non-native) plant and animal species dispersed by wind, water, vehicles, and other human activities. Roads may be the first points of entry for exotic species into a new landscape, and the road can serve as a corridor for plants and animals moving farther into the landscape. Invasion by exotic species may have significant biological and ecological effects if those species are able to displace natives or disrupt the structure and function of an ecosystem.

Indirect and Landscape-Scale Effects

The effects of roads on **aquatic habitat** are believed to be widespread, although direct, quantitative cause-effect links are difficult to document. At the landscape scale, correlative evidence suggests that roads are likely to influence the frequency, timing, and magnitude of disturbance to aquatic habitat. Increased fine-sediment composition in stream gravel—a common consequence of road-derived sediments entering streams—has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes and can reduce benthic organism populations and algal production. Roads can act as barriers to migration, lead to water temperature changes, and alter streamflow regimes. Improper culvert placement where roads and streams cross can limit or eliminate fish passage. Roads greatly increase the frequency of landslides, debris flow, and other mass movement. At the landscape scale, increasing road densities and their attendant effects are correlated with declines in the status of some non-anadromous salmonid species.

Roads can cause a wide variety of effects to **terrestrial wildlife**. Some species, such as gray wolf and grizzly bear, are adversely affected by repeated encounters with people. Roads can increase harassment, poaching, collisions with vehicles, and displacement of terrestrial vertebrates, which affect many large mammals such as caribou, bighorn sheep, mountain goat, pronghorn antelope, grizzly bear, and gray wolf. It is estimated that 1 million vertebrates are killed annually on roads in the United States. Direct mortality of large mammals on forest roads is usually low, except for those with a home range straddling a road. Forest roads pose a greater hazard to slow-moving migratory amphibians than to mammals. Nearly all species of reptiles seek roads for cooling and heating. Vehicles kill many of them, making well-used roads a population sink.

Chemicals applied to and adjacent to roads can enter streams by various pathways. The effect on **water quality** depends on how much chemical is applied, the proximity of the road to a stream, and the weather and runoff events that move chemicals and ediments. Dust produced by vehicles moving on unpaved roads reduces **visibility** and generates airborne particulates that can pose health hazards, such as in areas with soils containing asbestiform minerals.

Direct Socioeconomic Effects

A variety of products harvested from forests are being transformed into medicinals, botanicals, decoratives, natural foods, and other products, called **nontimber** or **special forest products.** The harvest of these products usually depends on road access. The Forest Service is required by law to permit access to **private inholdings** but can require the owners to comply with standards that apply to building roads on or through national forest land.

Economic pressures affect roads and road use, and roads have multiple economic consequences. Both **benefits and costs** are associated with building, maintaining, and using forest roads. The **economic** effects relate to forest access and user-communities, including loggers, silviculturists, fuels managers, and recreationists. The network of roads on national forest lands has both positive and negative effects on most Forest Service land management programs. Reducing road densities could result in increased timber-harvesting costs, for example. Roads have replaced stock drives for transporting sheep and cattle to and from mountain grazing allotments. Road-related issues associated with **energy and mineral resources** are access rights, property rights, and benefits and detrimental effects. **Public recreational users** of national forests depend on roads for access. Altering the road networks will affect such uses differently across the landscape.

Indirect Socioeconomic Effects

The increasing density of roads in and adjacent to many forest, shrub, and rangeland areas is an important factor in the changing patterns of disturbance by **fire** on the landscape. Roads provide access that increases the scale and efficiency of **fire suppression**, and roads create linear firebreaks that affect fire spread. The benefits roads provide for fire prevention and fire management carries an associated cost: increased access has increased the role of human-caused ignitions. And road networks have resulted in changes in fuel patterns and fire regimes at the broad scale.

Roads also affect many less measurable attributes of the national forests, including **passive-use values:** those values that people hold for things they may not expect to use themselves but that they believe should exist for future generations. For example, building roads in roadless areas may reduce passive-use value significantly; decommissioning of roads may increase such value. But decommissioning of roads also is likely to reduce active-use values. Roads themselves sometimes have **heritage value** because of historical or cultural significance.

The aim of this synthesis is to focus on the scientific information about the benefits, uses, and physical and biological effects of forest roads. Because all aspects of roads in forests have become of great interest to the American public, research is underway in many domains. This document represents the information available as of the date of publication.

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Contents

Introduction and Objectives

Roads have become vital components of the human use of forested systems. Without roads, development of the economic activity critical to the quality of modern life would have been difficult, and roads remain central to many forest uses today. Roads provide access for people to study, enjoy, contemplate, or extract resources from natural and modified ecosystems. Building and maintaining roads is controversial, however, because of the kinds of uses they enable, concerns about their short- and long-term effects on the environment, and the value that society now places on unroaded wilderness (Cole and Landres 1996, Williams 1998).

Decisions about roads—locating, building, maintaining, and decommissioning them—are complex because of the many tradeoffs required. The statement by Chomitz and Gray (1996) that "rural roads promote economic development, but they also facilitate deforestation" exemplifies recent experiences. And a tradeoff exists between access by roads for recreation and resource extraction with the potential effects of that access on biodiversity. Roads have been evaluated from physical, biological, and socioeconomic points of view, often under only one perspective in isolation from the others. Such an approach is useful for identifying issues, but it can lead to conflict and poorly informed policy choices because it may unnecessarily play one set of values against another. For example, a road justified only by economic criteria at the expense of ecological ones—or vice versa—is likely to be questioned by advocates of the missing criteria. A unified approach to analyze building, maintaining, or decommissioning roads is needed to allocate resources wisely. This report represents our attempt to summarize the known desirable properties of roads and their known effects on the landscape, based on the scientific information currently available.

The approach taken was to enumerate the known or hypothesized issues and then provide a summary of the scientific information available about those issues. We provide a synthesis that attempts to reveal where links between processes and effects suggest both potential compatible uses and potential problems and risks.

We find that roads cannot be separated from the ecologic, economic, social, or public land management context in which they exist or are proposed. A virtually limitless variety of context factors renders any single, generalized synthesis to be of limited applicability and value. An effective synthesis of all the interactions of roads, the environment, and people can best be attempted by looking at road systems in actual places where the myriad effects of roads are not hypothesized or generalized. For example, across a national forest or river basin, the array of access needs, economic dependencies, landscape sensitivities, downstream beneficial uses of waters, and so on can be reasonably well defined and will tend to differ greatly from any other place. A synthesis of the effects of roads in a specific context can be attempted by drawing local experts together to thoroughly evaluate road and access benefits, problems, and risks, to inform managers about what roads may be needed, for how long, for what purposes, and at what costs to the agency and society.

The Forest Service recently published a document *Roads Analysis: Informing Decisions About Managing the National Forest Transportation System* (USDA FS 1999), which can be considered a specific application of watershed analysis or a cumulative effects analysis, wherein the principal objective is to focus on road effects. For example, roads analysis and watershed analysis have common steps that include:

- Setting up the analysis
- Describing the situation

- Identifying issues
- Assessing benefits, problems, and risks
- Describing opportunities and setting priorities
- Reporting results and conclusions

Similar approaches to watershed analysis or cumulative effects analysis are being adopted widely by federal (for example, Regional Ecosystem Office [REO] 1995), state (for example, Washington Forest Practices Board 1995), and private (for example, NCASI 1992) agencies and organizations. The exact steps and organization of the analysis are somewhat modified by each application, but the conceptual framework is similar. The focus of each analysis can change, depending on the principal reason for doing it (such as timber production, wildlife, or ecosystem integrity); for example, an analysis focused on timber production in a watershed or region would look at effects on and of road development, water quality, wildlife, recreation, and economics. Exactly the same set of issues would emerge if the focus were on water quality, wildlife, or recreation. The perspective and conclusions might be different, but the issues and approach would be the same.

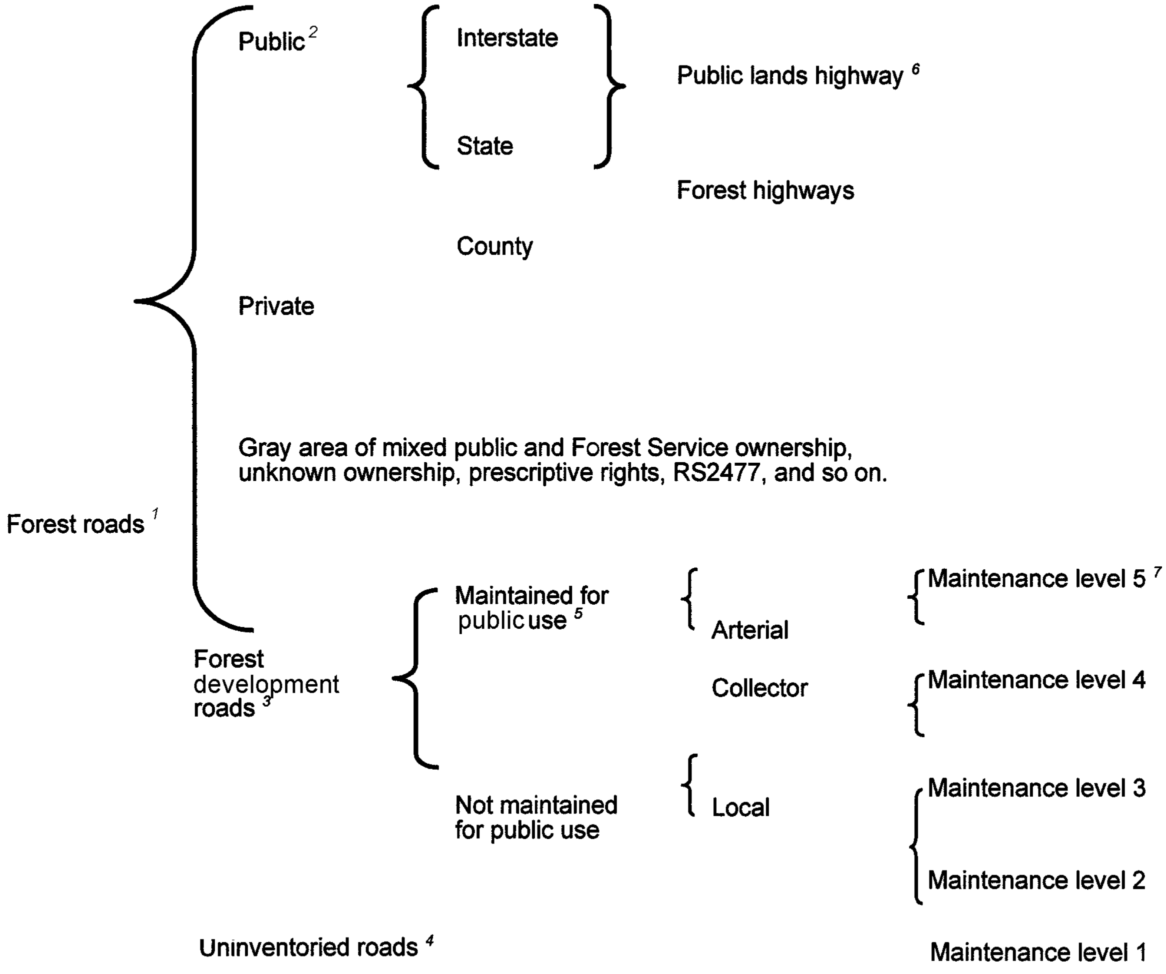
The roads analysis (USDA FS 1999) is intended to be an integrated, ecological, social, and economic approach to transportation planning. It uses a multiscale approach to ensure that the identified issues are examined in context, and it is based on science. Analysts are expected to locate, correctly interpret, and use relevant existing scientific literature in the analysis, disclose any assumptions made during the analysis, and reveal the limitations of the information on which the analysis is based. The analysis methods and the report are to be subjected to critical technical review.

This science synthesis complements the roads analysis by summarizing some of the available scientific information on how roads affect an array of ecological, social, and economic resources. The approach used in this document is mostly reductionist; it is not intended to be a comprehensive encyclopedia of all available knowledge about road effects; but this information, together with the extensive list of questions posed in the roads analysis, should assist interdisciplinary teams in understanding and applying the best available science appropriately to existing and potential road systems in specific geographic contexts, across the national forest system. Commonly used definitions for Forest Service roads are listed in figure 1.

General Considerations of Roads Networks

In this section, we consider what the body of scientific work on roads allows us to understand about how roads function in the landscape. This paper details specific positive and negative consequences of roads; here, we attempt to distill this information into key observations relevant to road policy considerations. The work is a synthesis of a large body of information from many sources. Inevitably, the synthesis creates potential for interpretations beyond the more generally accepted facts about roads contained in the rest of the document. Nevertheless, we believe they represent a reasonable set of principles consistent with the best scientific knowledge.

Road effects and uses may—somewhat artificially—be divided into beneficial and deleterious effects. In the former category, most variables relate to access, with a second group of beneficial uses not related to access. We identified the following access-related benefits or needs: timber acquisition, grazing, mining, recreation, fire control, land management, research and monitoring, access to private inholdings, restoration, community critical needs, subsistence, and the cultural value of the roads themselves. Non-access-related benefits or needs included edge habitat, fire breaks,



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[1] **Forest roads:** Roads wholly or partially within, or next to, and serving a national forest and needed to protect, administer, and use the national forest and to use and develop its resources.

[2] **Public roads:** Roads under the jurisdiction of, and maintained by, a public authority that are open to public travel

[3] **Forest development roads:** Forest roads under the jurisdiction of the Forest Service.

[4] **Uninventoried roads:** Short-term roads associated with fire suppression; oil, gas, or mineral exploration or development; or timber harvest not intended for forest-development transportation and not necessary for resource management. Regulations require revegetation within 10 years.

[5] **Maintained for public use:** Forest development roads open to unrestricted use by the general public in standard passenger cars, including those roads closed seasonally or for emergencies.

[6] **Public lands highways, forest highways:** A coordinated Federal Lands Highway Program includes forest highways, public lands highways, park roads, parkways, and Native American reservation roads under the jurisdiction of and maintenance by a public road authority other than the Forest Service and open to public travel.

[7] **Maintenance level 5:** Roads that provide a high degree of user comfort and convenience. Normally double lane, paved facilities, or aggregate surface with dust abatement; the highest standard of maintenance.

Maintenance level 4: Roads that provide moderate user comfort and convenience at moderate speeds. Most are double lane, and aggregate surfaced. Some may be single lane. Some may be dust abated.

Maintenance level 3: Roads open and maintained for travel by a prudent driver in a standard passenger car. User comfort and convenience are not considered priorities. Typically low speed, single lane with turnouts and native or aggregate surfacing.

Maintenance level 2: Roads open for use by high-clearance vehicles. Passenger car traffic is discouraged. Traffic is minor administrative, permitted, or dispersed recreation. Nontraffic-generated maintenance is minimal.

Maintenance level 1: These roads are closed. Some intermittent use may be authorized. When closed, they must have barricades, berms, gates, or other closure devices. Closures must exceed 1 year. When open, a road may be maintained at any other level. When closed to vehicular traffic, they may be suitable and used for nonmotorized uses, with custodial maintenance.

Figure 1—Legal basis and definitions for roads in the national forests.

the absence of economic alternatives for land management, some positive effects on water quality, and the jobs associated with building and maintaining these systems. This analysis uncovered factors that could lessen negative effects of roads by better integrating engineering approaches with knowledge of road effects.

Negative consequences include effects on hydrology, geomorphic features such as debris slides, sedimentation, habitat fragmentation, predation, road kill, invasion by exotic species, dispersal of pathogens, water quality such as chemical contamination, aquatic habitat, use conflicts, human actions (for example, trash dumping, illegal hunting, fires), the cost of lost solitude, local economies, soil productivity, communities, and biodiversity.

For each variable, we sought expert assistance from scientists actively engaged in research related to roads and asked for information in the following categories, with emphases on results and conciseness rather than exhaustive descriptions: issues relevant to the topic variable; science findings; an assessment of the reliability, confidence, and limitations inherent in the data; the degree to which the information could be generalized to larger geographic scales than those of the original research; the secondary links from this topic to other topics; and the ability of the existing knowledge to address the issues raised.

We note that the limitations of science set the bounds for subsequent interpretations, we offer a synthesis of the available scientific information, and we consider how these science-based observations might be used in developing future road policy.

Despite the shortcomings described, we believe that the available science on road effects can provide considerable guidance in evaluating benefits and costs associated with roads. Our interpretation of the scientific literature leads to the following observations.

Roads differ greatly—All roads are not created equal and do not behave the same. Road networks differ greatly in development through time and layout over terrain, and they carry this history into present performance. In many parts of the National Forest System, the major roads were built in the 1950s and 1960s, with secondary and tertiary feeder roads following as the road networks expanded into watersheds. In other areas, logging roads developed from previous road systems used for mining in the Rocky Mountain and Southwestern states or agriculture in the southern Appalachians, Ozarks, and New England. Thus, changes in road standards through time (for example, width, construction methods, position in the landscape) have affected different parts of road networks. Consequently, each road network commonly contains a collection of old and new types and standards of roads designed for various purposes that cross terrain of differing sensitivities. This mosaic of road segments has implications for how roads will be managed in the future (Gullison and Hardner 1993).

The geographic patterns of roads in forest landscapes differ substantially from place to place, with commensurate differences in environmental effects. In the glaciated terrain of southeastern Alaska, for example, main roads were built on the broad, major valley floors, and the high-value timber that grew on lower hillslopes was brought downhill to them. In forests along the west side of the Sierra Nevada in California, on the other hand, major roads were built along broad ridges, with secondary roads leading down into headwater areas. The main roads into western Oregon forests entered watersheds along narrow stream bottoms and then climbed the adjacent steep, unstable hillslopes to access timber extending from ridge to valley floor. These configurations, combined with local geology and climate, resulted in very different effects of roads on watershed, wildlife, vegetation, recreation, and disturbance processes.

Even in the same region, road effects differ by landscape position. Ridgetop, midslope, and valley floor roads all produce different effects, based on the topography they cross, the degree and type of interaction with stream networks, the stability and response to storms, and the effects on fire, wildlife, and vegetation.

Different phases of road development have different effects on the landscape. Distinguishing among the effects of building, maintaining, usage, decommissioning, or abandoning of roads is crucial because they usually affect the environment in several ways.

Road development history crucial to understanding effects—The effects of roads differ over time. Some effects are immediately apparent (such as loss of solitude or creation of edge), but others may require an external event, such as a large storm, to become visible (such as road-related erosion or mass movement). Still other effects may be subtle, such as increased susceptibility to invasion by exotics, pathogens noticed only when they become widespread in the landscape, or increased road use as recreation styles and motor vehicles change.

With time, roads often adjust to the ecosystems they are embedded in. Some segments blend with the landscape and reach a new ecological and hydrological balance, or better, a metastable state. Such a state will be different for a road transecting old-growth forest than for a road in an otherwise highly disturbed landscape. A critical issue in the decommissioning of a road is whether disrupting the new environmental balance created by the presence and aging of the road is desirable. As other segments of the road age, however, some features (such as culverts and disrupted subsurface drainage paths) become increasingly unstable; the probability of failure increases with road age. Sometimes, decommissioning a road can have significant environmental effects because the road has become part of the evolving landscape.

Decommissioning also can avert significant future environmental effects of the road. One last precaution in generalizing about the environmental effects of roads is to determine the age and condition of the road and evaluate the degree of landscape adjustment to the road and vice versa. Roads produce long-term legacies on the landscape. Many roads built by the Roman Empire centuries ago have disappeared from the landscape, but their legacies remain in the sediment layers of Italian lakes (Hutchinson 1973) and in strips of unique vegetation growing on limestone soils (derived from the limestone slabs used to build the road) in landscapes of acid podzolic soils (Detwyler 1971). In Lago di Montesori, Italy, the building and use of Via Cassia resulted in a pulse of eutrophication that lasted 2,000 years before it abated when the road was abandoned (Hutchinson 1973). Strips of fern populations in the Caribbean National and Luquillo Experimental Forests in Puerto Rico, serve as indicators of the skid trails abandoned more than six decades ago in these wet forests (Garcia-Montiel and Scatena 1994). These legacies are useful in historical reconstruction of landscapes because they help to explain the relevance of yesterday’s activities to today’s landscapes (Burel and Baudry 1990). In the process, more is learned about ecosystem resilience and how ecosystems continuously adjust to change.

We do not currently have sufficient information to develop a comprehensive picture of the construction or maintenance history or the current condition of the roads comprising our national forest road networks. Although much information on roads exists at a variety of scales (district, forest, region), and some national forests have invested in inventorying and developing road databases, no common framework or database exists for accessing road development information. For environmental consequences, little information exists on old, abandoned roads that still pose risks of failure. Other data important to defining effects, such as the location and configurations of road-stream crossings, are not available for most places. Without such a database, developing a comprehensive picture of where the road system currently stands, what parts of it need work, and where restoration activities should be focused will be difficult and analyses may be limited at best.

Knowledge of the State of Road Systems in National Forests Is Inadequate

Road inventories for the national forests are highly variable, frequently incomplete or inaccurate, and lack information needed to define benefits, problems, and risks. For most national forests, the inventory contains very limited, transportation-related data, such as road maintenance level and surface type. These data, though useful for some purposes, may be wholly inadequate to address such considerations as sedimentation hazards, migration barriers, landslide potential, road-stream connectivity, or other important aspects of the environmental effects of roads. Other useful data may exist in various forms, but because they are not systematically collected or maintained, they are nearly impossible to access for analysis. Without suitable data, some important aspects of the analysis of roads cannot proceed.

Roads create interfaces and ecotones—Because roads have great length, the interface surface between roads and the ecosystems of the landscape traversed is maximized. Naiman and Décamps (1997) recognized that the strength of the interactions at these interfaces differs with time and space, and it is controlled by the contrast between adjacent resource patches or ecological units. They compare these interfaces to semipermeable membranes regulating the flow of energy and materials between adjacent systems. They note that interfaces “have resources, control energy and material flux, are potentially sensitive sites for interactions between biological populations and their controlling variables, have relatively high biodiversity, maintain critical habitat for rare and endangered species, and are refuge and source area for pests and predators.” The road interface may be split into two zones (roadside and ecotone) to highlight the difference between vegetation along the roadside and vegetation in the zone at the interface of the road. That interface can be sharp or gradual and form an ecotone that differs from both the roadside and the adjacent natural ecosystem.

The width of the surface of a road differs from the width of its ecological influence (Auerbach and others 1997; Forman, in press; Forman and others 1997; Larsen and Parks 1997; Reck and Kaule 1993). For example, a road may be 30 feet wide, but it may influence an additional 80 feet of adjacent land because of disturbance during construction and the buffer zone for the pavement, making the road effectively 110 feet wide. That same road has an ecological influence that can extend an additional unknown distance from storm water runoff—influence over the home range of wildlife, geomorphic alterations upstream and downstream, distance its noise and dust carry, and views it provides.

Road management usually involves important tradeoffs—Almost all roads present benefits, problems, and risks, though these effects differ greatly in degree. Roads permit motorized access, which creates a broad spectrum of options for management but forecloses other options, such as wilderness, nonmotorized recreation, or some types of wildlife refugia. Even a well-designed and well-built road system inevitably creates a set of changes to the local landscape, and some values are lost as others are gained.

Tradeoffs accompany specific decisions about roads, such as construction method. Full-bench road construction, for example, may decrease the risk of fill slope failure, but it also may increase the potential for groundwater interception with attendant water quality risks.

In public wildlands management, road systems are the largest human investment and the feature most damaging to the environment. Thus the choices about what roads are needed, for what purposes, for how long, and at what cost—to public ecological resources as well as financial—are critical decisions in managing public lands.

Recent Efforts at Describing Roads in the Landscape May Be Helpful

Roads can be thought of as ecosystems—Synthesis of the effects of roads on terrestrial ecosystems may be facilitated by viewing roads as "techno-ecosystems," as recently described by Lugo and Gucinski (2000). Roads occupy ecological space (Hall and others 1992), have structure, support a specialized biota, exchange matter and energy with other ecosystems, and experience temporal change. Road "ecosystems" are built and maintained by people (techno-ecosystems; Haber 1990) and are characterized by open fluxes of energy and matter and a predominance of respiration over photosynthesis; that is, they are heterotrophic and highly subsidized systems. To appreciate that features associated with roads function as an ecosystem and interact with the surrounding forests requires thinking about the flow of materials, energy, and organisms along road corridors, vegetation zonation, the interaction with the human economy and human activity, and the external forces that converge on the road corridor (Donovan and others 1997; Forman 1995a, 1995b). (See fig. 2).

Roads connect and disconnect—Roads are corridors that can connect contrasting ecosystem types. Because roads provide a somewhat homogeneous condition through the length of the corridor, they provide opportunity for organisms and materials to move along the corridor, thereby increasing the connectivity (Merriam 1984) among those ecosystems interfacing with the road.

The degree of connectivity between roads and streams (that is, the number of stream crossings and areas where roads and streams are near enough to strongly interact) is recognized as a good general indicator of the interactions between the two and of potential effects roads can exert (Wemple 1994). Where both stream and road densities are high, the incidence of connections between roads and streams can be expected to also be high, resulting in more common and pronounced effects of roads on streams than in areas where road-stream connections are less common and dense. (fig. 3).

The economic benefits of roads could be seen as a function of connecting commodities, such as timber, minerals, recreational opportunities, and so on, with potential users.

Roads also can function to disconnect important features of ecosystems. Many roads built next to streams isolate or disconnect streams from their flood plains, with adverse effects to stream dynamics and associated aquatic biota. Roads can block the movement of some animals, such as wolves crossing wide roads or fish being blocked from their upstream movement by perched culverts.

Road density and fish populations correlate across a large area in the interior Columbia basin—One of the few examples of landscape-scale analysis of road influences has been the interior Columbia River basin environmental assessment (Quigley and others 1997). The evaluation of road density and forest and range integrity in that study may serve to illustrate landscape-scale interaction of roads with their surroundings. Forest and range indices of integrity were developed that showed sub-basins having the highest forest-integrity index were largely unroaded and comprised cold forest "potential vegetation groups," or a mixture of moist and cold forest groups. Of the five indicator variables used, the proportion of a subbasin composed of wilderness or roadless areas seemed most closely associated with subbasins having high integrity indices; 81 percent of the subbasins classified as having the highest integrity had

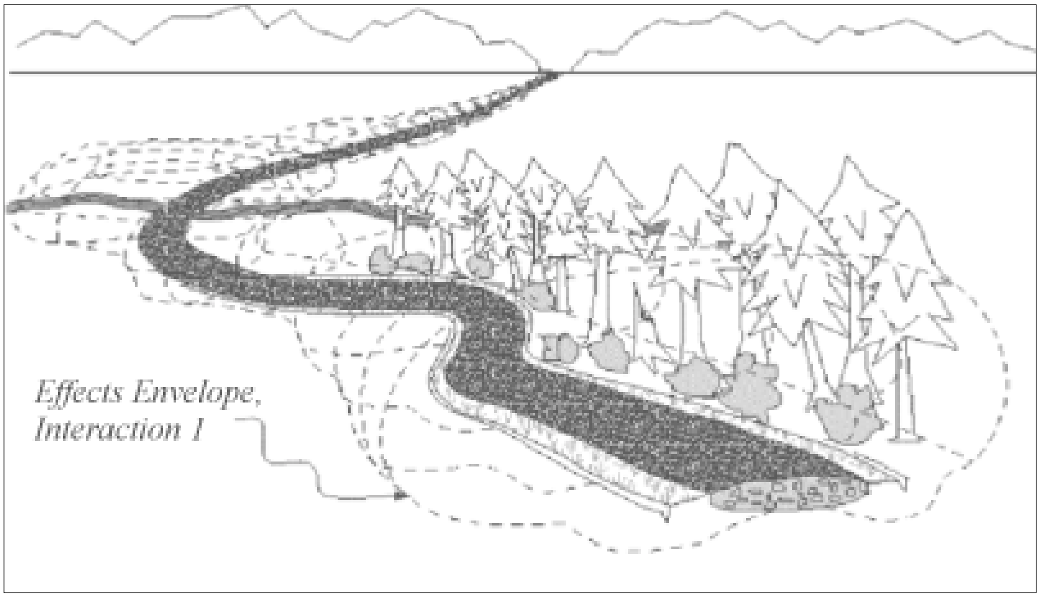

Figure 2—The volume of geographical space occupied by a road, whereby the distance of the road effect is used to define its width and height. The volume changes given the ecological conditions in the area the road traverses (from Lugo and Gucinski 2000).

relatively large proportions of wilderness and roadless areas (>50 percent). Conversely, of subbasins with the lowest integrity, 89 percent had low proportions of roadless and wilderness areas, 83 percent had relatively high proportions of at least moderate road density (0.27 miles/square mile). None of the seven subbasins having high rangeland integrity had areas of moderate or high road densities. The correlation of basin or subbasin integrity is not total, thereby suggesting that other variables and mechanisms are complex and nonuniform (but see text below for additional caveats).

Recreation surveys suggested the three most highly ranked uses of land administered by the Forest Service and Bureau of Land Management in the interior Columbia basin today are timber, fishing, and hunting. Projected major uses by 2045 will be a shift to motor viewing and day and trail use, even though this area has 70 percent of the unroaded areas of >200,000 acres remaining in the conterminous 48 states.

Strong fish populations were more frequently found in areas with low rather than high road densities. Supplemental analyses “clearly shows that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species.... They are less likely to use highly roaded areas for spawning and rearing, and, where found, are less likely to be at strong populations levels” (Lee and others 1997).

These findings are a “consistent and unmistakable pattern based on empirical analysis of 3,327 combinations of known species status and sub-watershed conditions, limited primarily to forested lands administered by BLM/FS” (Lee and others 1997). Although unroaded areas are significantly more likely than roaded areas to support strong populations, strong populations are not excluded from roaded watersheds. Possible reasons for this coexistence are that the inherent productivity of some areas allows fish populations to persist despite disturbances linked to roads; real or detectable effects on fish populations may lag behind the initial physical effects in watersheds where roads have been added in the last several years; and the scale of the subwatershed (18,000 acres

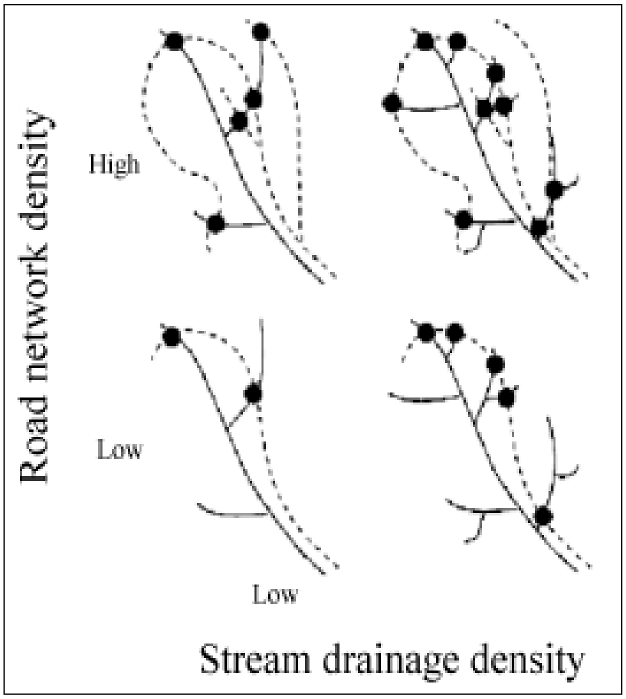

Figure 3—The incidence of road-stream connections, such as stream crossings (the black dots) is related to the density of both roads and streams in the landscape (Swanson and others 2000).

on average) at which strong populations are identified may mask a potential disconnect between the real locations of fish strongholds and roads (identified at resolution of 0.38 square mile). In general, greater short- or long-term watershed and ecological risks are associated with entering an unroaded area than with proceeding continuously with management activities in roaded areas to upgrade, maintain drainage, or close or obliterate existing roads.

Limitations of science—The existing science about roads goes far in establishing what and where problems are likely to arise. More than half a century of research and experience supports designing, building, and maintaining forest roads. Most of the major engineering problems associated with roads have been solved, and a wealth of information exists on many of the physical effects of roads, particularly on hydrologic and geomorphic watershed processes. Information on the biologic effects of roads is improving. Getting this knowledge into practice is more an economic, social, and political issue than a technical one. Less well understood but increasingly studied are the ways that the social and cultural settings of roads influence the benefits, problems, and risks that roads present.

Despite this extensive base of literature and understanding, a striking conclusion from our assessment of the current state of scientific understanding of roads is that virtually no attempt has been made to integrate this information into a comprehensive picture of how roads function in the landscape—physically, biologically, and socially. Despite the ubiquity of roads, no “science of roads” exists. Instead, many disciplines offer their perspectives: engineers study road design and performance, hydrologists evaluate effects of roads on water and sediment, ecologists consider effects on vegetation and wildlife, and transportation planners focus on road layout in relation to other forest resources and uses. Few efforts have been directed toward viewing the gamut of road benefits and effects systematically and simultaneously, or to developing general methods for evaluating risks posed by roads in individual watersheds. Further, the inventory and evaluation of roads is usually limited by ownership: The Forest Service focuses on roads in national forests and generally ignores roads within adjacent ownerships; states evaluate state highways; and the U.S. Department of Transportation evaluates federal highways.

We expect that implementing systematic analyses of road systems in national forests (as part of forest planning and other project planning; USDA 1999) will soon produce abundant examples of intermediate- and large-scale analyses. We hope that those analyses will look beyond ownership to produce a comprehensive evaluation of roads as a system. We have noted that the science information on the benefits of roads is not well developed. The form of scientific approaches for measuring benefits is largely based on economic analyses, which tend to focus on monetary cost differentials produced by the presence or absence of roads. Even in that arena, the data are not rigorously developed. Approaches from the social sciences are based on measurements of public perceptions and public desires, but the total data set does not comprise a highly developed scientific base.

Past studies (with the single, large exception of the interior Columbia River basin environmental assessment) have shed little light on the effects of roads across the whole landscape. Deciphering road effects at large spatial scales is difficult because past studies either focused on the performance of individual road segments, or else road effects were confounded by other simultaneous treatments. Most engineering studies, for example, look at the performance of specific road types (such as arterial, collector), features (road surfaces, cutslopes), or engineered structures (culverts) without examining how the road network functions in relation to adjacent hillslopes and an intersecting stream network. Where roads have been looked at in a watershed context, as in small watershed experiments, effects of roads often have not been distinguished from those of other treatments, such as logging or site preparation, that typically accompany roads. Treatments only of roads are rare and may continue for just a few years before other treatments are applied.

Despite the size of the forest road network, road effects have been examined in only a few places. Much of what we know about forest roads comes from studies in the Appalachians, Pacific Northwest, and Rocky Mountains—areas with known road problems. Given the wide variability in road history, age, construction methods, and use patterns in relation to topography, climate, and social setting, the narrow geographical scope of these studies limits their extrapolation to other regions or their usefulness in addressing more subtle effects.

Research has not typically considered an array of major effects and their interactions. We found only one study (either by way of case study or conceptual framework) addressing the broad range of major road effects. A recent report from the Transportation Research Board that addresses effects of motor vehicles—and by extension, roads—on climate and ecology focuses on the effects of vehicle emissions; only eight pages are devoted to a discussion of the effects of vehicle infrastructure (that is, roads), and the discussion of conserving biodiversity is limited to selected variables. Another recent paper focuses almost exclusively on the ecological damage posed by roads with scant attention to their potential benefits (Forman and others 1997). We know of no studies that provide a systematic way of evaluating risks and benefits from building, using, and removing roads. Such studies are needed to assess tradeoffs among the exceptionally diverse roles of roads in forest landscapes.

Recommendations

This overview of scientific information leads us to conclude that the emerging science of the effects of roads as networks in the landscape requires considerable new research. Because of the high degree of variability of roads from place to place and region to region, a framework for evaluating benefits, problems, risks, and tradeoffs among them would provide a powerful decisionmaking tool. We believe such a framework is now in place (USDA FS 1999). Conducting these analyses is well within the grasp of capable specialists, planners, and managers who can bring their expertise to the problem of reducing risks from past, current, or planned roads and targeting future road-restoration activities. The science pieces are already developed to analyze and integrate road systems and their effects.

Valid and useful analyses of road systems cannot proceed in the face of outdated, incomplete inventories lacking data needed to address important questions. Accurate and current road inventories that include information relevant to environmental effects analyses are needed.

Long-term and ongoing science initiatives would yield valuable information on how the effects of roads develop and change over time. Areas of research should include the effects of progressive road development and how road effects diminish or increase through time, even under constant road configuration. Some observations suggest, for example, that roads systems increasingly connect surface water flow paths to streams over decades, via gullies and landslides in steep terrain. Effects of road restoration practices also need to be evaluated in long-term studies, because both effects and practices are likely to evolve over time. Research on social and cultural perspectives on road use and presence is a key area for future work.

Organization of Sections That Follow

Several possible models might be used to organize a discussion of the ecological and physical effects of roads in forested landscapes. The most logical organization might start from the smallest scale of measurable effects and proceed to the landscape scales. At present, however, our knowledge is too imperfect and too fragmented to fully appreciate and integrate landscape-scale effects. Thus, we have used an approach that goes from the most direct effects to the secondary and indirect effects of forest roads. To a large degree, this model implies we will proceed from understanding effects of road segments to understanding effects of a road network.

We list physical effects first, stressing geomorphic and hydrologic processes, followed by effects on site productivity. Then we move to effects of habitat fragmentation, biological invasion, and other habitat changes that roads introduce. The direct effects—especially the physical ones, such as increased sedimentation and increased risk of slides and debris flows—are much affected by road design and placement on the landscape. Thus, when consequences of roads are aggregated at the landscape scale, the proportion of old roads to new ones that incorporate improved engineering design must be taken into account.

Indirect physical, biological, and landscape-scale effects, sometimes known only from empirical relations, constitute the next set, and include aquatic habitat effects both observed in instream consequences and broad-scale potential effects. Changes in the habitat of terrestrial vertebrates, road kill, and transmission of forest diseases by road traffic are even more complicated, in that they introduce effects not from the road itself, but from road use. Such effects clearly can be stopped by closing a road, but they also can be reduced or altered by changing patterns of road use, allowing for a range of options different from the options roads introduce just by their presence. Lastly, conserving biodiversity is such a broad and unexplained topic that we can sketch only a few of its aspects; we cannot state unequivocally what specific roles roads have in the interplay of populations, modified habitat, the new techno-ecosystem, road kill, and the complex ecological results when alien species modify forest landscapes. We also cannot separate the effects of roads from land-use changes on adjacent lands made accessible by roads; all modify species composition and survival of their populations.

We have addressed socioeconomic effects of roads in forest systems in a manner that follows the pattern introduced in the discussion of physical and biological effects: namely, we examine direct effects first, followed by a discussion of indirect effects or effects at a larger, landscape scale.

Some studies have separated road effects from land-use effects, including timber harvest on adjacent lands; other studies have not. Thus, this synthesis may have allowed these effects to be combined. Although we have made every effort to remove these confounding factors, the reader must carefully evaluate the data presented and consider to what degree we have succeeded.

The following sections are summary discussions of the interaction of roads with adjacent landscape components. They also briefly summarize the available information about the effects of roads on the environment and deliberately have been kept short with references provided for further study.

Direct Physical and Ecological Effects

Geomorphic Effects, Including Sedimentation and Landslides

Issues—More than 50 years of research and many case examples place the effects of forest roads on geomorphic processes squarely at the heart of the debate prompting reexamination of existing and future road networks on public lands. Geomorphic effects of forest roads range from chronic and long-term contributions of fine sediment into streams to catastrophic effects associated with mass failures of road fill material during large storms. The interactions of roads and land surfaces are often complex; for example, on one part of the hillslope, roads may trigger mass failures, and roads downslope from them may trap material derived from these failures. Roads and road building may alter channel morphology directly or may modify channel flow paths and extend the drainage network into previously unchannelized portions of the hillslope. Economic effects of road failures during storms has been discussed; less clearly understood are the cumulative or downstream consequences of road-related changes to geomorphic processes. Major issues motivating concern about road-related erosion include potential degradation of aquatic habitat and water quality and risks to public safety and structures downstream.

Findings—Roads affect geomorphic processes by four primary mechanisms: accelerating erosion from the road surface and prism itself by both mass and surface erosion processes; directly affecting channel structure and geometry; altering surface flow paths, leading to diversion or extension of channels onto previously unchannelized portions of the landscape; and causing interactions among water, sediment, and woody debris at engineered road-stream crossings. These mechanisms involve different physical processes, have various effects on erosion rates, and are not uniformly distributed either within or among landscapes. In steep forest lands prone to landsliding, the greatest effect of roads on erosion rates is from increased rates of mass soil movement after road building. Mass soil movements affected by roads include shallow (three to several feet deep) debris slides, deep-seated (depths of tens of yards) slumps and earth flows, and debris flows (rapid channelized and fluidized movements of water, sediment, and wood). Of these, effects of roads on debris slides and flows have been the most extensively studied, typically by landslide inventories using some combination of sequential aerial photography and ground verification. Accelerated erosion rates from roads because of debris slides range from 30 to 300 times the forest rate, but differ with terrain in the Pacific Northwest, based on a unit area in forest lands ranging from the U.S. Pacific Northwest to New Zealand (Sidle and others 1985). After the 1964 flood in the Pacific Northwest, Swanson and Dyrness (1975) documented increased rates of landslide frequency up to 30 times the rates in unmanaged forested areas. Similar inventories have been conducted elsewhere in the Western United States including Idaho (Megahan and others 1978), Washington (Reid 1981), and northern California, each documenting increased rates of landsliding in road areas relative to unmanaged forested areas. The magnitude of road-related mass erosion differs with climate, geology, road age, construction practices, and storm history. Several studies in the Eastern United States show that landslides are driven more by storm magnitude and geology than by land use. A threshold of 5 inches of rain per day (Eschner and Patric 1982) and metasedimentary geology are associated with large debris slides in the Appalachians. Road drainage can cause small slides in road fills; nevertheless, some major landslides originate in undisturbed forest land (Neary and Swift 1987, Neary and others 1986).

Road-related mass failure results from various causes. Typical causes include improper placement and construction of road fills and stream crossings; inadequate culvert sizes for water, sediment, and wood during floods; poor road siting; modification of surface or subsurface drainage by the road surface or prism; and diversion of water into unstable parts of the landscape (Burroughs and others 1976, Clayton 1983, Furniss and others 1991, Hammond and others 1988, Larsen and Parks 1997, Larsen and Simon 1993). Effects of roads on deep-seated mass movements have been much less extensively studied, although cases are documented of road building apparently accelerating earth-flow movement. This can occur by destabilizing the toe area or diverting water onto the earth-flow complex (Hicks 1982). Little is documented about the potential for increased mass failures from roads resulting from decay of buried organic material that has been incorporated into road fills or landings during road building. Anecdotal evidence is abundant that failures occur predictably after decay of the organic material.

Although mass erosion rates from roads typically are one to several orders of magnitude higher than from other land uses based on unit area, roads usually occupy a relatively small fraction of the landscape, so their combined effect on erosion may be more comparable to other activities, such as logging. Studies by Swanson and others (1981) in the Oregon Coast Range, for example, showed that although unit-area erosion from roads was 30 times greater than the increase from clearcutting alone, road-related landslide erosion accounted for just three times as much accelerated slide erosion in the watershed when the area in roads and clearcuts was taken into account. Road and clearcut erosion were nearly equal in a study in the west side of the Cascade Range in Oregon (Swanson and Dyrness 1975). In the Klamath Mountains of southwest Oregon, erosion rates on roads and landings were 100 times those on undisturbed areas, but erosion on harvested areas was 7 times that of undisturbed areas (Amaranthus and others 1985).

A related point is that only a few sites can be responsible for a large percentage of the total erosion. For example, major erosional features occupied only 0.6 percent of the length of roads studied by Rice and Lewis (1986).

Although road location, design, construction, and engineering practices have improved markedly in the past three decades, few studies have systematically and quantitatively evaluated whether these newer practices result in lower mass erosion rates (McCashion and Rice 1983). Retrospective analysis of road-related landslides in the Oregon Coast Range suggests some reduction in slide frequencies because of improved road siting and building (Sessions and others 1987). No large storms occurred during the study period, however, so these practices remain largely untested. Currently, several studies are ongoing to evaluate road-related mass movements and the influence of road design after several large floods in 1996 in the Pacific Northwest and 1997 in California. These studies are likely to substantially improve understanding of whether "best management practices" are effective in reducing mass erosion from roads, and which specific practices influence mass failure response.

Surface erosion from road surfaces, cut banks, and ditches represents a significant and, in some landscapes, the dominant source of road-related sediment input to streams. Increased sediment delivery to streams after road building has been well documented in the research literature for the Pacific Northwest and Idaho (Bilby and others 1989, Donald and others 1996, Megahan and Kidd 1972, Reid and Dunne 1984, Rothacher 1971, Sullivan and Duncan 1981) and in the Eastern United States (Kochenderfer and others 1997; Swift 1985, 1988). Rates of sediment delivery from unpaved roads are highest in the first years after building (Megahan and Kidd 1972) and are closely correlated to traffic volume on unpaved roads (Reid and Dunne 1984, Sullivan and Duncan 1981). Surface-erosion problems are worst in highly erodible terrain, particularly landscapes underlain by granite or highly fractured rocks (Megahan 1974b, Megahan and Ketcheson 1996). In the Eastern United States, poorly designed and managed forest access and county roads are major sources for higher sediment input rates to streams (Hansen 1971, Patric 1976, Van Lear and others 1995). Roads were identified as the major source of sediment in the Chattooga River basin, where 80 percent of the road sources are unpaved, multipurpose roads (forest and county) paralleling or crossing tributary streams (Van Lear and others 1995). The largest sediment losses were during road building and before exposed soils were protected by revegetation, surfacing, or erosion control materials (Swift 1985, 1988; Thompson and others 1996; Vowell 1985). Soil loss from skid roads in West Virginia ranged from 40 tons/acre during logging, to 4 tons/acre the first year after logging, to 0.1 ton/acre 1 year after logging was completed (Hornbeck and Reinhart 1964). Raw ditch lines and roadbeds are continuing sources of sediment (Miller and others 1985), usually because of lack of maintenance, inadequate maintenance for the amount of road use, excessive ditch line disturbance, or poorly timed maintenance relative to storm patterns (Swift 1984, 1988).

Extensive research has demonstrated that improved design, building, and maintenance of roads can reduce road-related surface erosion at the scale of individual road segments. Key factors are road location, particularly layout relative to stream systems (Swift 1988, USDA FS 1999), road drainage (Haupt 1959), surfacing (Burroughs and King 1989, Kochenderfer and Helvey 1987, Swift 1984), and cut slope and fill slope treatments (Burroughs and King 1989, Swift 1988). Many studies show that surfacing materials and vegetation measures can be used to reduce the yield of fine sediment from road surfaces (Beschta 1978, Burroughs and others 1984, Kochenderfer and Helvey 1987, Swift 1984).

Few studies have evaluated long-term and watershed-scale changes to sediment yields as roads are abandoned, obliterated, or restored. Personnel at Redwood National Park are undisputed experts in road restoration at a watershed scale; they have developed, tested, and applied road-restoration techniques at a scale virtually unprecedented throughout the world (Ziemer 1997). Since Redwood National Park was expanded in 1978, 134 miles of the 300 miles of road within park boundaries have been restored or obliterated. This work has removed about 1,300,000 cubic yards of material from stream crossings, landings, and unstable road benches. The volume of material is about equal to the long-term average annual sediment discharge near the mouth of Redwood Creek (Ringgold, n.d.). To evaluate the success of removing this volume of material, the delivery mechanism, timing, and proportion of the removed material that actually would have found its way to the channel without the restoration activity, the quantity of new material introduced by erosion caused by the restoration work itself, and the relative proportion of the treated areas compared to untreated areas at comparable risk in the basin must be known. Such evaluations are uncommon.

Roads interact directly with stream channels in several ways, depending on orientation to streams (parallel, orthogonal) and landscape position (valley bottom, midslope, ridge). The geomorphic consequences of these interactions, particularly during storms, are potentially significant for erosion rates, direct and off-site effects on channel morphology, and drainage network structure, but they are complex and often poorly understood. Encroachment of forest roads along the mainstem channel or flood plain may be the most direct effect of roads on channel morphology in many watersheds. Poorly designed channel crossings of roads and culverts designed to pass flow also may affect the morphology of small tributary streams, as well as limit or eliminate fish passage. Indirect effects of roads on channel morphology include the contributions of sediment and altered streamflow that can alter channel width, depth, local gradients, and habitat features (pools, riffles) for aquatic organisms (Harr and Nichols 1993).

Roads in midslope and ridgetop positions may affect the drainage network by initiating new channels or extending the existing drainage network. By concentrating runoff along an impervious surface, roads may decrease the critical source area required to initiate headwater streams (Montgomery 1994). In addition, concentrated road runoff channeled to roadside ditches may extend the channel network by eroding gullies or intermittent channels on hillslopes and by linking road segments to small tributary streams (Weaver and others 1995, Wemple and others 1996a). These effects of roads on the channel network have implications for slope stability, sedimentation, and streamflow regimes.

An emerging focus of the postflood studies in the Pacific Northwest is the importance of designing roads to accommodate disturbances (see “Hydrologic Effects” below), particularly in the area of road-stream crossings, which are implicated in most documented road failures (Furniss and others 1997). Another facet of this research is recognizing that roads can serve both as sources (by initiating landslides) and sinks (by trapping debris flows) of sediment during large events (Wemple and others 1996a).

Reliability of findings—These findings represent a broad synthesis of more than 50 years of research on geomorphic effects of roads in a wide range of physiographic and land-use settings. Although they are generally well supported by field, small watershed, and plot studies, specific effects of roads are strongly influenced by local factors, including road building techniques, soil and geology, precipitation and runoff regimes, and topography. As with hydrologic studies, evaluating effects of roads on geomorphic processes is further limited by the short timeframes (one to several years) during which such effects typically are monitored. Few studies have placed road effects in a broad landscape or watershed setting.

Generalizability—Most studies of roads have been conducted in only a few landscapes (the Pacific Northwest, Rocky Mountains, Appalachians, interior highlands, and Piedmont), so the ability to generalize to other terrains is limited. Statements about effects of roads on mass erosion are limited to those landscapes affected by such processes. A large part of the United States, including the Central States, Piedmont, and the coastal plain in the East, do not experience mass erosion processes in the forest. For the most part, only historical road-building practices (pre-1990) have been rigorously evaluated, either by scientists or by the landscape itself through large floods. Little is known, however, about geomorphic effects of old mining and arterial roads (older than 50 years).

Secondary links—The geomorphic and hydrologic effects of roads are closely related. Restoration strategies to reduce either geomorphic or hydrologic effects are likely to be quite different, however, which underscores the need to clearly identify objectives for restoration. For example, practices to reduce road network extension of surface flow paths by draining water back into the subsurface could have the unintended consequence of destabilizing fill slopes. Both the mass erosion and fine-sediment delivery issues are closely linked to concerns about aquatic habitat.

Conclusions—As with the hydrologic issues, evaluating geomorphic effects of roads needs to be addressed at several scales: individual road segments, intermediate-sized watersheds, and the entire road network in the river basin (which may include private lands and roads and roads built for a broad range of purposes, not just forest operations). Key directions for future research work are to systematically evaluate the relation between improved road practices and mass-erosion rates, particularly in light of mid-1990s floods in the Pacific Northwest and California; develop a conceptual and analytical framework for evaluating how roads in different landscape positions (valley bottom, midslope, ridgetop) interact with streams; develop empirical data on the amount of drainage-network extension and drainage-density increases resulting from roads in different geomorphic settings; and place geomorphic effects of roads in broader landscape contexts by using sediment budget and disturbance budget approaches.

Hydrologic Effects

Issues—The interaction between forest roads and water lies at the heart of several key issues surrounding the effects of roads on the environment. At the scale of individual road segments, designing and building roads to drain or channel water away from the road surface is one of the main problems facing road engineers, and it reflects the substantial effects that roads can have on hillslope hydrology. Road drainage problems and water and debris passage problems—especially during floods—are primary reasons for road failure, often with major structural, ecologic, economic, or social consequences. For example, of the $178 million spent on flood recovery on Forest Service lands in the Pacific Northwest Region after the 1996 floods, more than 70 percent was to fix road damage; most of the damage resulted from water drainage problems that, in turn, triggered mass movements (Cronenwelt, n.d.). At a broader scale, roads can influence the size and timing of streamflows from watersheds, with possible consequences for downstream channels and aquatic ecosystems. For these reasons, many road restoration projects are explicitly or implicitly focused on the ways roads influence the routing of water, with consequences for erosional processes.

Findings—Roads have three primary effects on water: they intercept rainfall directly on the road surface and road cutbanks and intercept subsurface water moving down the hillslope; they concentrate flow, either on the surface or in an adjacent ditch or channel; and they divert or reroute water from flow paths that it would take were the road not present. Most hydrologic and geomorphic consequences of roads result from one or more of these processes. By intercepting surface and subsurface flow, for example, and concentrating it through diversion to ditches, gullies, and channels, road systems effectively increase the density of streams in the landscape. This changes the amount of time required for water to enter a stream channel, which alters the timing of peak flows and hydrographic shape (King and Tennyson 1984, Wemple and others 1996a). Similarly, concentration and diversion of flow into headwater areas can cause incision of previously unchanneled portions of the landscape and initiate slides in colluvial hollows (Mongomery 1994). Diversion of streamflow at road-stream crossings is a key factor contributing to road failure and erosional consequences during large floods (Furniss and others 1998, Weaver and others 1995).

Hydrologically, different parts of the road system behave differently. All roads are not created equal and do not perform the same during storms, and the same road segment may behave differently during storms of different magnitudes. Recent, detailed examination of hydrographs at stream crossings with culverts shows that during the same storm, some road segments contribute substantially more flow to channels than others, primarily owing to differences in the amount of subsurface water intercepted at the cut bank (Bowling and Lettenmeier 1997, Wemple and others 1996b). As storms become larger or soil becomes wetter, more of the road system contributes water directly to streams. Slope position has a profound effect on the magnitude of hydrologic change caused by roads. Discharge from hill slopes, height of cut bank, density of stream crossings, soil properties, and response to storms all differ with slope position.

Although hydrologic effects of roads have been studied for more than 50 years, systematic studies with long-term measurement of the full range of potential interactions between water and roads are few. Most studies have emphasized geotechnical issues, including road design, culvert size and placement, and erosion control from road surfaces (see Reid and others 1997, for bibliography; Swift 1988). Of those studies that have attempted to look at the hydrologic behavior of roads, most have been part of small (typically 0.3 to 2 square miles) watershed experiments, where roads were a component of the experimental treatment, which often included other silvicultural practices. Key studies and locales of this type include those by Rothacher (1965, 1970, 1971, 1973), Harr and McCorison (1979), Harr and others (1975), Jones and Grant (1996), and Thomas and Megahan (1998) in western Oregon; Ziemer (1981, 1998) and Wright and others (1990) in northern California; King and Tennyson (1984) in central Idaho; Reinhart and others (1963), Hewlett and Helvey (1970), Swank and others (1982, 1988) in the southern Appalachians, Helvey and Kochenderfer (1988) in the central Appalachians; and Hornbeck (1973) and Hornbeck and others (1997) in the northern Appalachians. Very few studies have focused on the hydrologic behavior of roads alone; in the Pacific Northwest and Rocky Mountains, maximum measurement periods during which roads were the only treatment range from 1 to 4 years (Wemple 1994). Most studies have been conducted as “black box” experiments comparing streamflow hydrographs before and after road building, with little ability to identify key processes. Exceptions include the work of Megahan (1972), Keppeler and others (1994), and Wemple (1994) on subsurface flow interception and Luce and Cundy (1994) and Ziegler and Giambelluca (1997) on road-surface runoff. Few studies have focused on road effects, on hydrology in arid or tropical areas, or on areas dominated by snow hydrology, permafrost, and wetlands.

Even fewer published studies have explicitly considered how road networks affect the routing of water through a basin. We therefore have little basis to evaluate the hydrologic functioning of the road system at the scale of an entire watershed or landscape. Few published studies to date have identified how roads in different landscape positions might influence the movement of water through a basin. Montgomery (1994) looked at the effect of ridgetop roads on channel initiation, and Wemple (1994) documented the magnitude of drainage network enlargement caused by roads in different slope positions.

Based on studies of small watersheds, the effect of roads on peak flows is detectable but relatively modest for most storms; insufficient and contradictory data do not permit evaluation of how roads perform hydrologically during the largest floods. Roads do not appear to affect annual water yields, and no studies have evaluated their effects on low flows. In some studies, roads produced no detectable change in flow timing or magnitude (Rothacher 1965, Wright and others 1990, Ziemer 1981), but in other basins, average time to storm peak advanced and average peak magnitude increased after road building for at least some storm sizes (Harr and others 1975, Jones and Grant 1996, Thomas and Megahan 1998). In a study in Idaho, peak stormflow magnitude increased in one basin and decreased in another after road building, an effect the authors attribute to subsurface flow interception by roads and desynchronization of delivery of water to the basin outlet (King and Tennyson 1984). A whole-tree logging operation in New Hampshire that resulted in 12 percent of the area in roads (Hornbeck and others 1997) showed a maximum average increase of growing-season peak flows of 63 percent in the second year after harvest. This increase disappeared as the forest regenerated, and only 2 of the 24 peak flows in the 6th through the 12th growing seasons showed statistically significant increases. Dormant-season peak flows generally decreased because cutting changed snowmelt regimes. Helvey and Kochenderfer (1988) concluded that typical logging operations in the central Appalachians do not increase flows sufficiently to require larger culverts to accommodate them. Forest harvesting without roads in the southern Appalachians increased stormflow volumes by 11 percent and peak flow rates by 7 percent (Hewlett and Helvey 1970, Swank and others 1988). Harvesting an adjacent watershed with 4 percent of the area in roads increased stormflows by 17 percent and peak flows by 33 percent. Four years later, peak flows dropped to a 10-percent increase after 40 percent of the road system was closed and returned to forest (Douglass and Swank 1975, 1976). Collectively, these studies suggest that the effect of roads on basin streamflow is generally smaller than the effect of forest cutting, primarily because the area occupied by roads is much less than that occupied by harvest operations. Generally, hydrologic recovery after road building takes much longer than after forest harvest because roads modify physical hydrologic pathways, but harvesting principally affects evapotranspiration processes. The hydrologic effect of roads depends on several factors, including the location of roads on hillslopes, characteristics of the soil profile, subsurface water flow and ground-water interception, design of drainage structures (ditches, culverts) that affect the routing of flow through the watershed, and proportion of the watershed occupied by roads.

Most road problems during floods result from improper or inadequate engineering and design, particularly at road-stream crossings but also where roads cross headwater swales or other areas of convergent groundwater. Road redesign that anticipates and accommodates movement of water, sediment, and debris during infrequent, but major storms should substantially reduce road failures and minimize erosional consequences when failures occur. Recent studies after large floods in the Pacific Northwest highlight the importance of water diversion by roads and road-related structures (that is, plugged culverts, ditches) in contributing to road-related failures (Donald and others 1996, Furniss and others 1997). A typical failure resulted from culverts sized only to accommodate the flow of water, but not the additional wood and sediment typically transported during major floods. The culverts became obstructed and diverted water onto the road surface, into neighboring drainages unable to adjust to the increase in peak flow from the contributing basin, or onto unchanneled hillslopes. “Cascading failures” were common, where diversion or concentration of flow led to a series of other events, ultimately resulting in loss of the road or initiation of landslides and debris flows. Analysis of the probability of large floods and how they relate to the design life of roads indicates that most road crossings are likely to have one or more large floods during their lifetimes. Consequently, designing roads with large storms in mind is prudent and well within the reach of current engineering practices (Douglass 1977; Furniss and others 1991, 1997; Helvey and Kochenderfer 1988). The potential for stream diversion on wildland roads indicates that the environmental consequence of road failure during large storms is an option to consider.

Although the ability to measure or predict the hydrologic consequence of building or modifying a specific road network might be limited, general principles and models can be provided that, if followed, may decrease the negative hydrologic effects of roads. These principles will be useful during upgrading or decommissioning of roads to meet various objectives. A partial list of principles includes:

- Locate roads to minimize effects; conduct careful geologic examination of all proposed road locations.
- Design roads to minimize interception, concentration, and diversion potential, including measures to reintroduce intercepted water back into slow (subsurface) pathways by using outsloping and drainage structures rather than attempting to concentrate and move water directly to channels.
- Evaluate and eliminate diversion potential at stream crossings.
- Design road-stream crossings to pass all likely watershed products, including woody debris, sediment, and fish—not just water.
- Consider landscape location, hillslope sensitivity, and orientation of roads when designing, redesigning, or removing roads.
- Design with failure in mind. Anticipate and explicitly acknowledge the risk from existing roads and from building any new roads, including the probability of road failure and the damage to local and downstream resources that would result. Decisions about the acceptable probability and especially consequences of failures should be informed through explicit risk assessments. The many tradeoffs among road building techniques to meet various objectives must be acknowledged. For example, full bench road construction may result in lower risk of fill slope failure, but it also may increase the potential for groundwater interception; outsloping of the road tread may reduce runoff concentration on the road surface but also increase driving hazard during icy or slippery conditions.

Reliability of findings and generalizability—Hydrologic effects of roads are strongly influenced by landscape condition, road design and construction, and storm history. Generalizability of paired-watershed studies is limited by the short timeframes (one to several years) during which road effects alone are typically monitored. In addition, most road studies have been done in only a few landscapes where road problems are common (the Pacific Northwest, Rocky Mountains, and Appalachians), thereby limiting the ability to generalize to other terrain. The general principles represent reasonable interpretations of the available scientific knowledge, however. Some landscapes may be much more sensitive than others to certain key processes, such as interception of subsurface flow and drainage network extension resulting from gullying. For this reason, the specific range of hydrologic effects likely to be encountered needs to be evaluated by both regional and landscape scales.

Secondary links—The hydrologic effects of roads are strongly linked to their sediment and geomorphic effects. Other links can be found with wildlife (for example, road-created wetlands) and invasion by exotics (for example, microclimate related to water availability above and below the road prism), but these links have received little scientific attention.

Conclusions—Future efforts to redesign, restore, or remove road systems because of hydrologic concerns should have clear objectives: What hydrologic processes are considered problems? Where do they occur? What can be done about them? What degree of hydrologic alteration is considered acceptable? This type of evaluation of roads is best accomplished in the context of a watershed analysis (USDA FS 1999). Key areas for future research are to develop analytical models that allow managers to display the predicted hydrologic consequences of alternative road-network designs (these types of models are still in their infancy but should be more widely available in the next 2 to 3 years), expand process-based studies of how roads affect specific hydrologic mechanisms (for example, subsurface flow interception or channel network extension) in different geomorphic settings evaluate at the landscape scale the extent of links between the road and stream networks in different landscapes, and relate type and size of road failures to specific design practices and landscape position.

Site Productivity

Issue—The presence of roads commits a soil resource, and where roads occupy formerly productive land, they affect site productivity.

Findings—Forest roads can have significant effects on site productivity by removing and displacing topsoil, altering soil properties, changing microclimate, and accelerating erosion. The direct effects of taking land out of production by removing trees and displacing soil, or removing soil during building and maintaining roads, has been estimated to range from 1 to 30 percent of the landscape area in managed forest lands (Megahan 1988a). In the Western United States, tractor and ground-cable systems average about 10 percent of the area affected by roads to support harvest operations, and skyline and helicopter operations average 2 percent (Megahan 1988b). Studies in Eastern U.S. forests have consistently found that 4 to 5 percent of the total forested area is taken out of forest production by building roads during logging operations, although more than 50 percent of this area may be reforested within 8 years, but at reduced growth rates and productivity. Total road length required to support logging operations depends on the harvest and silvicultural systems and topographic configuration, but the area disturbed may be surprisingly consistent (Douglass and Swift 1977, Robinson and Fisher 1982, Swank and others 1982, Swift 1988).

Measurable declines in tree growth are common where soil is excavated to build the road prism. Evidence of off-site effects of roads on productivity is conflicting, though road-associated mass erosion may scour soil from steep slopes. Road building changes soil physical properties including depth, density, infiltration capacity, water holding capacity, and gas exchange rate, nutrient concentrations, and microclimate. Fertile topsoils, often containing most of the organic matter and plant nutrient capital of a site, frequently are buried under road fills or sidecast and may be rendered inaccessible to plant roots. Trees can grow on any portion of a closed road, but they can grow only on cut and fill slopes on open roads. Sites are harshest and soils poor or nonexistent on road cuts and the cut portion of road treads. Tree height and diameter growth is reduced on these portions of the road (Smith and Wass 1979, 1980, 1985). Growth is sometimes enhanced on or below fill portions of roads because of reduced competition and greater soil depth. Pfister (1969) documents a 30-percent increase in height growth of western white pine (*Pinus monticola* Dougl. ex D. Don) adjacent to outsloped roads. Megahan (1988a) suggests that this increase is due to enhanced soil moisture below outsloping roads. Smith and Wass (1980) document significant declines of 23 percent in height growth for lodgepole pine (*Pinus contorta* Dougl. ex Loud.) and 20 percent for Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) below insloped roads, which they attribute to loss of available water through redirected drainage flow. Improper fill placement and drainage can cause upslope groundwater to rise, and the changed soil moisture kills trees (Boelter and Close 1974, Stoeckeler 1965), although not commonly. Loss of nutrient capital is inevitable with soil disturbance from road building (Swanson and others 1989), but isolating this effect from other site changes has proved difficult. An indirect indication of nutrient loss is the marked growth response of plants on road fills after fertilizer is applied. Fertilizer applied to a granitic road fill in Idaho increased growth of vegetation by 32 to 116 percent (Megahan 1974a), but such increases are not documented after fertilizer is applied on undisturbed soils. Both surface and mass-erosion rates increase after road building, and often roads accelerate erosion on the slope below. Downslope damage generally is associated with mass erosion when a landslide originates from a road and causes scour on lower slopes or gullies related to concentrated road drainage (Megahan 1988a). This problem is widespread on steep slopes of the Pacific States and in the northern Rocky Mountains (Burroughs 1985, Swanson and others 1981), although Megahan (1988b) estimates that productivity is reduced on about 0.3 percent of forested land at a broad scale. These effects may range from decades (Ice 1985) to more than 85 years (Smith and others 1986). Road treads are highly compacted compared to natural soils, but compaction is not a productivity issue so long as roads are open and the running surface is bare. Road decommissioning must take compaction into account in restoring productivity, and various “ripping” treatments are routinely applied to decompact road surfaces.

Reliability, confidence, and generalizability—Direct effects of roads—including lost productivity because of the area occupied by roads themselves, and diminished productivity on cut slopes and road treads on closed roads—are well documented and general in geographic extent. Losses of productivity associated with road-caused, accelerated erosion are site specific and variable in extent, but they are commonly reported for all steep-slope landscapes. Rates of reforestation along road fills are high in the Pacific Northwest and Eastern United States and slower in the inland West and Southwest. Road-caused nutrient imbalances or declines often are confounded by other effects (notably soil moisture losses) in Western States.

Conclusions—A substantial amount of information is available on productivity in road fills and cut slopes and strong anecdotal, but obvious, evidence of lack of productivity on road treads. Information on effects of roads on adjacent site productivity is limited, and variable results confound attempts to generalize and accurately predict effects.

Secondary links—Applying salt to roads is discussed in “Water Quality” and its effects on plant damage are discussed in “Forest Diseases,” both below. Erosional processes and rates are discussed extensively in “Geomorphologic Effects,” above. Loss of site productivity represents a long-term economic loss, and quantifying such losses is confounded by the difficulty in establishing or even estimating the degree of soil productivity changes associated with roads.

Habitat Fragmentation

Issues—Natural populations of animal species are reduced by habitat loss caused by road building and by the animals' avoidance of areas near roads. Populations can be fragmented into smaller subpopulations, thereby causing increased demographic fluctuation, inbreeding, loss of genetic variability, and local population extinctions.

Findings—Habitat loss has broader effects than just the conversion of a small area of land to road surface. Roads fragment by changing landscape structure and by directly and indirectly affecting species. Habitat effects of roads on the landscape include dissecting vegetation patches, increasing the edge-affected area and decreasing interior area, and increasing the uniformity of patch characteristics, such as shape and size (Reed and others 1996). Whenever forest roads are built, changes in habitat and modified animal behavior will lead to changes in wildlife populations (Lyon 1983). Road-avoidance behavior is characteristic of large mammals such as elk (*Cervus canadensis*), bighorn sheep (*Ovis canadensis*), grizzly (*Ursus arctos horribilis*), caribou (*Rangifer tarandus*), and wolf (*Canis lupus*). Avoidance distances of 300 to 600 feet are common for these species (Lyon 1985). Road usage by people and their vehicles has a significant role in determining road avoidance by animals. In a telemetry study of movement by black bear (*Ursus americanus*), bears almost never crossed interstate highways, and they crossed roads with little traffic more frequently than those with high traffic volumes (Brody and Pelton 1989). Bobcats (*Lynx rufus*) crossed paved roads in Wisconsin forests less than expected, possibly to minimize interactions with vehicles and people (Lovallo and Anderson 1996). A few studies have related genetic changes in populations simply to the presence of roads (Forman and others 1997), but the distribution of roads in the environment also must be considered. Road density is a useful index of the effect of roads on wildlife populations (Forman and others 1997). Wolves in Wisconsin are limited to places with pack-area mean road densities of 0.7 mile/square mile or less (Mladenoff and others 1995). Some studies have shown that a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates (Rudis 1995).

Reliability, confidence, and limitations—The evidence is strong that forest roads displace some large mammals and certain birds such as spotted owls (*Strix occidentalis)* and marbled murrelets (*Brachyramphus marmoratus*) and that displaced animals may suffer habitat loss as a result. Effects of roads on small mammals and songbirds are generally described as less severe, with changes expressed as modifications of habitat that cannot readily be classified as detrimental or beneficial. This interpretation is also probably true for amphibians and reptiles.

Generalizability—For large mammals, general principles have been explained, above, that can be applied to project decisions.

Secondary links—Habitat fragmentation is linked to other habitat-related topics and also links with access-related topics, particularly timber, where the density and distribution of roads is a key technical and economic question.

Conclusions—Specific issues related to wildlife can be addressed directly. Integration with other technical, economic, and social issues (such as timber availability and recreational access) have to be dealt with by management.

Habitat

Issues—Road building introduces new edge habitat in the forest. The continuity of the road system also creates a corridor by which edge-dwelling species of birds and animals can penetrate the previously closed environment of continuous forest cover. Species diversity can increase, and increased habitat for edge-dwelling species can be created.

Findings—Roads and their adjacent environment qualify as a distinct habitat and have various species, population, and landscape-scale effects (Baker and Knight 2000, Dawson 1991, van der Zande and others 1980). Some research has attempted to describe habitat modifications caused specifically by roads, but most of this work is species and site specific (Lyon 1983). Surveys of songbirds in two national forests of northern Minnesota found 24 species of birds more abundant along roads than away from them (Hanowski and Niemi 1995). Close to half these species were associated with edges, including birds like crows (*Corvus brachyrhynchos)* and blue jays (*Cyanocitta cristata)* that use roads as corridors to find food. Turkey hens (*Megapodiidae*) in North Carolina nested near closed and gated logging roads and used them extensively in all stages of brood development (Davis 1992). One study showed that habitat in the roadside right-of-way supports a greater diversity of small mammals than do adjacent habitats (Adams and Geis 1983), but this finding may not apply to forest roads with only narrow cuts and fills on either side. The similarity between forest roads and transmission-line rights-of-way may be important in assessing the contribution of roads to habitat. Studies have shown that wide transmission-line corridors support grassland bird communities of species not found in the forest, and narrow corridors produce the least change from forest bird communities (Anderson and others 1977). The same study notes that increasing edge diversity of birds, for instance, may negatively affect abundance of interior species (see “Biological Invasions,” below).

Reliability, confidence, and limitations—Limited species and site-specific data exist describing the immigration of particular species into habitat created by roads. Detailed information on specific habitat characteristics affected by the building and presence of roads is lacking. The relation of microclimate, vegetation distribution, and water supply to the road network needs to be described.

Generalizability—In general, road building fragments habitat and creates habitat edge, thereby modifying the habitat in favor of species that use edges. Edge-dwelling species generally are not threatened, however, because the human-dominated environment has provided ample habitat for them. Any habitat modifications attributed to the road may be insignificant compared to the effects of the activity, such as timber harvest, for which the road was built.

Secondary links—Links exist to other habitat-related topics and also to biological invasions.

Conclusions—Science information about the underlying principles related to this issue is incomplete. Further study is needed before anything more than site- and species-specific analyses can be undertaken.

Biological Invasions

Issues—A widely cited generalization about biological invasion is that it is promoted by disturbance. Building roads and subsequently maintaining them (including ditch clearing, road grading, and vegetation clearing) in the interior of a forest represents disturbances that create and maintain new edge habitat. These roadside habitats can be invaded by an array of exotic (non-native) plant species, which may be dispersed by “natural” agents such as wind and water as well as by vehicles and other agents related to human activity. Roads may be the first point of entry for exotic species into a new landscape, and the road can serve as a corridor along which plants move farther into the landscape (Greenberg and others 1997, Lonsdale and Lane 1994). Some exotic plants may then be able to move away from the roadside into adjacent patches of suitable habitat. Invasion by exotic plants may have significant biological and ecological effects if the species are able to disrupt the structure or function of an ecosystem. Invasion also may be of concern to land managers, if the exotic species disrupt management goals and present costly eradication problems.

Findings—Although few habitats are immune to at least some invasion by exotic plants, predicting which species will become pests usually is difficult. Assessing the scale of a biological invasion problem is complicated by the lag between when an exotic is introduced and when it begins to expand its distribution and population size in a new area. Cowbirds *(Molothrus ater)*, for example, can be introduced into forested environments by roads and subsequently affect populations of Neotropical migratory birds through nest parasitism. The spread of pathogens where roads act as vectors is described in "Forest Diseases," below. Few environmentally benign approaches to exotic plant control or eradication have been tested.

Reliability, confidence, and limitations—Field studies of exotic plants tend to focus on a particular geographic region, and observed patterns of road-supported invasion may not apply to other regions. In general, however, observations suggest that biological invasion is often a negative effect of extending roads into forest interiors. Such effects should be considered in the design and execution of road network extensions.

Generalizability—Observations in different settings suggest that the exotic species that successfully invade and the scale of invasion problems differ regionally. Some exotic species can become significant pests, and others remain fairly benign.

Secondary links—Consequences of biological invasions link to habitat quality issues (including changes in plant community structure and function), other edge effects, and effects on sensitive or threatened species.

Conclusions—Information to assess the degree of risk relies on case studies; the risks may be slight or significant. A less than ideal science base exists for identifying which exotic species pose the greatest threat and what preventive or remedial measures are appropriate. Retrospective studies may help identify directions. One study showed that abandoned roads had fewer exotics (both in number of species and frequency of individuals) than did roads that were in use.

Indirect and Landscape-Scale Effects

Aquatic Habitat

Issues—The effects of roads on aquatic habitat are believed to be widespread and profound, and evidence is documented through empirical associations and direct mechanistic effects, although the mechanistic effects become fuzzy when direct, quantitative, cause-effect links are sought. Several studies correlate road density or indices of roads to fish density or measures of fish diversity. Mechanisms include effects of fine sediment, changes in streamflow, changes in water temperature caused by loss of shade cover or conversion of groundwater to surface water, migration barriers, vectors of disease, exotic fishes, changes in channel configuration from encroachment, and increased fishing pressure. A growing body of work indicates that the complexity of habitat and the predictability of disturbance influences species diversity. At the landscape scale, correlative evidence suggests that roads are likely to influence the frequency, timing, and magnitude of disturbance, which are likely to influence community structure.

Findings—Increased fine-sediment composition in stream gravel has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes. Increased fine sediment can reduce benthic organism populations and algal production. Increased sediment production associated with roads is discussed in detail in “Geomorphic Effects,” above. Survival of incubating salmonids from embryos to emergent fry has been negatively related to the proportion of fine sediment in spawning gravels (Chapman 1988, Everest and others 1987, Scrivener and Brownlee 1989, Weaver and Fraley 1993, Young and others 1991). Increased fine sediment in stream gravel can reduce intragravel water exchange, thereby reducing oxygen concentrations, increasing metabolic waste concentrations, and restricting movements of alevins (Bjornn and Reiser 1991, Coble 1961, Cordone and Kelley 1960). Survival of embryos relates positively to dissolved oxygen and apparent velocity of intragravel water, and positively to gravel permeability and gravel size (Chapman 1988, Everest and others 1987). Consequently, juvenile salmonid densities decline as fine sediment concentrations increase in rearing areas (Alexander and Hansen 1986, Bjornn and others 1977, Chapman and McLeod 1987, Everest and others 1987, Shepard and others 1984). Increases in fine sediment also can reduce winter carrying capacity of streams by loss of concealment cover (Bjornn and others 1977, Chapman and McLeod 1987, Thurow 1997) and by increasing the likelihood of predation (Chapman and McLeod 1987). Pools function as resting habitats for migrating adults, rearing habitats for juveniles (Bjornn and Reiser 1991), and refugia from natural disturbances (Sedell and others 1990). Pools that lose volume from sediment (Jackson and Beschta 1984, Lisle 1982) support fewer fish (Bjornn and others 1977), and fish that reside in them may suffer higher mortality (Alexander and Hansen 1986). Similarly, populations of tailed frogs can be severely reduced or eliminated by increased sedimentation (Corn and Bury 1989, Welsh 1990), presumably because of their dependence on unembedded interstitial areas in the stream substrate where they hide and overwinter (Brown 1990, Daugherty and Sheldon 1982). Increased sediment reduces populations of benthic organisms by reducing interstitial spaces and flow used by many species and by reducing algal production, the primary food source of many invertebrates (Chutter 1969, Hynes 1970).

The effects of roads are not limited to those associated with increases in fine-sediment delivery to streams; they can include barriers to migration, water temperature changes, and alterations to streamflow regimes. Improper culvert placement at road-stream crossings can reduce or eliminate fish passage (Belford and Gould 1989), and road crossings are a common migration barrier to fish (Clancy and Reichmuth 1990, Evans and Johnston 1980, Furniss and others 1991). In a large river basin in Washington, 13 percent of the historical coho habitat was lost as a result of improper culvert barriers (Beechie and others 1994). Roads built adjacent to stream channels pose additional effects. Changes in temperature and light regime from removing the riparian canopy can have both positive and negative effects on fish populations. Sometimes increased food availability can mitigate negative effects of increased summer water temperatures (Bisson and others 1988). Beschta and others (1987) and Hicks and others (1991) document negative effects, including elevation of stream temperatures beyond the range of preferred rearing, inhibition of upstream migrations, increased disease susceptibility, reduced metabolic efficiency, and shifts in species assemblages. Streamflow stability and predictability (size, timing, duration, and frequency) also strongly influence salmonid densities by influencing reproductive success and overwintering survival (McFadden 1969). For example, high flows after spawning can wash out eggs or displace fry, thereby increasing mortality (Latta 1962, Mortensen 1977, Shetter 1961).

The effect of roads on peak flows is relatively modest (see “Hydrologic Effects,” above), and the issues of changing stability and predictability because of roads may be of little importance to aquatic habitat suitability.

Road-stream crossings have effects on stream invertebrates. Hawkins and others (in press) found that the aquatic invertebrate species assemblages (observed versus expected, based on reference sites) were related to the number of stream crossings above a site. Total taxa richness of aquatic insect larvae (mayflies, Ephmeroptera; stoneflies, Plecoptera; and caddisflies, Trichoptera) were negatively related to the number of stream crossings. Another study (Newbold and others 1980) found significant differences between macroinvertebrate assemblages above and below road-stream crossings.

Several studies at broad scales document aquatic habitat or fish density changes associated with road density or indices of road density. Eaglin and Hubert (1993) show a positive correlation with numbers of culverts and stream crossings and amount of fine sediment in stream channels, and a negative correlation with fish density and numbers of culverts in the Medicine Bow National Forest. Macroinvertebrate diversity negatively correlates with an index of road density (McGurk and Fong 1995). Increasing road densities are associated with decreased likelihood of spawning and rearing of non-anadromous salmonids in the upper Columbia River basin, and populations are negatively correlated with road density (Lee and others 1997).

Reliability, confidence, and limitations—Research evidence of increased erosion and sediment delivery to streams resulting from roads is strong. Subsequent habitat changes from such processes as pool filling and cobble embeddedness are well documented, but these effects depend heavily on channel geometry, flow regimes, and so on. Thus, they range widely in time and space. Measured changes in stream temperature after canopy removal are strong but biological response is highly variable, and existing literature speculates on possible mechanisms. Empirical evidence relating road density to habitat and population response at landscape scales is fairly new. The study by Lee and others (1997) has a large database and is analytically sound, but it demonstrates a statistically valid population response only for non-anadromous salmonids. Because roads are not distributed randomly on the landscape, these studies can be confounded by other landscape variables that may control biological response. This issue is addressed by Lee and others (1997).

Generalizability—Broad-scale patterns in the distribution of roads and fish suggests that the effects of roads are common and widespread across a range of environments and conditions (Bettinger and others 1998, Lee and others 1997). Changes in aquatic habitat resulting from increased erosion and sediment delivery are highly controlled by lithology and slope, however. Road-derived sediment in granitic terrain typically results in an increase in the proportion of fine bedload. In fine-textured parent materials, suspended load may increase but not change pool filling and cobble embeddedness. Changed timing and size of peak and low flows resulting from roads have different implications for storm-generated and snowmelt-dominated hydrologic regimes, and they result in different biological effects for oversummer and overwinter egg survival. The effect of cover removal on elevated stream temperature depends on the rate of vegetation recovery and appears to be brief in the Eastern United States (Swift 1983).

Secondary links—Responses by aquatic habitat depend on geomorphic and sediment changes associated with roads. Road-associated changes in nutrients and hazardous chemical spills are also linked but are issues addressed elsewhere in this report.

Conclusions—Road effects on aquatic habitat and population response are well documented and overwhelmingly negative, but results differ among sites. Measures of the cumulative effects of roads that are closely related to mechanism (for example, the length of roads connected by direct surface-flow paths to streams or the miles of potential habitat blocked by culverts) would be more likely to produce stronger relations between roads and aquatic habitat elements than would road density.

Landscape-Scale Effects on Fish

Issues—The decline of anadromous fish in many parts of the country, especially the salmonids in the West, has led to much research on the diverse causes. Among those, the relation of roads to intensity of land use and adverse effects on aquatic habitats has been discussed in several recent studies and publications (Meehan 1991, Naiman and others 1992, Spence and others 1996). The discussion centers on three themes: the correlation of road density to fish habitat and fish populations is not strong; the legacy of past road building is so vast and budgets for maintaining roads so low that the problems will be with us for a long time; and road building practices have improved in the last decade to the point where we need not worry about the effects of roads on aquatic systems. The scientific assessment for the interior Columbia basin provided an opportunity to examine these issues at a broad, landscape scale in this ecoregion.

Findings—Roads contribute more sediment to streams than does any other land management activity (Gibbons and Salo 1973, Meehan 1991), but most land management activities, such as mining, timber harvest, grazing, recreation, and water diversions, depend on roads. Most of the sediment from timber harvest activities is related to roads and road building (Chamberlain and others 1991, Dunne and Leopold 1978, Furniss and others 1991, MacDonald and Ritland 1989, Megahan and others 1978) and the associated increases in erosion rates (Beschta 1978, Gardner 1979, Meehan 1991, Rhodes and others 1994, Reid 1993, Reid and Dunne 1984, Swanson and Dyrness 1975, Swanston and Swanson 1976). Serious degradation of fish habitat can result from poorly planned, designed, located, built, or maintained roads (Furniss and others 1991, MacDonald and others 1991, Rhodes and others 1994). Roads also can affect water quality through applied road chemicals and toxic spills (Furniss and others 1991, Rhodes and others 1994), and the likelihood of toxic spills reaching streams has increased with the many roads paralleling them.

Roads directly affect natural sediment and hydrologic regimes by altering streamflow, sediment loading, sediment transport and deposition, channel morphology, channel stability, substrate composition, stream temperatures, water quality, and riparian conditions in a watershed. For example, interruption of hillslope drainage patterns alters the timing and magnitude of peak flows and changes base stream discharge (Furniss and others 1991, Harr and others 1975) and subsurface flows (Furniss and others 1991, Megahan 1972). Road-related mass soil movements can continue for decades after roads have been built (Furniss and others 1991). Such habitat alterations can adversely affect all life stages of fish, including migration, spawning, incubation, emergence, and rearing (Furniss and others 1991, Henjum and others 1994, MacDonald and others 1991, Rhodes and others 1994).

Poor road location, concentration of surface and subsurface water by cross-slope roads, inadequate road maintenance, undersized culverts, and sidecast materials all can lead to road-related mass movements (Lyons and Beschta 1983, Swanston 1971, Swanston and Swanson 1976, Wolfe 1982). Sediment production from logging roads in the Idaho batholith was 770 times higher than in undisturbed areas; about 71 percent of the increased sediment production was due to mass erosion (Megahan and Kidd (1972), leaving 29 percent due to surface erosion.

In granitic land types, sedimentation is directly proportional to the road distance (Jensen and Finn 1966). For instance, 91 percent (66,000 cubic yards) of the annual sediment production by land-use activities (72,200 cubic yards) in the South Fork of the Salmon River (Idaho) is attributed to roads and skid trails (Arnold and Lundeen 1968). King (1993) determined that roads in the Idaho batholith increase surface erosion by 220 times the natural rates per unit area. Roaded and logged watersheds in the South Fork of the Salmon River drainage also have significantly higher channel-bed substrate-embeddedness ratings than do undeveloped watersheds (Burns 1984).

Roads greatly increase the frequency of landslides, debris flow, and other mass movements (Dunne and Leopold 1978, Furniss and others 1991, Megahan and others (1992). Mass movement along the west side of the Cascade Range in Oregon was 30 to 300 times greater in roaded than in unroaded watersheds (Sidle and others 1985). Megahan and others (1992) found that 88 percent of landslides in Idaho are associated with roads. Roads were the primary factor in accelerated mass movement activity in the Zena Creek drainage (Idaho batholith) after the 1964-65 winter storms (Gonsior and Gardner 1971). Of 89 landslides examined along the South Fork of the Salmon River, 77 percent originated on road hillslopes (Jensen and Cole 1965). Cederholm and others (1981) found increases (above natural rates) in the percentage of fine sediment in fish spawning habitat when road density exceeded 2.5 percent of the Clearwater River watershed in Washington. Increased stream-channel sedimentation in Oregon and Washington watersheds east of the Cascade Range also is associated with road density (Anderson and others 1992).

Road-stream crossings can be a major source of sediment to streams and result from channel fill around culverts and subsequent road-crossing failures (Furniss and others 1991). Plugged culverts and fill-slope failures are frequent and often lead to catastrophic increases in stream channel sediment, especially on abandoned or unmaintained roads (Weaver and others 1995). Unnatural channel widths, slope, and streambed form are found upstream and downstream from stream crossings (Heede 1980), and these alterations in channel morphology may persist for long periods. Channelized stream sections resulting from riprapping roads adjacent to stream channels are directly affected by sediment from side casting, snow removal, and road grading; such activities can trigger fill-slope erosion and failures. Because improper culverts can reduce or eliminate fish passage (Belford and Gould 1989), road crossings are a common migration barrier for fish (Clancy and Reichmuth 1990, Evans and Johnston 1980, Furniss and others 1991).

Key aspects of aquatic habitat are pools and instream wood (positive attributes) and fine sediment (negative attribute). From an analysis of stream-inventory data for the Columbia River basin (Lee and others 1997), pools declined with increasing road density and were highest in wilderness areas. Relations between wood and surface fines were less clear. In Oregon and Washington, where wood frequency was measured, it was higher for Forest Service lands managed as wilderness or in areas with moderate use; it was significantly related to road density in the northern Cascades, southern Cascades, Columbia Plateau, northern glaciated mountains, and Blue Mountains but not in the Upper Klamath. Only the Lower Clark Fork and central Idaho mountains had sufficient data to model the relation of wood frequency to surface fines. In these latter two areas, the relation with road density was not significant, although the highest mean values of five sediments were associated with the highest road-density class.

Analysis of fish distribution and status data for seven species of anadromous and resident salmonids in the Columbia basin showed that the frequency of strong populations generally declined with increasing road densities. Additional analyses of road effects focused on four non-anadromous species, because effects of roads and other land uses on anadromous species may be masked by migrational and ocean-related factors (for example, dam passage, predation, harvest). Three species showed significant road effects when either occupied spawning and rearing areas were distinguished from unoccupied areas or strong status was differentiated from depressed status. The analysis suggested a decreasing likelihood of occupancy, or a decreasing likelihood of strong status if occupied, with increasing road density. No other variables except ground-slope showed the consistent patterns across all species shown by the road-density measures.

The investigation of the influence of roads on population status clearly showed an increasing absence and a decreasing proportion of strong populations with increasing road density for several subgroups of fish. Additional evidence suggested that the lowest mean road-density values (number of road miles per unit area) are always associated with strong population status.

This trend is apparent for Yellowstone cutthroat trout *(Onchorynchus clarki bouvieri)*, even though it was the only subgroup that did not show a significant road effect in a logistic regression analysis. The lack of statistical significance in the face of apparent trends, however, points to complex interactions among the explanatory variables that are not adequately addressed in the relatively simple logistic model. Consistent, significant effects for other species may be further testament to the presence and pervasiveness of the effects. Strong relations between roads and the distribution and status of these species were detected despite the potential confounding effects of other variables (such as harvest, non-native introductions, and other habitat factors).

These results show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. These species are less likely to use highly roaded areas for spawning and rearing and, if found, are less likely to have strong populations. This consistent pattern is based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, limited primarily to forested lands administered by the Forest Service and the Bureau of Land Management. The relation would not be expected to be as strong on the con-forested, lower gradient lands administered by the bureau. Of the four species examined, the redband trout is the only one supported by the low-gradient lands. Only in forested, high-elevation areas could redband trout status be clearly associated with road-density changes.

Most aquatic conservation strategies acknowledge the need to identify the best habitats and most robust populations to use as focal points from which populations can expand, adjacent habitat can be usefully rehabilitated, or the last refugia of a species can be conserved in unroaded areas where biophysical processes are still operating without effects from many human disturbances. These refugia also provide necessary experimental controls for evaluating the effects of land management activities in other areas. The ecological importance of unroaded areas has been highlighted in the Columbia basin assessment as well as other reports (FEMAT 1993, Henjum and others 1994).

The overlap of unroaded areas—both within and outside designated wilderness areas—with stronghold watersheds for fish and with important conservation watershed efforts in the Columbia basin also was examined. Designated wilderness and unroaded areas are important anchors for strongholds throughout the basin. Unroaded areas occupy 41 percent of the area with known and predicted strongholds in the east-side environmental impact statement area. One-third of this area is outside designated wilderness. Of the known and predicted strongholds in the upper Columbia basin area, 68 percent are unroaded, of which 37 percent are outside wilderness.

Aquatic integrity in the Columbia basin was analyzed in relation to road densities and integrity ratings for other resources (forest, range, hydrology). Forest clusters with the highest integrity ratings were associated with low road densities; low integrity ratings corresponded with moderate or higher road densities. For example, the range cluster with the highest aquatic and composite integrity also had mostly low road densities. But the relations between road densities and integrity ratings for other range clusters were more variable.

The legacy of road building in the Pacific Northwest is enormous. The FEMAT report (1993) notes that federally managed forest lands in the range of the northern spotted owl contain about 180 000 kilometers (111,600 miles) of roads. A major portion of this road system may constitute a potential threat to riparian and aquatic habitats through sedimentation. An estimated 250,000 stream crossings (about 1.3 per kilometer [2.3 per mile]) are associated with these roads, and a significant number of culverts are thought to be unable to withstand storms with a recurrence interval greater than 25 years (FEMAT 1993), a hypothesis tested and affirmed by the February 1996 flood. Analysis suggests more than 205 000 kilometers (127,000 miles) of roads are on Forest Service and Bureau of Land Management lands in the Columbia River basin. Many stream crossings exist, with high densities of crossings in steep, highly dissected terrain and low densities in drier and flatter terrains. Many of the culverts or stream crossings are expected to perform poorly in flood events with recurrence intervals of more than 25 years, similar to their west-side counterparts identified in the FEMAT report. Even with adequate culvert size, lack of maintenance of a road network of this size could lead to significant road-drainage problems and accompanying effects on aquatic habitat.

Budgetary constraints on land management agencies may lead to lack of maintenance, resulting in progressive degradation of road-drainage structures and functions, increased erosion rates, and the likelihood of increased erosion (Furniss and others 1991). Problems are greatest with older roads in sensitive terrain and roads functionally abandoned but not adequately configured for long-term drainage. Applying erosion prevention and control treatments to high-risk roads can drastically reduce risks for future habitat damage and can be both effective and cost-effective. In watersheds that contain high-quality habitat and have only limited road networks, large amounts of habitat can be secured with small expenditures to apply storm proofing and decommissioning activities to roads (Harr and Nichols 1993).

For federal forests with moderate to high road densities, the job of maintaining roads may be expensive because many road networks have not been inventoried to determine their influence on riparian or aquatic resource goals and objectives. Substantial increases in sedimentation are unavoidable even when the most cautious road-building methods are used (McCashion and Rice 1983, Megahan 1980). Improving road-building and logging methods, however, can reduce erosion rates and sediment delivery to streams. The amount of sedimentation or hydrologic alteration from roads that aquatic species can tolerate before a negative response appears is not well known, though general effects of sediments on fishes are known. Sediment exceeding natural background loads can fill pools, silt spawning gravels, decrease channel stability, modify channel morphology, and reduce survival of emerging salmon fry (Burton and others 1993, Everest and others 1987, MacDonald and others 1991, Meehan 1991, Rhodes and others 1994).

Rice (1992) documents an 80-percent reduction in mass erosion from forest roads and about a 40-percent reduction in mass erosion from logged areas in northern California that resulted from improvements in forest practices beginning in the mid-1970s. Megahan and others (1992) used the BOISED sediment-yield production model to evaluate the effects of historical and alternative land management in an Idaho watershed (South Fork Salmon River). They report that current management practices, properly implemented, could reduce sediment yield by about 45 to 90 percent when compared with yields caused by the historical land use in their study watershed. If the improved road design currently practiced by the Boise National Forest is used, however, total accelerated sediment yields are still 51 percent more than natural ones. These improved road designs plus maximum erosion mitigation lead to 24-percent increases over natural yields in unroaded areas. Helicopter logging results in 3-percent increases over natural yields, and wildfire increases sediment yield about 12 percent over natural loads (Megahan and others 1992).

Megahan and others (1995) evaluated the effects of helicopter logging and prescribed burning on south-facing slopes of headwater drainages in the Idaho batholith by using paired watersheds monitored from 1966 to 1986. Average annual sediment yields show a statistically significant increase of 97 percent persisting for the 10 years of posttreatment study after logging and burning. Accelerated surface erosion primarily result from the prescribed burning, not the helicopter logging, because burning results in most of the bare-soil exposure and in connecting the affected area to streams. Surface erosion rates in the logged and burned areas are about 66 times greater than those on undisturbed slopes. The conclusion is that current best management practices can reduce sediment yields compared with historical practices. But the risk of increased sedimentation from forest management continues, particularly with such activities as road building, timber harvest, and prescribed burning.

Temporary roads may have fewer adverse effects than do permanent roads, depending on the extent to which they are decommissioned. As indicated by the analyses for the Columbia basin, distinguishing the direct effects of roads from the cumulative effects of other activities associated with roads is sometimes difficult. Thus, temporary roads may reduce the direct effects of roads, but effects of activities for which the temporary roads were built still will affect the environment.

Reliability, confidence, and limitations—The relations among roads, aquatic species and their habitats, and other variables analyzed for the Columbia basin were developed from predicted road density data developed from actual subsampled road data and a rule-based model. The method used in developing road density classes is not a substitute for actually mapping roads, but the rule-based model approach provides a tool for predicting road densities across a large landscape, when existing road data are incomplete or out of date. Also, the rule-based model assures that the method used in developing road densities is consistent throughout the Columbia basin. The final road density model had inherent uncertainties because of incomplete data layers, limitations of the sampling design, and the limitations of a rule-based model.

A few road types could not be predicted by using this rule-based approach, despite its general utility. For instance, Yellowstone National Park was assigned a road density class of *none* because no unique rule-based model combinations existed for predicting the park's road system. Roads inside the park are based on human recreational interests, which were not accounted for in the model.

Generalizability—Because the Columbia basin assessment was designed specifically as a broad-scale analysis, the relation of roads and aquatic species and their habitats can be applied at the large-landscape scale. Those relations may not be the same for federally managed lands outside the Pacific Northwest, particularly the Columbia basin, although aquatic habitat loss and alterations, which include effects of roads, are associated with the decline of many fish species throughout North America (Miller and others 1989). Those general relations also may differ at finer scales because of specific biophysical characteristics, such as geology and soils, and use of actual rather than predicted road densities.

The declines in population status of non-anadromous salmonids in the Columbia basin should be viewed as indicating the types of responses that may be experienced by other native aquatic species in similar habitats. The species most like the non-anadromous salmonids in distribution or habitat requirements would be expected to show the most similar responses. This group would include the anadromous species—such as steelhead, stream chinook salmon, and Pacific lamprey—that broadly overlap in range with the non-anadromous salmonids and use many of the same habitats for significant portions of their life. No logical reasons exist to expect anadromous fishes to be immune to the effects of habitat change evident in the non-anadromous species. The ranges of other species—including sculpins, dace, and some suckers—also overlap considerably, and these species may follow similar trends in population abundance and distribution.

Although unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds. Several possible reasons for this coexistence have been suggested: The inherent productivity of some areas allows fish populations to persist despite disturbances linked to roads; real or detectable effects on fish populations may lag behind the initial physical effects in watersheds where roads have been built in the last several years; and the scale of the subwatershed (19,800 acres on average) at which strong populations are identified may mask a potential disconnect between the real locations of strongholds and roads (which are identified at 1-square-kilometer [0.39-square-mile] pixels). This issue of scale can be resolved with a midscale or subwatershed analysis. The fact that strong salmonid populations can coexist in many roaded areas provides opportunities to determine the reasons, which may be instructive for both watershed restoration and future road building. Given current information, the assumption that because roads and strong fish populations coexist in some watersheds, they will in others is not prudent, however. In general, greater short- or long-term watershed and ecological risks are associated with entering an unroaded area than with proceeding cautiously with management activities in roaded areas to close and obliterate existing roads. The data strongly suggest a closer examination of the stronghold subwatersheds and their roaded condition.

Secondary links—The effects associated with roads reach beyond their direct contribution to disruption of hydrologic function and increased sediment delivery to streams. Roads provide access, and the activities that accompany access magnify the negative effects on aquatic systems beyond those caused solely by the roads themselves. Activities associated with roads include fishing, recreation, timber harvest, livestock grazing, and agriculture. Roads also provide avenues for stocking non-native fishes.

Unfortunately, inadequate broad-scale information on many of these attendant effects for the Columbia basin prevents identification of their component contributions. Similarly detailed analyses are needed to address the relations between roads and fish at a landscape scale in other ecoregions.

Conclusions—The range of specific case studies for broad-scale assessment of road relations in the Columbia basin provides a substantial base of information on which to evaluate the direct effects of roads and the cumulative effects of activities associated with roads on aquatic habitats and species in the Northwest.

Terrestrial Vertebrates

Issue—Effects of roads on vertebrate populations act along three lines: direct effects, such as habitat loss and fragmentation; road use effects, such as traffic causing vertebrate avoidance or road kill; and additional facilitation effects, such as overhunting or overtrapping, which can increase with road access.

Findings—In recent research in the interior Columbia River basin, Wisdom and others (2000) identify more than 65 species of terrestrial vertebrates negatively affected by many factors associated with roads. Specific factors include habitat loss and fragmentation, negative edge effects, reduced densities of snags and logs, overhunting, overtrapping, poaching, collection, disturbance, collisions, movement barriers, displacement or avoidance, and chronic, negative interactions with people. These factors and their effects on vertebrates in relation to roads are summarized from Wisdom and others (2000) as follows:

> Road construction converts large areas of habitat to nonhabitat (Forman 2000, Hann and others 1997, Reed and others 1996); the resulting motorized traffic facilitates the spread of exotic plants and animals, further reducing quality of habitat for native flora and fauna (Bennett 1991, Hann and others 1997). Roads also create habitat edge (Mader 1984, Reed and others 1996); increased edge changes habitat in favor of species that use edges, and to the detriment of species that avoid edges or experience increased mortality near or along edges (Marcot and others 1994).
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> Species dependent on large trees, snags, or logs, particularly cavity-using birds and mammals, are vulnerable to increased harvest of these structures along roads (Hann and others 1997). Motorized access facilitates firewood cutting, as well as commercial harvest, of these structures.
>
> Several large mammals are vulnerable to poaching, such as caribou, pronghorn antelope, mountain goat, bighorn sheep, wolf, and grizzly bear (Autenrieth 1978, Bruns, 1977, Chadwick 1973, Dood and others 1986, Greer 1985, Gullison and Hardner 1993, Horejsi 1989, Knight and others 1988, Lloyd and Fleck 1977, Luce and Cundy 1994, Mattson 1990, McLellan 1990, McLellan and Shackleton 1988, Mech 1970, Scott and Servheen 1985, Singer 1978, Thiel 1993, Van Ballenberghe and others 1975, Yoakum 1978). Roads facilitate this poaching (Cole and others 1997).
>
> Gray wolf and grizzly bear experience chronic, negative interactions with humans, and roads are a key facilitator of such interactions (Mace and others 1996, Mattson and others 1992, Thiel 1985). Repeated, negative interactions of these two species with humans increases mortality of both species and often causes high-quality habitats near roads to function as population sinks (Mattson and others 1996a, 1996b; Mech 1973).
>
> Carnivorous mammals such as marten (*Martes americana*), fisher (*M. pennanti*), lynx (*Lynx canadensis*), and wolverine (*Gulo luscus*) are vulnerable to overtrapping (Bailey and others 1986, Banci 1994, Coulter 1966, Fortin and Cantin 1994, Hodgman and others 1994,

Hornocker and Hash 1981, Jones 1991, Parker and others 1983, Thompson 1994, Witmer and others 1998), and overtrapping can be facilitated by road access (Bailey and others 1986, Hodgman and others 1994, Terra-Berns and others 1997, Witmer and others 1998). Movement and dispersal of some of these species also is believed to be inhibited by high rates of traffic on highways (Ruediger 1996), but this has not been validated. Carnivorous mammals such as lynx also are vulnerable to increased mortality from highway encounters with motorized vehicles (as summarized by Terra-Berns and others 1997).

Reptiles seek roads for thermal cooling and heating, and in doing so, these species experience significant, chronic mortality from motorized vehicles (Vestjens 1973). Highways and other roads with moderate to high rates of motorized traffic may function as population sinks for many species of reptiles, resulting in reduced population size and increased isolation of populations (Bennett 1991). In Australia, for example, 5 million reptiles and frogs are estimated to be killed annually by motorized vehicles on roads (Ehmann and Cogger 1985, as cited by Bennett 1991). Roads also facilitate human access into habitats for collecting and killing reptiles.

Many species are sensitive to harassment or human presence, which often are facilitated by road access; potential reductions in productivity, increases in energy expenditures, or displace-ments in population distribution or habitat use can occur (Bennett 1991, Mader 1984). Exam-ples of such road-associated effects are human disturbance of leks (sage grouse [*Centrocercus urophasianus*] and sharp-tailed grouse [*Tympanuchus phasianellus*]), nests (ferruginous hawk [*Buteo regalis*]), and dens (kit fox [*Vulpes macrotis*]). Another example is elk avoidance of large areas near roads open to traffic (Lyon 1983, Rowland and others 2000), with elk avoidance increasing with increasing rate of traffic (Wisdom and others 2000, Johnson and others 2000).

Bats are vulnerable to disturbance and displacement caused by human activities in caves, mines, and on rock faces (Hill and Smith 1984, Nagorsen and Brigham 1993). Cave or mine exploration and rock climbing are examples of recreation that could reduce population fitness of bats that roost in these sites (Nagorsen and Brigham 1993, Tuttle 1988). Such activities may be facilitated by human developments and road access (Hill and Smith 1984).

Ground squirrels often are targets of recreational shooting (plinking), which is facilitated by human developments and road access (Ingles 1965). Many species of ground squirrels are local endemics; these small, isolated populations may be especially vulnerable to recreational shooting and potentially severe reductions or local extirpations of populations.

Roads often restrict the movements of small mammals (Mader 1984, Merriam and others 1988, Swihart and Slade 1984), and consequently can function as barriers to population dispersal and movement by some species (Oxley and Fenton 1974).

Many granivorous birds are attracted to grains and seeds along roadsides and as a result have high mortality from collisions with vehicles (Vestjens 1973). And pine siskens (*Carduelis pinus*) and white-winged crossbills (*Loxia leucoptera)*, for example, are attracted to road salt, which can result in mortality from vehicle collisions (Ehrlich and others 1988).

Terrestrial vertebrates inhabiting areas near roads accumulate lead and other toxins that originate from motorized vehicles, with potentially lethal but largely undocumented effects (Bennett 1991).

In summary, no terrestrial vertebrate taxa seem immune to the myriad of road-associated factors that can degrade habitat or increase mortality. These multifaceted effects have strong management implications for landscapes characterized by moderate to high densities of roads. In such landscapes, habitats are likely underused by many species that are negatively affected by road-associated factors. Moderate or high densities of roads sometimes index areas that function as population sinks that otherwise would function as source environments were road density low or zero.

Reliability, confidence, and limitations—General effects of roads and road-associated factors on a wide variety of vertebrate taxa are well documented from a broad range of studies conducted in North America, Europe, and other areas (Bennett 1991, Forman and Alexander 1998, Mader 1984, Trombulak and Frissell 2000, Vestjens 1973). Reliability of such effects at large, landscape scales, and for many taxa, is compelling and unequivocal. Reliability of site-specific, small-scale effects, with focus on single species, is less certain. For many species at local scales, the array of factors that could affect habitats or populations have been neither well studied nor documented. Despite such limitations, current knowledge of broad-scale effects on a variety of taxa is highly certain and provides an overarching paradigm from which likely or presumed effects on single species at local scales can be inferred. The many factors associated with roads suggests that mitigating such effects succeeds best at large scales, when focused on multiple species, and when based on a combination of aggressive road obliteration and protection of roadless areas (Trombulak and Frissell 2000).

Generalizability—Although the summary of road-associated effects on vertebrates described here is taken from research conducted in the interior Columbia River basin (Wisdom and others 2000), results likely apply to several species occupying a diversity of forest and rangeland environments in North America. At least four reasons account for this presumed high generalizability: the road and road-associated effects described by Wisdom and others (2000) were synthesized from research conducted across the world; the synthesis focused on multiple species encompassing diverse taxa and environmental requirements; the synthesis addressed an extreme range of environmental conditions on federal lands administered by the Forest Service, the Bureau of Land Management, and state, private, and tribal landowners; and the synthesis focused on large-scale, overarching effects common to many species and conditions.

Secondary links—Many road-associated effects on terrestrial vertebrates are intimately linked to managing human activities related to road access. Accordingly, mitigation of road-use effects requires effective control of human access to roads related to managing livestock, timber, recreation, hunting, trapping, and mineral development.

Conclusions—Comprehensive mitigation of the full array of road-associated effects on terrestrial vertebrates of conservation concern poses one of the most serious of land management challenges. Balancing such mitigation with socioeconomic desires will be controversial and contentious. Comprehensive efforts to mitigate road-associated effects on terrestrial vertebrates is well suited to testing as a large-scale management experiment developed and implemented jointly by managers, researchers, and the public.

Road Kill

Issues—Large numbers of animals are killed annually on roads. In selected situations, such as for some amphibians with highly restricted home ranges, populations of rare animals may be reduced to dangerous sizes by road kills.

Findings—An estimated 1 million vertebrates a day are killed on roads in the United States (Lalo 1987). Studies show that the number of collisions between animals and vehicles is directly related to the position of the nearest resting and feeding sites (Carbaugh and others 1975). Because most forest roads are not designed for high-speed travel, and the speed of the traffic is directly related to the rate of mortality, direct mortality on forest roads is not usually an important consideration for large mammals

(Lyon 1985). An exception is forest carnivores, which are especially vulnerable to road mortality because they have large home ranges that often include road crossings (Baker and Knight 2000). Forest roads pose a greater hazard to small, slowly moving, migratory animals, such as amphibians, making them highly vulnerable as they cross even narrow forest roads (Langton 1989). Nearly all species of reptiles use roads for cooling and heating, so many of them are killed by vehicles. Highways and other roads with moderate- to high-speed traffic function as population sinks for many species of reptiles, resulting in reduced and increasingly isolated populations (Wisdom and others 2000). Predators and scavengers are killed while they feed on road-killed wildlife, as are other species attracted to roads because of salts or vegetation, or because roads facilitate winter travel (Baker and Knight 2000). Although countless animals are killed on roads every year, documented road-kill rates are significant in reducing populations of only a few rare species in North America, and these kills generally are on high-speed highways (Forman and others 1997).

Reliability, confidence, and limitations—A large body of data documents annual road kill, and wildlife science can describe the factors that put wildlife at risk, but little research has focused on how to mitigate the effects on wildlife populations.

Generalizability—Most road-kill questions will be related to individual species and geographic sites, but general principles such as the frequency of travel between known resting and feeding areas for individual species can be used in project decisions.

Secondary links—Road-kill issues link to habitat fragmentation, predation, and access issues.

Conclusions—The issues can be addressed based on site and species. Difficulty will arise in integrating road kill with the social and economic issues related to mitigation.

Forest Diseases

Issues—In general, the existence of roads seems to have little effect on forest tree diseases, but there are some examples where building or using roads caused significant local effects. Nearly always, the negative effects can be ameliorated through simple modifications in how they are built and used. The one benefit of roads, as it pertains to tree diseases, is to provide access for silvicultural activities that protect resources, such as the ability to inoculate decay fungi into trees to create wildlife habitat (Bull and others 1997). One negative effect includes the movement of people on the roads, which allows the pests to be introduced. Road building also may set the stage for an insect attack that further stresses the trees and then a disease outbreak that kills them (Boyce 1961).

Findings—A significant forest disease problem associated with roads is Port-Orford-cedar root disease. This disease of Port-Orford-cedar (*Chamaecyparis lawsoniana* (A. Murr.) Parl.) is a root disease caused by the fungus *Phytophthora lateralis*. Spores of the fungus are carried in water or contaminated soil to uninfected areas. Roads of any sort in the very limited geographic range of the primary host provide a way to move soil—along with the fungus—from infected to uninfected areas. Spread of the fungus can be checked by careful planning to reduce entry to uninfected areas, road closures, partial road closures during wet weather, attention to road surfaces and drainage of possibly contaminated water to streams, wash stations to remove soil from vehicles before entry to uninfected areas, and sanitation strips to remove host plants from near roadsides (Kliejunas 1994, Roth and others 1987, Zobel and others 1985). Building and maintaining roads may exacerbate root diseases. Wounded trees and conifer stumps created and not removed during road building provide infection courts for annosus root disease; the disease may then spread through root contacts to kill a patch of trees

(Otrosina and Scharpf 1989). Trees damaged or stressed by road building—through direct wounding of stems and roots, covering of roots with side castings, or compacting of soil over roots—become susceptible to various tree diseases. Armillaria root disease is benign in deciduous stands where only injured trees are attacked but more serious in conifer stands where pockets of disease are initiated (Shaw and Kile 1991). Oak decline is associated with poor sites, older stands, and road building or other disturbance (Wargo and others 1983). Black stain root disease (*Leptographium wagneri*) attacks stressed conifers associated with disturbance, especially compaction caused by road building; in pinyon pine (*Pinus monophylla*), it is associated with roads and campsites (Hansen 1978, Hansen and others 1988, Hessburg and others 1995). Droopy aspen disease is associated with road building and compaction, but the pathogen identity is unknown (Jacobi and others 1990, Livingston and others 1979). Sap streak disease in sugar maple is associated with compaction from roads and from direct injury to trees (Houston 1993).

Road building can be planned to help reduce the spread of some forest tree diseases: mistletoe is spread by the forcible ejection of the mistletoe seeds. In young plantations or pole-sized stands, roads can subdivide an area to prevent mistletoe seeds from reaching a healthy stand (Hawksworth and Wiens 1996). In Texas, roads could be planned to separate a portion of a stand with oak wilt from healthy trees. The act of building the road (if extensive enough) severs root connections and prevents tree-to-tree movement of the pathogen (Appel and others 1995, Rexrode and Brown 1983). In other areas, new or established roads may have the unintended effect of breaking the continuity of host roots and thus halting the spread of laminated root rot (*Phellinus weirii*) and other root diseases (Hadfield 1986, Thies and Sturrock 1995).

Roads indirectly contribute to disease spread by giving people access to remote forests and ways to transport material long distances. New pockets of both oak wilt and beech bark disease (Houston and O'Brien 1983) may have resulted from moving firewood from the forest to a homesite (Appel and others 1995, Rexrode and Brown 1983). Pitch canker (*Fusarium subglutinans*) was recently reported on Monterey pine (*Pinus radiata*) in California; previously, it had been found on little-leaf and slash pines in the South. A single introduction is thought to be responsible; 117 vegetative compatibility groups are found in Florida but only 5 in California, and 70 percent of the isolations in California are from a single group, likely carried on a tree transported as an ornamental (Correll and others 1992, Storer and others 1995). Campers who use roads to get to remote sites in Colorado and other states have caused significant mortality by carving on aspen and birch, which provides pathways for various fungi that cause cankers and quickly kill the trees. Many trees are unintentionally damaged, for example, when campers hang a gas lantern on a branch too close to the trunk of a tree, thereby causing heat damage.

One abiotic disease has caused significant damage. In the Lake Tahoe basin in California, trees were killed by salt put on the roads to reduce ice. This problem also has appeared in some areas of the Midwest and east coast (Kliejunas and others 1989, Scharpf 1993, Scharpf and Srago 1974). Needle and rust diseases spread long distances by spores and do not appear to be influenced by roads or road building.

Reliability, confidence, and limitations—Field studies tend to focus on a single disease or an insect-disease complex; many of these centers are associated with or influenced by compaction or tree damage associated with roads.

Generalizability—Problems, where they exist, appear to be specific to the pathogen, host, and site.

Conclusions—In general, land managers appear to have the information and technology needed to handle most road, road building, and disease interactions. Additional science-based information is needed to understand and manage the interactions between compaction and black stain root disease and between compaction and droopy aspen disease.

Predation

Issues—The introduction of roads into the closed forest environment creates corridors by which predators can enter and affect native populations.

Findings—Forest roads create corridors by which predators, especially people, can enter the forest environment and affect wildlife populations. Nest depredation of songbirds may increase by predators attracted to edges. Evidence for edge effects, however, is highly variable (Paton 1994). Although evidence has been found for local edge effects in cowbird parasitism and nest depredation, their effects on bird populations is not documented. Geographic location and large-scale patterns in the amount of forest and nonforest habitats may be more important in determining the reproductive success of forest songbirds (Donovan and others 1997, Robinson and others 1995). Forest carnivores apparently travel on roads in winter when snow is deep, and thus the road system alters and enhances their ability to move (Paquet and Callaghan 1996). Wolves and grizzly bears are two key species that have chronic, negative interactions with people, and roads are a key facilitator. Repeated, negative interactions of these two species with people increase mortality of both species and often cause high-quality habitats near roads to be population sinks (Wisdom and others 2000). High road densities are associated with a variety of negative human effects on several wildlife species (Brocke and others 1988). People directly affect snakes by collecting, harassing, and killing them (Wisdom and others 2000). Increases in illegal hunting pressure, facilitated by roads, also negatively affect populations. Moose, wolves, caribou, pronghorn antelop, mountain goat, and bighorn sheep are particularly vulnerable to this kind of predation (Lyon 1985, Wisdom and others 2000).

Reliability, confidence, and limitations—Limited data exist on the effects of introducing natural predators as a result of road building. The evidence is strong that human predation, either legally in game management programs or illegally, is greatly facilitated by roads and can significantly affect populations of animals.

Generalizability—General principles related to human effects on wildlife populations are understood by wildlife managers and can be applied to species and site-specific management.

Secondary links—Predation links to other habitat-related topics, such as fragmentation and road kill, and also to people-related topics such as recreation.

Conclusions—Species-specific issues related to predation facilitated by roads can be addressed for specific sites. Predation related to illegal hunting facilitated by improved access can be addressed by legal measures, or, where legal remedies are ineffective, by closing or decommissioning roads where wildlife values are high.

Biodiversity and Conservation

Issues—Previous issues in this section may be synthesized by the concept of biodiversity. Biodiversity is, in simplest terms, the variety of life and its processes (Keystone Center 1991). Recent syntheses (Heywood and Watson 1995) emphasize the reciprocal relation between biodiversity—conceived as genetic and species diversity—and ecosystem function. The many species comprising the biodiversity of an area play roles essential to ecosystem function and are the source of variation that enables an ecosystem to adapt to change. The healthy, functioning ecosystem, in turn, supports the many species living within it. Appreciating this reciprocity means that biodiversity can be taken as a natural measure of the ecosystem as a whole and thus can integrate the many concerns listed.

Some species may play more important roles than others in the normal functioning of an ecosystem. For example, keystone species may define the major structural elements of an ecosystem, as Douglas-fir does for forests in the Pacific Northwest, or they may—by virtue of their position in a complex trophic structure—act to maintain the diversity as keystone predators do for herbivores. On the other hand, the many species that do not appear to serve an important role in an ecosystem constitute a reservoir of potential adaptation to change. Because an ecosystem cannot predict change, the diversity of species acts as a hedge against it.

Biodiversity is vital to long-term ecosystem function, and human activities that decrease biodiversity can impair it. Our working hypothesis, then, is that measures of biodiversity provide the best integrative assessment of the effects of roads on ecosystems.

Findings—Roads can have major adverse effects on biodiversity, many of which are already described (Forman and Collinge 1996). A recent review by Forman and Hersperger (1996) usefully distinguishes these aspects of the road-biodiversity interaction:

- Road density: As road density increases, thresholds may be passed that cause some species to go locally extinct. The probability of extinction depends, in part, on body size, with larger animals requiring larger residual populations to prevent their extinction.
- Road-effect zone: The effects of roads can extend over some distance from their centers, such that their “effective widths” can be many times their actual widths.

Reliability, confidence, and limitation—The confidence in the general negative relation between roads and biodiversity is high. The current primary limitation, however, is on the utility of measures of biodiversity for assessing road effects. First, both the status of keystone and other important species must be assessed, which seems fairly straightforward. But, second, the status of the pool of all the other species that form the basis for adaptation to change must be assessed, and how to do this assessment is much less clear.

Landscape ecology as well as fragmentation and viability analysis contain relevant scientific uncertainties. Two critical uncertainties must be resolved to understand how roads affect fragmentation and population viability. First, in the mechanistic analysis of the effects of roads and roadlike entities, such as power lines, on landscape fragmentation and species viability, the question of the “effective width” of roads is open. Kiester and Slatkin (1974) predict that, for species using conspecific cuing for movement strategies and habitat selection (likely most vertebrates), a spatially localized source of mortality in an area of otherwise suitable habitat can act as an active sink, drawing individuals in as residents die, making it likely that the new individuals will die as well. Consider a road traversing the habitat of a territorial or conspecific-cuing species. Those individuals whose home range overlaps a road have some probability of being hit each time they venture across it. Eventually they are killed, and their neighbors, in the process of constantly testing the boundaries of their home ranges, move into the vacated area next to the road and themselves run the risk of road mortality. The question is, How far from a road does this probability of mortality spread? Second, at the landscape scale, the relation between patterns of dispersal of individual species and measurements of fragmentation must be clarified. Current information (Schumaker 1996) indicates that most of the commonly used measures of fragmentation do not predict habitat connectivity for individual endangered species; rather, a model of fragmentation must be derived from species-specific dispersal characteristics. This kind of analysis is now available for only a few species.

Generalizability—Exactly how roads affect biodiversity in any particular place is a matter of the devil being in the details. The results given here would generally apply to any area.

Secondary links—Appreciation of biodiversity itself is an important part of the passive-use value of biodiversity. In particular, the aesthetic appreciation of biodiversity through an understanding of how biodiversity is sublime (rather than just beautiful) is now leading to a new link between biodiversity and passive-use value (Kiester 1997).

Conclusions—Forman and Hersperger (1996) conclude " ...that a quantum leap in focus on the ecological effects of roads is warranted, and that the foundations are in place for effective research, planning, public education, and action."

Water Quality

Issues—Roads provide access to and increase the opportunity for applying a variety of chemicals in national forests. Some applications target the roads, such as with road surface treatment; other chemicals are intended for adjacent ecosystems to control pests and fertilize vegetation. Materials also are added to roads by traffic, such as asbestos from brake linings, oil leakage, and accidental spills. Some portion of applied and spilled chemicals eventually reaches streams by drift, runoff, leaching, or adsorption on soil particles. Roads also increase the nutrient delivery to streams by removing vegetation, rerouting water flow paths, and increasing sediment delivery. And roads increase the likelihood of toxic spills associated with accidents along streamside corridors.

Findings—Chemicals applied on and adjacent to roads can enter streams by various pathways. The likelihood of water-quality deterioration from ground applications is a function of how much chemical is applied, the proximity of the road to a stream, and the rainfall, snowmelt, and wind events that drive chemical and sediment movement. The risk is a function of the likelihood of water-quality deterioration and exposure of organisms, including people, and how susceptible the organisms are to the pollutant or pollutants. (A large proportion of Forest Service roads are low standard and few if any chemicals are applied, so the risk of chemical contamination for most Forest Service roads is relatively low.) Chemicals are applied directly to roads and adjacent rights-of-way for various purposes, including dust abatement, stabilizing the road surface, deicing, fertilizing to stimulate plant growth on road cuts and fills, and controlling weeds and the invasion of nonweedy plants onto the roadway (Furniss and others 1991, Norris and others 1991, Rhodes and others 1994). Applied chemicals can enter streams directly when they are applied, but little is known about the effects of these chemicals on stream biota (Furniss and others 1991). Norris and others (1991) provide a comprehensive review of the types and amounts of fertilizers, pesticides, and fire retardants applied to forests in the United States, although little information is given to distinguish road-related from aerial applications. They report that most herbicides are applied by ground-based equipment, presumably using roads for access; that ground-based applications in or near aquatic zones can result in chemicals entering streams by drift or direct application; and that these problems are more serious when the chemicals are applied from the air. Movement of sediment containing adsorbed chemicals is possible, and the risk increases with increasing persistence (Norris and others 1991). The amount of input by this pathway is thought to be small, however; it is a more likely pathway for entry of salts applied for de-icing and of fertilizers applied to road fills.

Increased nutrient supply to streams from roads is proportional to the area disturbed and maintained free of vegetation and the amount of sediment delivered. Increased nutrients rarely have detrimental effects on stream water quality, but they may modify the composition of aquatic biota (Hawkins and others, in press). Few studies examining watershed responses to logging separate the effect of road building from those of the broader disturbance associated with removing timber. In one such study, Swank (1988) monitored stream chemical composition during the pretreatment, road building, logging, and posttreatment phases in a cable-logged watershed in the southern Appalachian Mountains. No stream chemical response was found to result from the road-building phase of the watershed treatment. Nutrient movement to streams often increases significantly after timber harvest operations (Frederiksen and others 1973, Hornbeck and others 1973, Likens and others 1970, Pierce and others 1972, Swank and Waide 1988). The primary intent of these studies was to assess onsite nutrient losses, with changes in water quality a secondary concern. All cited studies report increases in nitrogen cation and phosphorus concentrations in streams after treatment. In general, nutrient loss to streams is roughly proportional to how much vegetation was removed. For example, three studies at Hubbard Brook in New Hampshire compared three treatments: clearcutting with a herbicide treatment to suppress vegetation regrowth (Likens and others 1970), clearcutting without suppressing regrowth (Pierce and others 1972), and strip cutting of one-third of the forest (Hornbeck and others 1973); the three studies found nitrogen concentrations in streams reduced, most by the first treatment, less by the second, and least by the third. These findings suggest that residual or reestablished vegetation immobilizes released nutrients, thus diminishing the disturbance effect. Although roads might not respond in the same way because of drainage rerouting, we expect that nutrient mobility is proportional to the area maintained in a disturbed, non-revegetated state.

Hazardous chemical spills from vehicle accidents can pose a direct, acute threat of contamination to streams. The risk of hazardous chemical spills resulting from vehicle accidents adjacent to waterways is recognized and documented by the National Forest System and by state transportation departments (IDT 1996). Risk-analysis models of accident-related chemical spills are available, but they are designed for paved roads in nonmountainous terrain. Models take into account risk to human health, traffic frequency, vehicle type, and proximity to water. Possible contaminants include any substance being transported, such as fuel, pesticides, chemicals used in mining, fertilizers, and fire retardants.

Reliability, confidence, and limitations—Both anecdotal and scientific bases for linking increased access provided by roads to increased use of a wide variety of introduced chemicals are strong. Potential delivery to streams is mainly anecdotal, and few models are available for predicting delivery. Evidence for increased nutrient delivery to streams from disturbance by roads is strong, but it is confounded by other management activities such as logging.

Generalizability—The use of chemicals that are potential contaminants is well known and often described. The likelihood of routinely or accidentally spilled chemicals is related to type and frequency of traffic, but determining probabilities of spills accurately is difficult or impossible, especially for accidents. The likelihood of contaminants reaching a stream differs widely from site to site; it is most strongly controlled by stream proximity and road drainage features. Soluble and persistent elements and compounds adsorbed on sediment particles have increased probability of contaminating waterways.

Secondary links—Roads have strong links to aquatic health and biological response. A large body of literature exists on bioassays, but little information is available on transport, toxicity, and persistence of potential contaminants in natural systems. Terrestrial effects of chemicals, such as damage to vegetation by road salt, are not addressed here.

Conclusions—Most of the information is anecdotal or requires extrapolation from other studies (nutrient issues). The degree to which aquatic organisms are affected by applied and routinely spilled chemicals is poorly known or not understood in most places. Better information on effects is needed to make decisions about chemical application, road drainage control, and road location. Better models of chemical spill risks on forested roads are needed.

Air Quality

Issues—Dust emitted into the atmosphere by vehicles moving on unpaved roads contributes to reducing visibility and to suspending airborne particulates that can pose health hazards. Issues revolve around the contribution of national forest roads to regional and urban air pollution and what effects maintaining, paving, and shutting down roads on national forests have on this problem. Roads built into or surfaced with serpentinitic rock may contain asbestos-type minerals that could pose a hazard to people exposed to dust from the road surface.

Findings—Scientific literature on this topic is scarce. A study of degraded visibility and its causes in 16 national parks and wilderness areas on the Colorado Plateau, by the Grand Canyon Visibility Transport Commission (available online at http://www.nmia.com/gcvtc/), found that dust from unpaved roads could be a contributing factor. Soils in the Southwest are often very fine textured, and once dust is made airborne by vehicles, it can remain suspended for a long time and be transported long distances by the wind. The commission recommended that the Environmental Protection Agency (EPA) require further study and mitigation of these effects.

The amount of dust emitted into the atmosphere is estimated by a formula that considers the number and speed of vehicles traveling on a road in a given period, the relative humidity, and the composition of the road surface. This model was developed and reviewed by the Department of Transportation and the EPA. Related information about calculations for paved roads can be found at http://www.epa.gov/ttn/chief/ap42/ch13/related/c13s02-1.html.

Dust emissions also raise issues of human health. Where national forests are close to urban areas, dust from national forest roads can contribute to the burden of airborne particulate matter from a wide variety of sources including transportation and industrial activities. The fine fraction of airborne particles with diameters less than 2 microns have been found to contribute to human health problems and increased mortality, especially in young children, old people, and people with lung problems such as asthma and emphysema. Particles of this size and smaller cannot be effectively cleared by human lungs and therefore accumulate. How much road dust from forest roads contributes to the fine particulates in urban atmospheres is not currently known for most cities because the EPA is just beginning wide-spread monitoring of fine particulates, and reliable results will take at least 3 years to gather.

Unpaved roads built into or surfaced with serpentine materials can generate dust containing asbestos or asbestiform minerals. Although few such roads exist, methods have been developed to determine the extent of ambient asbestos coming from them.

During commercial use of unsurfaced roads, watering or other dust-abatement treatment (such as the addition of lignin sulfonate or calcium chloride) is often required by the Forest Service or other road manager to reduce dust emissions and conserve the fine fraction of the road surface. Such treatments do not accompany noncommercial uses, however, and they include most of the traffic for such roads.

The EPA has proposed a regional haze rule calling for more regions to do the kind of analysis done by the Grand Canyon Commission. Such analyses are likely to find similar emissions from unpaved roads and similar visibility problems elsewhere. EPA’s recent tightening of the National Ambient Air Quality Standard on the effects of fine particles on human health are likely to require similar analyses of particle emissions, especially as they affect urban air quality. Analyzing the entire transportation system, including national forest roads, would be a logical approach to finding the most efficient means of controlling air pollution. Under emissions-trading scenarios, treatments, like paving or closure to reduce emissions of particles from national forest roads might qualify for highway funds, as cost-effective adjuncts to upgrading major arterials to reduce air pollution.

Reliability, confidence, and limitations—The basic models of dust emission and transport down-wind are generally reliable and widely used by the EPA in regulatory decisions. Much of the basic data to make these calculations for national forest roads have not been collected; thus, most estimates of the emissions are based on very coarse estimates of the conditions that produce dust emissions. Effects of the amount of road maintenance on emissions also are not well understood. The effects of road closures on dust emissions are not easily predicted because they depend on the details of how traffic is rerouted from closed sections and what emissions are created by the rerouted traffic pattern.

Generalizability—Models of emissions are relatively easy to generalize to many parts of the country, if reliable data are collected to use in them.

Secondary links—Reductions in visibility negatively affect recreational values because beauty is one of the major attractions to national forest visitors. Improving national forest roads to reduce dust emissions could be linked to regional transportation plans aimed at reducing air pollution. Such a link might make Forest Service roads eligible for highway funds.

Conclusions—Emissions from national forest roads would need to be included in regional analyses of air emissions. Models to make these analyses are available, but data to represent national forest roads would have to be collected and included in the analysis.

Direct Socioeconomic Effects

Timber Programs

Issues—Road closures are expected to strongly affect Forest Service timber programs. On federal timberlands, the timber program and an extensive road network evolved simultaneously. Many roads were built by purchasers or with purchaser credits from timber sales, but these roads served a variety of users. By the late 1980s, about 25,000 timber sales were recorded per year (of more than $300) supplying 14 percent of the U.S. timber harvest. This harvest supported some 125,000 direct jobs in many communities, mostly in the Western United States. By 1997, the proportion of total U.S. harvest supplied from federal lands had dropped by half because of efforts to protect various habitats for species at risk of extinction.

Along with the evolution of the existing road network went the development of logging systems designed for site conditions, soil-compaction concerns, and costs. Such systems (except for some forwarder systems) are designed to minimize skid distances, both in harvest units and at road-based landings. The most commonly used logging systems (cable yarding or ground-based skidding systems) depend on direct access to a stand. Helicopter and cut-to-length (harvester-forwarder) systems depend on access to nearby stands (usually less than a mile).

Findings—In steep terrain, reducing road densities may require longer cable yarding distances, and because yarding distance is a significant cost factor, especially in thinnings (Hochrein and Kellogg 1988; Kellogg and others 1996a, 1996b) timber harvesting costs likely will increase. In addition, greater reliance could be placed on helicopter logging, which would increase logging costs by as much as 2.5 times. Another result could be more wood left behind in the forest because logs must be bucked to their optimum length to maximize the payload of the helicopter.

In gentler terrain, a reduction in road densities could lead to an increased use of cut-to-length (harvester-forwarder) systems or more reliance on cable yarding. Primary transportation distance (movement of logs from stump to landing) is a variable significantly affecting the productivity of ground-based skidding (Tufts and others 1988) as well as harvester-forwarder systems (Kellogg and Bettinger 1994). Lanford and Stokes (1996) note, however, that at least with similar primary transportation distances in the Southeast, harvester-forwarder systems have comparable costs per unit harvested to traditional ground-based skidder systems, yet with lower environmental effects. If cable yarding replaced some ground-based systems, costs could increase by 1.4 times or more (Kellogg and others 1996b).

Logging cost increases (all else held constant) would reduce the likelihood that proposed sales would sell and lead to reduced harvest. The Forest Service's Washington, DC, office provided an estimate of the extent of these harvest reductions. They estimated that harvests would be reduced by 6 percent in the Northern Region (Montana, northern Idaho, North Dakota, and northwestern South Dakota), 90 percent in the Intermountain Region (southern Idaho, Nevada, Utah, and western Wyoming), and 17 percent in the Pacific Northwest Region (Oregon and Washington). If the issue involves only the use of secondary roads into sale units or just reliance on temporary roads for local sale access, then these effects may be overstated.

More difficult to determine are the long-term effects of focusing future management activities in only the roaded sections of national forests, where one of the primary management tools is stand manipulation through timber-sale contracts. Some management activities, such as prescribed fire, are not road dependent but most of the techniques for stand manipulation require some type of access.

Another issue is how changes in one region relate to changes elsewhere in North America. Reductions in federal timber harvest largely in the West are offset by increases in harvest elsewhere (mostly in Canada and on private timberlands in the South). These offsetting changes are usually sufficient to reduce consumer effects to modest, so that the largest effects are borne by producers (and their employees) in the affected regions.

Reliability, confidence, and limitations—Studies document the effect of skid distances and different logging systems on logging costs (Kellogg and Bettinger 1994, Kellogg and others 1996a, Lanford and Stokes 1996, Tufts and others 1988). Some of these studies were used to support timber appraisal processes. The effect of higher logging costs (because of more expensive logging systems) on stumpage prices has been well documented in the literature (for example, Jackson 1987); stumpage values have to be greater than logging costs for sales to be sold. Increasing logging costs, all else held constant, will result in fewer sales (or more sales being below cost). The effects listed in the findings are uncertain after one to two years because of the ability timber sale planners have to redesign timber sales, including their ability to change harvest unit locations.

Generalizability—The results are generalizable. What does differ are the values for timber throughout the West and the opportunities for less road-dependent logging systems.

Secondary links—The secondary effect of greatest concern is the potential loss of access to stands for forest management activities that remove individual trees. Although much of the current controversy is over final harvest, many other silvicultural practices depend on timber-sale contracts and timber removals to achieve various stand and landscape conditions. Often the forest road network was designed to allow access to multiple stands. Identifying the optimal network in light of potential additions or reductions in roads is difficult (Dean 1997). In addition to considering the loss of access, planners need to consider costs of alternative road building or rebuilding, landslide risks, and expected environmental effects, when they evaluate road management alternatives (Sessions and others 1987). Algorithms to incorporate road management alternatives in forest planning efforts have been described for traditional optimization techniques (Jones and others 1991), as well as heuristic methods (Bettinger and others 1998, Weintraub and others 1995). The effects of road management alternatives on timber programs is a site-specific problem, depending on the road system that exists, the road management alternatives examined, and the condition (age, volume, and so on) of the harvestable timber stands affected by the alternatives. For example, areas of mature forest stands in nonreserved land allocations may be most affected by near-term changes in the road network.

Conclusions—Roads and timber-program issues have been much studied, including attention to the ability to trade off more intensive management on the roaded parts of national forests with the unroaded portions. The ability to address immediate effects (say, for the next fiscal year) is very high, but beyond several years, the ability to predict effects greatly diminishes because no opportunities are available for mitigating the effects of changes in sale location or design. Finally, economic effects tied to changes in timber flows are very real. Roughly 10 direct jobs are generated for each 1 million board feet of harvest from national forests in the West. In addition, payments in lieu of taxes account for significant parts of local government funds in much of the rural West.

From a planning perspective the ability to examine tradeoffs in road system alternatives is moderate. Examinations into the theoretical complexity of road network planning problems have led to the development of planning models designed for integrating road decisions with land management decisions (Bettinger and others 1998; Jones and others 1986, 1991; Nelson and Brodie 1990; Sessions and Sessions 1997; Weintraub and others 1994, 1995; Zuuring and others 1995). These models are particularly useful for measuring tradeoffs among the quantifiable management benefits and costs associated with changes in the road network. Not all issues relevant to a decision can be adequately quantified, however, because the output or response relations are not known or are just being developed. For example, the response variables can be complex and may depend on activities in adjacent stands (see Bettinger and others 1998). In addition to the complex planning model, data development (both geographic information system

[GIS] and associated tabular inventories) is one of the main challenges. The ability to collect and use GIS data as well as the attributes of a road system (and related resources) is evolving and, over time, analyses now based on current data will progressively become more precise and accurate.

Nontimber Forest Products

Issues—A variety of products harvested from the abundant biotic resources of the North Temperate Zone forests are being transformed into medicinals, botanicals, decoratives, natural foods, and a host of other novel and useful products. These renewable, vegetative natural resources harvested for personal or commercial use are called nontimber or special forest products. Consumer forces, changing social climate, and expanding global markets are contributing to the increasing development of these products as viable economic options for sustaining rural communities. Ginseng (*Panax quinquefolius*), goldenseal (*Hydrastis canadensis*), coneflower (*Echinacea angustifolia*), and St. John's wort (*Hypericum perforatum*)—all plants found on national forest lands are major contributors to a multibillion-dollar herbal and botanical industry. Access to these resources has important economic value to those rapidly growing industries. Plants harvested from the wild are "wildcrafted" by harvesters from local communities or contract crews brought in from elsewhere. Particularly for the local harvesters, who operate under the permit system of various public and private land ownerships and who often have low income, access by road to the resource becomes a critical cost factor. In addition, roads create openings important to maintaining diverse species in abundance. How roads will affect the survival and sustainability of nontimber forest products and how access to nontimber forest products will be influenced remain important issues. Both issues are important to the people and communities that already depend on these herbs, shrubs, lichens, fungi, algae, and micro-organisms as part of their economy.

In 1992, the herbal-medicinal market was estimated at just under $1 million and growing at a rate of 13 to15 percent per year (Mater 1997). Traffic USA, a program of the World Wildlife Fund that monitors commercial trade in wild plants and animals, estimates annual retail sales of medicinal plants in the United States in 1997 at $1.6 billion and rising. Of the 25 top-selling herbs in U.S. commerce (Brevoort 1998), more than 50 percent are included in the 1,400 plant species found and traded in the United States. Moss and lichens, harvested extensively from public forest lands and exported to worldwide markets, were valued at more than $14 million in 1995 (Vance and Kirkland 1997). Demand is increasing for huckleberries and mushrooms, important foods harvested for commercial and personal use. In 1995, less than 1 million pounds of the matsutake (*Tricholoma magnivelare*) mushroom were harvested, but in 1997, in one 8-week period, 1.2 million pounds were harvested, which provided the Forest Service with $365,935 in revenue from permit sales (Smith, n.d.). Floral greens are an important mainstay for several markets in the Pacific Northwest. A 1989 study (Schlosser and others 1991) showed that the total value of floral and Christmas greens earned $128.5 million in product sales with about $48 million paid to harvesters, which supported the employment of about 10,000 people and about 675,000 acres in production west of the Cascades. On a single ranger district (Hood Canal Ranger District, Olympic National Forest) from February 1996 through February 1997, 1,500 permits were sold for commercial harvest of greens, bringing in revenue of $63,835. Christmas boughs have continued to increase in demand, and by 1995, harvest in the Pacific Northwest was approaching 20 million pounds per year (Savage 1995).

Findings—Market growth is documented (Mater Engineering 1992, 1993a, 1993b). Collection activities permit information, environmental and other assessments, and maps with roads indicated are part of the written procedures and permitting instructions at forests and districts affected by special forest products. Costs of harvest are recognized as a factor in permit prices, and they influence contract bids in these assessments. Market value is related to cost; increasingly difficult access as plants become scarce may be factored into market value. An assessment in the Southern Region (Alabama, Arkansas, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, Oklahoma, Puerto Rico, South Carolina, Tennessee, Texas, Virgin Islands, and Virginia) identified dozens of plants and products for which free use and commercial permits are issued. Illegal collection is considered a problem in many areas, and some documentation exists in Oregon with the Bureau of Land Management, Forest Service, and state enforcement personnel. Although not explicitly, roads play a role in illegal taking, as well as in monitoring harvest activities. Other reports and inventories have maps indicating roads that offer access to nontimber forest products and often act as a means of pinpointing the desirable harvesting areas. For example, in the special forest products inventory (Karen Theiss and Associates 1996) created for Trinity County, California, roads were used extensively to describe how to find areas where wildcrafters could harvest a particular species.

Reliability, confidence, and limitations—Much of the documentation that relates to special forest products can be found in forest and environmental assessments and in recent reports and papers published in journals and books (Molina and others 1997, Savage 1995, Thomas and Schumann 1993, Vance 1997). In some of these documents, roads are addressed directly about use and compliance with reciprocal agreements where they are in effect. Historically, special products have been administered as a byproduct of timber contracting and road building. The same benefits accrued by recreational collectors of mushrooms, berries, and so on in those areas also could be enjoyed by commercial harvesters. No formal documentation of these benefits going to commercial harvesters is available. Note that some states (e.g., Oregon) require anyone transporting any such product, including firewood, on public roads to have a legal permit or bill of sale.

Generalizability—Generalizing the need for roads or road decommissions for nontimber forest products is impossible. Some populations of harvestable species will benefit from the disturbance caused by building and maintaining roads, and other populations will be harmed. Although enforcement of illegal harvest might be hampered, so would legal harvest. But market forces adjusting for reduced harvest (product scarcity) is unpredictable, and whether any increased value would be transferred to the harvester is not known.

Secondary links—Habitats and plant community structure of some commercially harvested species are linked to roads. From an assessment of 45 commercial species in Oregon, 30 percent can be found in openings and along roadsides. It also is well known that certain species require undisturbed mature forest and would not benefit from the gaps and disturbance caused by roads. Because of the specific habitat requirements of, for example, wild ginger, pitcher plants, and shade-loving mosses, roads would not directly benefit these plants. Some of these species are listed as sensitive, and ready access threatens their survival. Documentation exists for habitat requirements of almost all commercial plants and fungi. Other habitat concerns are related to maintaining roads. A special forest products inventory created for Trinity County, California, suggests that harvesters stay away from roadsides because some Bureau of Land Management and Forest Service districts routinely spray herbicides and pesticides.

Communities and sustainable economies—Many rural areas need more sustainable and diversified economies, for which they may require assistance. The Forest Service recognized this need and developed economic action programs aimed to help communities strengthen their local economies through a range of forest-based resources, including nontimber forest products.

Conclusions—Information on habitat requirements for many of the commercial species is available, and retrospective studies may show how road closures affect species composition; for example, in the prevalence of native versus exotic species (Parendes and Jones 2000). Developing appropriate policies and implementing them for most special forest product species would benefit from information and models that predict regional and general effects from building or closing roads on the species' harvest and sustainability. Information on the economic effects on various components of the industry—from harvester's overhead to product price—is needed. These questions must be answered to determine how building or decommissioning roads would affect the sustainability of particular commercial species and hence the sustainability of the economies reliant on them.

The effects of roads on the economic, social, and biological factors and their effects outlined above need to be documented. Although roads are generally recognized as major components of recreational and commercial-harvest activities that affect hundreds of species in the national forests, systematic studies that integrate these components, much less any individual component, have not been carried out. Only fragmented information on these biological resources, products, uses, values, and habitat considerations is available. Case studies will provide information on local or regional scales, but a comprehensive model of the relation of roads to special forest products nationally requires a comprehensive special forest products database. In addition, an integrated strategy for special forest products that addresses community and resource sustainability together would benefit from targeted and integrated research-based information.

Grazing and Rangeland Management

Issues—According to the 1995 draft RPA program, about 46.2 million acres of national forest lands are considered suitable for livestock grazing. Producing livestock can be an important part of local economies, and livestock grazing is deeply rooted in the culture of the American West and sanctioned by legislation. Grazing was first authorized on national forest lands by the Organic Administration Act of 1897 and confirmed by many later appropriations acts (USDA FS 1989). The Public Rangelands Improvement Act of 1978 reinforced a national policy that public rangelands were to be "managed...so that they become as productive as feasible for all rangeland values." The network of roads on national forest lands has both positive and negative effects on rangelands and the administration of the grazing program. Roads have mostly replaced driveways as a means for transporting sheep and cattle to and from mountain allotments. As a result, these driveways have dramatically improved in rangeland health. Until the 1970s, livestock driveways were considered "sacrifice areas" in the range-management discipline (Stoddart and Smith 1955). Thus, national forest roads can promote ecosystem management objectives along alternative transportation corridors, which they replace. Roads can simultaneously lead to ecosystem changes that reverse rangeland management objectives, however, and increase the administration of the range management program. Administratively, national forest roads allow range conservationists to access allotments quickly by using vehicles rather than horses. But the same roads can produce conflicts between users of the national forests, such as between livestock grazing and recreation interests. And roads can reduce permittee operating costs by providing motorized access to allotments.

Findings—Essentially no scientific information exists that analyzes the ecological, administrative, or economic effects of roads on administering the Forest Service range-management program. Preliminary unpublished analyses from the interior Columbia River basin ecosystem management project addressed the road issue from the perspective of ecological responses to the presence or absence of roads. The analyses found correlations between changes in vegetation composition, riparian functioning, and fire regimes and the presence of forest roads. They could not conclude any cause-and-effect relations from these correlations, however. The program also found higher road densities to be associated with diminished ecological integrity, including those based on range criteria.

To assess the importance of national forest roads for administering the grazing program, as well as their economic value to permittees, an ad hoc interdisciplinary team was formed to provide a nominal assessment. The findings below reflect the input of the team:

- Roads in national forests are essential for administering the grazing program, allowing timely access to allotments. Compliance enforcement was mentioned in particular as an activity greatly benefiting from forest roads. The principal reasons cited were that agency downsizing has resulted in high workloads for remaining range conservationists, which does not allow them sufficient time to carry out their duties; guard stations have been closed; Forest Service personnel no longer have the option of spending nights in the field in some places; and many allotment plans incorporate Forest Service roads into their approved grazing system or as driveways to and from the allotment; for example, in the Black Hills, all driveways are along roads.
- Roads can reduce permittee operating costs by providing motorized access to allotments. The team estimated that, if all national forest roads were closed, permittee costs would increase by three to five times. These costs would accrue from increased riding time, cost of horses and riders, and added equipment costs (such as horse trailers). The grazing program derives benefit from only part of the road system, however, and if arterial and collector roads remained open, the expected cost increases would be less, from none to a twofold increase.
- Roads can heighten conflicts among users of national forests, such as cattlemen and recreationalists, although some evidence shows that concerns about road conditions actually can cause some forest visitors to slightly, but measurably, shift their focus of attention from grazing encounters to roads (Mitchell and others 1996).

Reliability, confidence, and limitations—No peer-reviewed studies have assessed the effects of national forest roads, or roads in general, on livestock grazing or ecosystem management. The results from the Columbia River basin program are tentative and show no causal relations. The results of studies examining the influence of roads on forested landscapes must be carefully extended because the results from studies in Eastern forested landscapes may not apply to Western forested landscapes (Miller and others 1996). The results of the interdisciplinary-team assessment are heavily weighted towards the Rocky Mountain Region (Colorado, Kansas, Nebraska, South Dakota, and eastern Wyoming) and thus may not represent a national perspective.

Generalizations—National forest roads are an important part of range-allotment plans. Roads are also important for administering the grazing program on national forest lands. Ecologically, roads may have a negative effect on rangelands; however, the environmental effects of not having roads are unknown. The team concluded that closing some roads would be acceptable from the perspective of managing the grazing program if the process was systematically evaluated first.

Secondary links—Effects of roads on spread of non-indigenous weeds (biological invasions), wildlife-livestock interactions, and recreation-grazing interactions (particularly with four-wheeling interests) are important.

Conclusions—No science-based information was found on how national forest roads affect livestock grazing. Many questions remain, including the cost of closure to permittees, and the effects of road closure on administering range management programs, including the weeds program, and on compliance.

Energy and Mineral Resources

Issues—The road-related issues associated with energy and mineral resources fall into three overlapping categories: access rights, property rights, and benefits and negative effects. The extractive industries want, and have certain legal rights to, access to public lands to explore for energy and mineral deposits. The access may be on existing forest roads or may require building new roads. The Forest Service road system facilitates providing energy and mineral resources extracted from public lands, which can benefit society. The negative environmental effects of roads used in support of nonrenewable resource extraction are covered in the earlier sections of the synthesis. Mineral developments and oil fields in and of themselves can affect the environment negatively, such as by loss of habitat, increased noise, and added particulate emissions in the air and water, but these effects can be attributed only secondarily to roads; that is, without the road, mineral development might not have taken place.

These issues are a consequence of the inherent nature of the resources and their treatment under existing law. The defining characteristic of energy and mineral resources is nonrenewability; energy and mineral resources are finite, so extraction inevitably leads to resource exhaustion. Depleted deposits must be replaced either through domestic exploration and mine or field development or through importation. In many places, national forest lands are underlain by deposits of nonrenewable resources, some of which are privately held, that make demand for access inevitable.

Federal law and Forest Service policy clearly support exploration for and extraction of resources from public lands. Leasable resources (that is, metallic minerals found on acquired lands and all energy resources) are managed under the Mineral Leasing Act of 1920. Locatable minerals, primarily the metallic ones on public domain lands, are managed under the Mining Law of 1872. Saleable minerals (that is, common varieties such as gravel) are managed under the Mineral Materials Act of 1947. These laws predate the National Forest Management Act of 1976 and the Multiple Use Sustained Yield Act of 1960.

Findings—Under the Mining Law of 1872, U.S. citizens and firms have the right to explore for and stake claims to selected minerals on all public domain lands not specifically withdrawn from mineral entry. Claims are valid in perpetuity or can be converted to private property rights (that is, patented) assuming that appropriate legal requirements are fulfilled. The Forest Service cannot unilaterally deny exploration access to national forest public domain lands, although the agency does have the right to withdraw specific areas from further mineral entry. The agency cannot prevent staking of a claim on these lands, and a claim holder is entitled to use the surface for activities attendant to exploring for, developing, and extracting minerals, within the limits set by federal, state, and local environmental laws. The agency cannot block an otherwise legal patent (that is, deny a claim holder the right to convert the claim to private property). The Congress can, and has, placed a moratorium on new patents, but the moratorium could be lifted in the future. In any event, hundreds of thousands of patented and unpatented claims are already held within the administrative boundaries of the national forests.

The Forest Service has considerably more control over the location of exploration and development activities for leasable minerals than it has for locatable minerals. For national forests and grasslands with completed oil and gas leasing EISs, petroleum exploration activities are restricted to areas designated as appropriate in those documents. The regions also are taking an active role in directing access for leasable minerals. For example, the Northern Region is attempting to restrict oil and gas exploration to areas relatively near existing roads. This approach is not without potential for controversy, however. Decommissioning of roads could be perceived as a de facto withdrawal of the adjacent lands from exploration. The circuit courts are split on the question of whether failure to offer lands for lease is tantamount to withdrawal.

The Forest Service is required by law to provide reasonable access to valid existing mineral rights, regardless of their form, whether unpatented claim, lease, or private property, as a patented claim or subsurface mineral right. An unpatented claim is an implied property right that can be held, sold, or inherited, and access is regulated under the Mining Law of 1872. Patented claims are private property, and access is regulated under the Alaska National Interest Land Conservation Act of 1980 (ANILCA). Coal, oil and gas, and mineral leases also offer a limited form of property right. The rights to individual energy and mineral resources may be held by different legal entities, and the mineral rights may be severed from the surface, which is termed a “split estate.” Access to unpatented inholdings, patented claims, leases, and severed mineral rights can be restricted but seldom denied. Access may be by the existing road system or require new roads. The Forest Service is neither required by law nor expected by industry to build or maintain energy and mineral access roads. Roads built for other reasons (for example, in support of recreation development) might be paid for by the Forest Service but also be used by a mining or energy firm. The firm is always required to maintain the road or to pay for road maintenance called for by their activities; they frequently pay through a reimbursement arrangement with the agency.

The Forest Service can affect the location and design of roads built on national forest lands to support energy and mineral activities. In addition, the agency can sometimes place stipulations on access by limiting road use to certain months, permitting aerial access only, or precluding surface occupancy. Constraints that are unduly expensive to fulfill or so restrictive as to make an otherwise economic mineral deposit uneconomic, however, might well be perceived as denying reasonable access. Temporary roads often are built to facilitate energy and mineral exploration activities. Building plans are subject to review and approval by the agency. If no discovery is made, the exploration firm obliterates the road. Otherwise, the road could be upgraded to permanent status, depending on the circumstances and legal authority. Public use of the road might sometimes be limited because road condition acceptable to the mineral industry might be neither acceptable to, nor safe for, the general public. In addition, other means of access, particularly for exploration, do not require roads, including access by helicopter, foot, horseback, and all-terrain vehicles.

The energy and minerals industries use the existing road system in exploration, development, extraction, and reclamation activities. Only a small portion of the entire road system is affected in any given year, but assuming use of most roads over the long term would be reasonable. Designating a subset of the existing road system as having no future benefit to the industry is not feasible because geographic targets for exploration and development change in response to technological advances and market fluctuations. Limiting mineral exploration access to areas where minerals have already been or are being extracted could preclude future discoveries. Road closures or decommissionings are controversial. Firms wanting to rebuild obliterated roads could face long delays because of the lengthy approval process now in place for building new roads. Such delays could disrupt multiyear exploration and development plans and financing.

The energy and mineral resources produced from national forest lands are essential to the manufacturing, farming, building, and power-generating industries, with a value of $4.3 billion in 1995. Forest Service production represents only a small part of the total value of U.S. production, however. For example, the value of copper produced on national forest lands represents only 1 percent of total U.S. copper. Sometimes, production from national forest lands is a significant percentage of domestic production; national forests produced 80 percent of domestic lead in 1995. Significant amounts of coal and molybdenum also are produced from national forest lands. These contributions to the domestic economy are made possible by use of the forest road system.

Reliability, confidence, limitations, and generalizability—Some case law on energy and mineral access and property rights can be applied more broadly than to the specific litigation reported in it. And for certain situations, existing case law, statutes, and regulations clearly demonstrate the right to reasonable access for existing mineral rights. In numerous other situations, however, the right to access for energy and mineral exploration and development is less clear-cut. Unresolved access issues are associated with both ANILCA and Section 8 of the Lode Law of 1866 (R.S. 2477), which granted right of way across unreserved public domain lands. Considerable debate continues on the degree to which this right has been modified by subsequent legislation.

Secondary links—Roads built to provide access for energy and mineral exploration and development often are heavily used for other purposes. Secondary links can be found to recreation, species endangerment, biological invasions, and many other areas. The effects from energy- and minerals-related roads and road usage are comparable to those of other roads in the Forest Service system built to the same specifications and carrying the same types and amount of traffic. Unpaved Forest Service roads frequently are topped with a layer of aggregate or crushed stone, and the material often has been extracted from Forest Service lands. Thus, the extent of the road system also has implications for the volume of aggregates extracted; fewer miles of road built and maintained implies fewer tons of aggregate and crushed stone extracted.

Conclusions—The legal issues surrounding energy and mineral road access and usage will require the input of the Office of General Council: Pamela Piech (202/720/2515) is an expert on the Mining Law of 1872; James Snow (202/720/6055) is an expert on RS2477 and ANILCA. Little or no research has been published on the secondary links associated with energy and mineral road usage. One key area for future nonlegal research is to determine the landscape-scale effects of energy and mineral development; for example, extensive oil-field road networks may lead to habitat fragmentation.

Another need is to determine exactly which roads are currently being used for access to explore, develop, extract, and reclaim. Quantifying the effects on road condition of nonrenewable resource activities by number and size of vehicles is also important, and another management need is to identify the roads leading to or adjacent to valid existing mineral rights.

Resource-Based Outdoor Recreation

Issues—Almost all the different types of public recreational uses of national forests depend in one way or another on roads for access. Whether, when, and where various recreational uses occur depend on the availability of access to, and the extent and location of, the road system. Altering this system is likely to have widespread and differing effects across different types of uses. In considering the future of roads on national forests, the general question is, “What are the direct, indirect, and secondary effects on recreation from possible changes in national forest road systems?” More specifically, “What are the direct effects of changing the class, spatial density, ecological distribution, maintenance, and total mileage of national forest roads on the density, placement (ecologically and socially), mix, economic value, experience quality, and amount of recreation uses?” As well, “What are the indirect effects on access to views of natural scenery and on the quality of scenic resources, and what are the secondary effects on the economic and social viability of communities in the area and the condition of the forest ecosystem?” Answers to these and many other questions are needed as input when national forest road policies are considered and in seeking to optimize net benefits across multiple roads.

Findings and hypotheses—The relations between roads and recreation on national forests is highly complex and includes many direct, indirect, and secondary links that are not well understood. Research findings specifically addressing these links are limited and uneven across the questions we have posed. Indirect evidence and related research provide the following insights and hypotheses:

- Roads provide corridors of access to a variety of national forest sites, settings, and viewing opportunities for widely diverse users. Almost all recreation use in national forests depends to some degree on road access. Sightseeing, driving outdoors for pleasure, and developed camping are examples of activities that directly use roads as a part of the recreation experience. Backpacking, white-water boating, and birdwatching are examples of activities usually away from roads, but the user still must access areas of interest by using them. Altering road systems can disrupt long-established access and use patterns and, at least in the short run, result in not meeting visitors’ expectations. Less road mileage or maintenance, or both, can lead to uneven shifts in recreational opportunities across different user, socioeconomic, and ethnic groups who depend differently on roads for access.
- Roads provide staging access to remote areas and wilderness, but the presence of roads can at the same time reduce opportunities for solitude and perceptions of wildness. The amount, placement, and class of roads are positively correlated with the amount and concentration of recreational uses. But visible roads, greater numbers of users, and sounds from motor vehicles can interrupt solitude and perceptions of wildness for wilderness and other backcountry users.
- As demand for forest recreational opportunities continues to grow locally, regionally, and nationally, even a stable amount and condition of forest roads likely will result in increased congestion, lowered satisfaction, and user conflicts. Outdoor recreation trends show recent strong growth in participation across a wide spectrum of activities and segments of the American public (Cordell and Bergstrom 1991). Projections

show this growth is likely to continue well into the future for all nature-based activities except hunting (Bowker and others 1999). At the same time, access to private lands is continuing to decrease and be limited to lessees and friends of the owners (Cordell and others 1999). Public lands are likely to be the destinations of choice for increasing numbers of people looking for high-quality outdoor recreational experiences in natural settings. Several national parks already have limited motorized access to bus tours or other public transportation as one way to address increased congestion from private cars. Continued growth in demand without increases in road systems or limits to use of private cars likely will lead to lowered satisfaction and more conflicts at the more popular national forests (Tarrant and others 1999). Changes in satisfaction likely will differ significantly by setting (for example, as distinguished in the recreation opportunity spectrum [Tarrant and others 1999]). Direct recreational access, the character of and access to scenic views, and provision of increasingly sophisticated visitor services (including rescue and medical services) will depend on the character of the road system in place.

Reliability, confidence, and limitations—Data on national forest use and the relations of roads to that use are unreliable, but a national project is underway to develop an improved use-monitoring system. Data from the customer project provide insights into user perceptions of experience quality related to national forest attributes, including roads (Tarrant and others 1999). Social group differences between users of roaded, near road, and backcountry settings are available for the U.S. population in general, and to some degree for national forest users. Science-based methods are available for examining in more depth the relations between roads, recreational use, visitor satisfaction, and economic values and effects. Little research exists to guide management for optimizing recreational benefits from roads and globally optimizing multiple benefits across the broad range of national forest road uses.

Secondary links—Even though increased use (on the same or fewer miles of forest roads) or changes in the mix of recreational uses, or both, may increase aggregate visitor spending (and thus general economic effect), the distribution of economic effects among economic sectors and regions is likely to be altered. The biophysical effects of recreational use on forest ecosystem conditions are confined mostly to near-road zones, the site of most use. The biophysical condition of affected sites tends to stabilize after each successive increment of recreation use, although the resulting condition may be unacceptable to managers, users, or both. Specific links between recreational use and conditions of ecological components and links between recreational use and other resource uses are not well known.

Conclusions—Quantitative and qualitative methods, research underpinning the recreation opportunity spectrum, and a wealth of related published and unpublished literature dealing with economic values (Bergstrom and Loomis 1999); secondary economic effects (Archer 1996, Bergstrom and others 1990); visitor perceptions and behavior (Tarrant and others 1999; Williams and Patterson, in press), resource and social capacity (Shelby and Heberlein 1986); conflicts, consumption, and future projections of road-based recreation (Cordell and Bergstrom 1991, Bowker and others 1999, Cordell and others 1999), and social justice assessment are available. For the most part, however, existing databases and literature have only indirectly addressed the hypotheses described above that deal specifically with the relations between roads and recreation (for example, Knight and Gutzwiller 1995). Substantial research is needed to better understand direct and indirect relations between road-system characteristics, recreational use, and ecosystem conditions, including issues such as the introduction of exotics, soil erosion, habitat fragmentation, forest-product harvesting, wildlife disturbance, riparian vegetation, and fire.

Indirect Socioeconomic Effects

Fire

Issues—The increasing density of road networks in and adjacent to many forest, shrub, and rangeland areas has been an important factor in changing patterns of disturbance by fire on the landscape. Roads provide access that has increased the scale and efficiency of fire suppression, and roads have created linear firebreaks that affect fire spread. These factors can be useful in both fire suppression and prescribed fire operations. In addition, road access has undoubtedly contributed to increased frequency of human-caused ignitions in some areas.

Findings—That improved road access leads to increased efficiency and effectiveness of fire-suppression activities is a long-held tenet of fire fighting. Much of the effectiveness of past fire-suppression policies probably can be attributed to increased access for ground crews and equipment, particularly under weather and fuel conditions where fire behavior is not severe. Under the severe conditions associated with intense, rapidly spreading fires, the value of forest roads for access or as fuelbreaks is likely to be minimal. Although little has been published in the science literature to quantify these effects, a study in southern California concluded that the road network had been a key factor in determining what suppression strategies were used, both in firefighter access and because roads were widely used for backfiring and burning-out operations (Salazar and Gonzalez-Caban 1987). Early studies of fuelbreak effectiveness in southern California came to similar conclusions (Green 1977). Daily costs of fire-fighting activities unfortunately are of little value in answering the question of how much road access increases efficiency, because fire-fighting agencies tend to put money and resources into fighting fires with access, which confounds the results. In spite of this, strong anecdotal evidence supports this effect.

An important issue in the Western United States is building new roads to allow harvest and prescribed fire to reduce fuel accumulations in ecosystems where past management (principally fire suppression and harvest) have increased the risk of large, severe wildfires (Lehmkuhl and others 1994). The principal concern here is the tradeoff between reducing the effects of wildfire and increasing the risks of road effects on aquatic habitat. In the Columbia basin, scientists concluded that “it is not fully known which causes greater risk to aquatic systems, roads to reduce fire risk, or realizing the full potential risk of fire,” and that more research is needed (Quigley and others 1997). Some potential considerations in setting priorities for forest health treatments have been suggested in an adaptive management framework for addressing this concern (Rieman and Clayton 1997). We currently have few data on how these processes might be affected by road networks, although a study after the 1987 Stanislaus fires in California suggests that cross-slope road networks reduced sediment delivery to debris basins (Chou and others 1994).

The benefits that roads provide for fire prevention and fire management carry an associated cost. For purposes of simplicity, we will highlight them here in place of a second fire section under the “undesirable or negative effects.” Indirect effects of increased access have increased the role of human-caused ignitions, particularly in areas of expanding urban and rural development into wildland interfaces (Hann and others 1997). The high rate of human-caused fires in the Blue Mountains of eastern Oregon is associated with high recreational use in areas with high road densities (Hann and others 1997). The importance of human-caused ignitions as an issue may depend on what resources are considered of concern. For example, in the Southwest, numbers of ignitions go up with access, but numbers of ignitions are not limiting to maintaining fire regimes, but fuel loadings and climatic conditions are (Swetnam and Baisan 1996). Numbers of ignitions are important determinants of fire risk, however, in areas such as wildland-urban interfaces for which maintaining historical fire-regime patterns is not the overriding issue. In addition, numbers of ignitions are important determinants of fire risk in some wildland-urban interfaces where fire intensities are often higher (such as chaparral), and active suppression of ignitions by people may be critical to maintaining historical fire patterns (Conard and Weise 1998).

Road networks have resulted in changes in fuel patterns and fire regimes at the broad scale. If we accept that road networks have been important in effectively suppressing fire and that they alter fire patterns on the landscape, then road systems are, in some sense, linked to changes in fuel patterns and fire regimes. Before fire-suppression activity in the Western United States, fuels were maintained at relatively low amounts in dry forest types, with high fuel loads restricted to small, isolated patches (Agee 1993). As access increased, areas burned by wildfire declined, at least through the 1960s. As a result of suppression supported by access (in part), fuel accumulations increased and areas with moderate to high fuel loadings became larger and more contiguous. This pattern of change has been documented for the entire upper Columbia River basin, where scientists assert that fire suppression has generally been more effective in roaded areas, which has resulted in roaded areas in the upper basin departing further from unaltered biophysical templates (as measured by dominant species, structures, and patterns) than have the unroaded areas (Hann and others 1997). Roads (along with other human disturbances such as clearcutting) contribute to new disturbance patterns at the landscape scale, both by increasing efficiency of fire fighting and providing barriers to fire-spread that are different from natural barriers (Swanson and others 1990). Increased emphasis on removing roads in certain environmentally sensitive areas will reduce access for fire suppression and prescribed fires, potentially leading to increased fuel accumulation and fire hazard in some areas.

Reliability, confidence, and limitations—Logic and anecdotal evidence for the contention that road access increases effectiveness and efficiency of fire suppression efforts are strong, but quantifying this issue in terms of cost savings or size and severity of fires is not well documented. The scientific support for the contention that roads serve as firebreaks is strong, but how important this effect is in controlling the pattern of fire on the landscape is not clear; the ecological implications of this pattern change also are not clear. The secondary effect of roads providing access for timber harvest that has resulted in changing mosaics of fire is strong; the ecological consequences, while strong, are highly variable. Long-term effects on changing fire regimes in the Western United States are well documented. Increased access probably leads to increased human-caused ignitions, but the implications of this increase differ from area to area. Increased ignitions at urban-wildland interfaces are likely to be a problem, but it may be unimportant in affecting fire regimes in less-developed landscapes in the West. Building roads to provide access to reduce fuel in fire-suppressed forests is likely to enhance this activity, but it may carry added risks to aquatic environments over the risk of fire alone.

Generalizability—Most of the concerns addressed here apply primarily to the Western United States. In much of the East, road networks are well developed and relatively stable because of terrain and vegetation differences. Wildfire interactions are likely to be similar to those described for the West, but the effects are likely to be significantly less. In the Southeast, where use of prescribed fire is widespread, roads are frequently used as firebreaks. Much of this activity is on private lands, however, and a high proportion of the road network is state and county highways rather than Forest Service roads.

Secondary links—Fire issues are linked to issues of forest (ecosystem) health and aquatic habitat.

Conclusions—In general, the importance of roads for providing access and firebreaks is well established, although literature on cost-to-benefit ratios is lacking; most evidence is anecdotal. The issue of road access to lessen fire risk and improve forest health in unroaded areas is heating up, and little published research is available to fall back on for resolving the debate.

Forest Research, Inventory, and Monitoring

Issues—Among the benefits that roads provide is access for research, timber and non-timber forest inventories, and monitoring. Although the economic scale of these tasks may be low compared to some other activities, the knowledge derived may be key for managing other access-related uses, in addition to the more general objectives sought. Hence, understanding the relation of roads to inventory and monitoring activities is not a trivial issue.

Findings—Although finding sufficient data for a complete and wide-ranging analysis is difficult, the role roads play in inventory and monitoring access (that is, the cost per plot) can serve as a surrogate for the larger problem. Plot-survey contracts are based on four categories in which the proximity to roads plays a significant part. For example, costs run about $600 per plot when roads allow access to within 0.25 mile of the plot sites. In the same region, cost rises to $1,300 per plot in roadless areas open only to foot access. In the Pacific Northwest, the nearly 650 wilderness plots, of a total of 11,360 in all terrain, had survey costs only about 23 percent greater ($1,460 per wilderness; $1,174 per nonwilderness plot). The data did not permit comparing the cost difference of road-accessed plots in the Pacific Northwest Region over the montane sites in the Pacific Southwest Region, however. More extreme conditions are encountered in Alaska, where roadless areas are vast, yet helicopter access is permitted. The average cost per plot for roadless areas in the Alaska interior has averaged $4,000 per plot for 170 plots. Obtaining good data for comparing areas covered by these approaches is generally difficult because photo-interpretation based on aerial photo coverage is used to supplement ground-survey efforts.

Reliability, confidence, and limitations—Problems of access to survey plots for research, inventory, and monitoring will clearly raise costs of operations. The exact differences can be quantified by taking terrain differences, size of roadless areas, and means of permitted entry into account. For this study, we used only a few data points from limited regions to understand the extent of this issue. More comprehensive analyses are possible with existing data, given the resources to do them. The data are sufficiently robust to suggest that the cost elements relating to access constitute a factor in research, inventory, and monitoring. Whether the magnitude of the contribution of such uses constitutes a significant economic component when compared to, say, recreation is not clear, however.

Generalizability—The data examined for this order-of-magnitude approach were taken from limited observations originating in the Pacific Southwest, Pacific Northwest, and Alaska, with Alaska representing extreme conditions. Corroboration for the observed higher cost resulting from the absence of road access was attained qualitatively for the Eastern Region of the Forest Service.

Secondary links—Access issues have similar aspects whether extraction (such as timber, mining, and grazing), recreation, inholdings, or related activities are considered. The links do suggest that coordination of overlapping uses be a variable examined when road density and road-network planning are considered.

Private Inholdings

Science-based sources of information have not been found on the relations between roads and private inholdings. The following propositions are therefore offered as hypotheses based on judgment, not scientific findings. These propositions do not necessarily apply to inholdings dedicated to mineral and energy exploration or extraction, which are covered in “Energy and Mineral Resources,” above.

- The Forest Service is required by law to permit access to private inholdings.
- The Forest Service can require private inholding owners or lessees to comply with official regulations and standards that apply to building roads on or through national forest land. The regulations and standards are documented in writing as official policy, but they are subject to interpretation and application in specific cases by agency line officers.
- The Chief (of the Forest Service) may consult appropriate national forest policy offices and line officers about the sources of scientific documentation used in practice and official regulations, standards, and procedures applicable to roads on or through national forest lands that provide access to private property.
- In general, the scientific documentation of ecological and human effects of roads on or through national forest land provided elsewhere in this synthesis applies to roads that provide access to private inholdings.
- No scientific basis exists for stating propositions about whether the Forest Service subsidizes access to private inholdings or the effect, if any, of Forest Service roads on the market, use, and passive-use values of private inholdings.
- The Chief needs inventory information about the type, number, acreage, location, use, value, and so on of private inholdings on national forest land and the extent to which private inholdings use national forest roads for access. At present, no systematic inventory procedure or documentation can provide comprehensive and valid information of that type.

Nonmarket and Passive-Use Value

Issues—A comprehensive understanding of the economic effects of roads in the national forests must include both effects that can be measured in dollars (market effects) and those with no direct dollar values (nonmarket effects). The influence and importance of market values to land management decisions is obvious, and measuring and comparing effects of management decisions that affect market values are relatively simple. For example, the cost of building and maintaining a road into a forest can be readily compared to the income generated from harvesting the timber accessed by that road. Also important, but far more difficult to measure and compare, are the things people care about for which no market exists, such as access for hunting, bird watching, and wilderness experience.

Natural resource economists have invested much effort over the last several decades to develop and test methods for estimating nonmarket values. The methods can produce useful information, but they are costly and their validity has not yet been demonstrated sufficiently to satisfy many economists (Arrow and others 1993, Cambridge Economics 1992, Mitchell and Carson 1989, Portney 1994).

Economists generally classify nonmarket values as either active or passive. The term "active-use value" applies to goods and services used in some activity like recreational fishing, skiing, or camping. The term "passive-use value" includes two categories (Peterson and Sorg 1987, Randall 1992): things people appreciate without actually using them or even intending to use them (like a distant wilderness or an endangered plant or animal) are called "existence values"; and things people want to remain available for others (such as their descendants) to use and appreciate are called "bequest values."

Environmental economists often define and measure these nonmarket values in monetary terms, but monetary valuation is often not possible, cost-effective, or appropriate. All nonmarket consequences of national forest roads and of any changes to these roads must be considered in road management and policy decisions. For example, passive-use values are likely to strongly affect decisions about preserving areas without roads or about removing existing roads to create roadless areas. Thus, the nonmarket consequences need to be identified in some way—either in monetary terms or by some other means.

Under regulations of the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA), as amended, 42 U.S.C. 9651 (c), a United States Court of Appeals for the District of Columbia ruled in 1989 that passive-use values "...reflect utility derived by humans from a resource and thus, prima facie, ought to be included in a damage assessment." Thus, if Forest Service roads significantly alter passive-use value, whether positively or negatively, such value needs to be considered in road policy and management decisions. Failure to include these nonmarket values in an economic evaluation, when such values are judged to be important, presents the manager with biased information that could lead to inefficient and unfair allocation of resources.

Significant questions: Under what conditions do people assign passive-use value to national forest landscapes or their attributes? Forest Service officers responsible for road policy and management need to know the forest landscape conditions to which people assign passive-use or other nonmarket values, how such values differ among individuals and groups of people, the strength or significance of the value assigned, how changes in the landscape affect the nonmarket values, and how such values trade off with other forest-related values assigned by affected people.

Do Forest Service roads, road policies, or road management actions strongly affect passive use and other nonmarket values? If so, how and why? A related question is whether the effects of roads on nonmarket values affect people differently and differ by landscape. For example, if the supply of landscape that provides passive-use value is sufficiently large in a given region, small increments of road building or decommissioning may not affect people very much. Many small encroachments could produce severe cumulative effects, however.

Findings—People do assign passive-use value to natural resources, especially roadless areas and natural areas with unique characteristics. And the passive-use value often exceeds the active-use value served (or potentially served) by road access (Bengston and Fan 1997; Brown 1993; Driver and others 1987, 1996; Payne and others 1992; Walsh and others 1984, 1990).

Building roads in roadless areas may reduce passive-use value significantly; decommissioning roads may increase such value. Building roads into roadless areas may serve values that require such access, however, and decommissioning roads may obstruct values and uses that require access. Decisionmakers need to consider all these tradeoffs. Individuals and affected groups often disagree aggressively about the passive-use value of specific roaded and roadless areas and the effects of building or decommissioning those roads (Bengston and Fan 1997). Thus an equity (or distribution) question must be considered: Whose desires should the Forest Service fulfill when stakeholders' values conflict? What criteria should be used to decide among them? What approaches can be taken to resolve the conflict?

The effects of roads on passive-use value differ by location and circumstance. Differences in the quality and uniqueness of landscapes modify passive-use-value effects from building or decommissioning roads. The relation between supply and demand also will affect the extent and strength of a passive-use value. For example, if many substitutes for a given roadless landscape exist, building a road in that area may have little or no effect on its passive-use value, just as the hunter's killing of a single elk does not reduce the passive-use value of elk because the species is still abundant. Likewise, if an abundance of roads are provided to resources that people want for active use, decommissioning or closing one road will have little effect. People with strong attachments to a special place, use, or road may suffer loss, however, unless they can find and adapt to a substitute.

Validly and reliably measuring changes in passive-use and other nonmarket value is costly and can sometimes exceed the cost of being wrong. Managers of national forest roads must understand such values, however, and the circumstances under which they are significant decision factors, to assure that the values can be included where appropriate. A survey-based method called contingent valuation (contingent valuation generally uses surveys or interviews to determine how much people say they would be willing to pay for some nonmarket good) that asks people to state their willingness to pay for nonmarket values can provide a useful indication of relative magnitude, but applying it to passive-use value of public goods is where the method is most vulnerable to flawed results, criticism, and controversy. Studies must be designed and applied carefully and the results interpreted cautiously. Other methods, such as value juries (Brown and others 1995), focus groups, public hearings, and other forms of public participation also can provide useful information. Quantitative measures should be taken only when the scale of the problem justifies sufficient investment for scientifically rigorous results.

If fully and correctly disclosed, the cost of opportunities foregone by preserving a roadless landscape can serve as the price to be paid for the values served by preservation. Preserving a roadless area may sometimes cause an opportunity cost in the form of alternative uses foregone, such as timber harvest, developed recreation, or fire suppression. If the opportunity cost has been fully disclosed to the decisionmaker, a decision to preserve a roadless landscape is a policy acknowledgment that the value created exceeds that opportunity cost. In a decision about whether to designate an area as roadless, opportunity cost can sometimes serve as the price to be paid for whatever values, including intangibles, are served by the designation. Stakeholders and decisionmakers can then decide—by judgment, negotiation, or analysis—whether the gain is worth the price (Bell 1996; Fight and others 1978, 1979; Randall and others 1979).

Reliability and degree of confidence—The scientific literature supports the general propositions that roadless natural landscapes and unique natural features and resources generate passive-use and other nonmarket values; that such values differ among individuals, groups, and landscape conditions; and that disagreement about nonmarket value fuels conflict. Legal precedent also validates policy concern. The effects of roads on passive-use and other nonmarket values have not yet been studied extensively, and the validity and reliability of methods for measuring the necessary values are still questionable.

Generalizability—No science-based procedures, analytical methods, formulas, tables of values, or handbooks are available for applying the general principles we have outlined to specific decisions or to transfer measured values from one place to another. Each project-scale decision requires original human-dimension inventory and assessment techniques, either by technical measurement or through public involvement. Managers making decisions on whether to build or remove roads in specific places always need to consider the principles and questions defined in the findings section. Roadless areas may have significant passive-use and other nonmarket value, depend-ing on the people affected and the availability of substitutes, but obtaining the required information requires original inventory and assessment for each decision. Expensive procedures may not be appropriate where the scale of the problem does not justify the cost.

Research in progress is exploring nonmarket active-use-value transfer (that is, generalizing by formulas and tables) among different site-specific situations. The results thus far are encouraging but not conclusive, although they may offer useful guidance in some situations (Rosenberger and Loomis 2000). We are not aware of any similar work on passive-use-values.

Secondary links—Passive-use value affects public attitudes toward the Forest Service as well as public willingness to accept and support proposed forest policies and plans. Roads and roadless areas sometimes take on symbolic meaning in the broader context of environmental concerns about such things as biodiversity, pollution, and ecosystem health. Passive-use value associated with symbolic issues triggered by changes in road distribution can be an important cause of conflict and litigation.

Conclusions—Extensive scientific evidence exists on passive-use and other nonmarket values in general and on applying them to unique natural environments, environmental accident damage assessment, and sensitive species. Little scientific evidence is available on the relations among roads, roadless landscapes, and passive-use value, however. Published studies demonstrate that people often do assign significant passive-use value to natural areas, including roadless ones, in specific places (Bishop 1978; Brookshire and others 1986; Carson and others 1999; Cicchetti and Wilde 1992; Ciracy-Wantrup 1968; Crowards 1997; Farmer and Randall 1998; Freeman 1993; Krutilla 1967; Krutilla and Fisher 1975; Loomis and White 1996; Mazzotta and Kline 1995; Morton 1999; Walsh and others 1984, 1990). National forest roads can be an important cause of ecological degradation. Under the right conditions and taken together, those studies also imply that national forest roads can cause a significant loss of passive-use values. The actual effect on passive-use value will be specific to the site and situation, however; the only refereed studies we found that document the specific relation between roads and passive-use value are Brown and others (1996) and Champ and others (1997). Rosenberger and Loomis (2000) compiled a comprehensive tabulation of nonmarket recreational values, including a bibliography of 162 studies.

Additional studies are needed to test hypotheses or estimate parameters that apply to specific decisions. General methodological and theoretical research not specifically focused on forest roads is ongoing in several disciplines, including environmental economics, sociology, psychology, political science, and anthropology. Several approaches are being pursued, including social and psychological surveys, ethnographic studies, methods for effective citizen participation, focus groups, citizen and value groups, and monetary valuation. The needed and ongoing research is long term, however, and must not delay making decisions in the short term, based on the best available current knowledge.

Heritage and Cultural Value of Roads

Issues—In addition to satisfying the American penchant for sightseeing by car and other forms of recreation requiring auto travel, roads and their features themselves sometimes have heritage value because of historic significance or architectural features. Roads also may affect areas considered sacred by American Indians or other religious groups. These issues can affect the legal and political framework for Forest Service road policy and management because important historical, social, and cultural values are often part of developing, maintaining, or decommissioning roads. Forest planning for transportation and for individual roads should incorporate information on heritage and cultural values for both roaded and unroaded areas.

Findings—Roads and associated features are part of the history of the nation. Some features are significant for their association with exploration and settlement, others for accomplishments in engineering, and still others for reasons of local history and culture. Roads and other transportation features figured prominently in the early nonindigenous settlement and development of the nation. Roads that were or are significant in this way include early Spanish roads, such as El Camino Real (the Royal Highway) in California and New Mexico; those that follow the routes of American Indian trails (Davis 1961); military roads such as Cook's trail, which crosses the forests of northern Arizona (Scott 1974); and some early routes established for commerce, such as the Santa Fe Trail, which crosses the Cibola National Forest. Given their historical role, such roads (many still in use) often are eligible for the National Register of Historic Places. Of equal importance, historic roads often have special meaning to people who live near them or have used them. Route 66, for example, which crosses the Kaibab National Forest, is considered historically valuable for its role in establishing regular, all-season east-west automobile transportation to California (Cleeland 1988, 1993).

Features forming part of or associated with a road may be historically or culturally valuable for their own merits (Fraser 1987). Bridges and other features built by the Civilian Conservation Corps often are fine examples of engineering and considered eligible for the National Register of Historic Places (Throop 1979). Many such bridges are on Forest Service roads. Roads also may have heritage value as part of a cultural landscape, such as the landscapes associated with homesteading, ranching, or logging. Even roadside advertising can have local cultural significance, such as the hand-painted message along an abandoned highway in the Cibola National Forest that claims "Curandera cures all." The National Park Service and the U.S. Committee of the International Council on Monuments and Sites recognized the heritage value of transportation corridors in a conference held in 1993 (USDI 1993).

Building, maintaining, and decommissioning roads can affect historical and cultural values. Roads often directly affect historical and archaeological sites. Building, maintaining, or decommissioning roads can damage or destroy archaeological sites (Spoerl 1988) with earthmoving equipment used on buried and surface remains, such as structures and other cultural materials. Roads also affect sites indirectly by increasing erosion or by making sites accessible to vandals. Less tangibly, but no less important, roads often affect areas that American Indians consider sacred, may limit their ability to conduct ceremonies that require privacy, and may even diminish the sacred qualities of such places. Building new roads, or adding to existing ones, can affect sacred areas that may qualify for the National Register of Historic Places as Traditional Cultural Properties (Parker and King 1990). The Cibola National Forest has recently been in litigation initiated by Sandia Pueblo over plans to rebuild a road through Las Huertas Canyon in New Mexico. The pueblo claims that the canyon is eligible to be a Traditional Cultural Property. A larger issue in this case is that the road and the traffic it brings affect use of the area for pueblo ceremonies. In northern California, similar issues surrounded the case of the Gasquet-Orleans Road on the Six Rivers National Forest (Theodoratus and others 1979), which concerned road building and resource extraction in an area that local American Indians considered sacred. The dispute over this road lasted many years, and its repercussions continue to be felt.

Generalizability—The findings are partially generalizable to all national forests but not to all decisions. As with sensitive species, some issues arise where heritage and cultural values are especially significant. Because of legal requirements and the intensity of concern among affected stakeholders, however, assessing cultural and heritage values is essential in every Forest Service decision about building or decommissioning roads.

Secondary links—Inadequate participation in road policy decisions by affected stakeholders concerned with heritage or cultural values can lead to litigation and political conflict. It also can stimulate symbolic opposition to the Forest Service on other fronts that even direct amelioration of the heritage or cultural concerns cannot resolve.

Conclusions—Good information is available on cases encountered by the Forest Service; it is generally after the fact, however, and pertains to actions taken to resolve conflicts caused by failure to consider the issues early and effectively in policy and management decisions. Existing information about heritage and cultural values relating to roads and roadless areas often may not be adequate; ongoing inventories tend to be project-specific rather than part of the general program. Obtaining information about sacred places from some American Indian groups is difficult because Forest Service styles of communication and negotiation often are incompatible with these cultures, and revealing sacred values and identifying sacred places to outsiders may be thought to imperil the values in need of protection.

Documentation—Much of the documentation for the heritage and cultural values of roads resides in administrative documents in the 50 state historic-preservation offices and the Advisory Council on Historic Preservation.

Economic Effects and Development

Issues—Both benefits and costs are associated with building, maintaining, and continued use of Forest Service roads. Likewise, benefits and costs are associated with removing existing roads. The issues revolve around whether the good things outweigh the bad things and what the extent of roads should be in national forests.

Findings—Some economic activity is supported by building and maintaining roads: economic activity also is supported by decommissioning roads. Analyses for the 1995 RPA program suggest that about 33 jobs economy wide (nationally) are supported per $1 million expenditure on building and maintaining roads (Alward and others 2000). A reasonable speculation might be that roughly the same rate of employment would be supported by removing existing roads and restoring the land underlying them. Road building and removal represent one-time stimuli to the economy, but maintaining roads is a recurring stimulus. After a road is removed, the jobs supported by road maintenance cease.

The major effects of roads on local economies, however, would be expected to result from the economic activity those roads support by providing access to the national forest and to communities in or near it. On Forest Service roads, that activity includes logging, silvicultural operations, and recreation, among others. Also supported is economic activity that depends on recreation, such as guides, outfitters, and rafting permittees. The roads also provide access for land management and firefighting operations.

Indirect (and approximate) indications of the amounts of economic activity that might be associated with changes in Forest Service roads can be obtained from several sources. Reports indicate that timber harvest from national forests supports about 16.5 jobs economy wide (in the local area) per million board feet harvested (USDA FS 1996). That estimate is conservative because it is based on summed local-area models. Recreational use of national forests supports a range of 1,000 to 2,000 jobs economy wide (nationally) per million trips, depending on the primary activity, based on analyses done for the 1995 RPA program (Alward and others 2000, Archer 1996).

Use of public lands, in general, follows roads. In Alaska, for example, intensity of use by both hunters and nonconsumptive wildlife users follows road corridors (Miller and McCollum 1997). Further, we hypothesize that more casual users—such as scenery gazers, picnickers, car campers, and day hikers that constitute the bulk of national forest recreationists—probably stay closer to the road than do some hunters and backpackers, the minority of national forest recreationists.

Whenever timber is cut and removed from the forest, roads will be needed; even helicopter logging at some point converts to road use by truck hauling. One issue is the quality of the roads and the length of their lives; that is, whether they are permanent and remain after timber harvesting ceases, or temporary and closed after harvest. Permanent roads are available for other activities over time, primarily recreation and management activities. Temporary roads are available for timber activity and some incidental activity during harvest, but when the roads are closed, benefits accruing from those roads cease. That the cost of maintaining a road over time could sometimes outweigh the cost of removing it at the end of one timber harvest cycle and rebuilding it for the next one is at least conceivable. Environmental effects (and cost) of multiple entries and decommissioning of temporary roads must be balanced against those of a single permanent road. Permanent roads cost more to build and maintain than temporary ones, with increased potential for degrading the ecosystem, but they can result in more benefits over longer periods than temporary roads because of the access they allow.

Roads affect spatial patterns of forest use. Changes in roads change those patterns. Recreational users are particularly attracted to or driven away from particular areas by the availability and ease of access. With decreased access to the national forest, some users might drop out and give up outdoor recreation. Others would shift their use to other areas, some on Forest Service land and others off. The result would be reduced economic activity in the locale where forest access was decreased and increased economic activity in areas where displaced users moved. In general, the effects would be reversed if access were increased. Sometimes, however, increased access could lead to decreased use and result in less local economic activity; for example, where new roads and associated commercial activity degrade a viewshed, which could decrease visits to view autumn foliage.

Another result of spatial shifts in recreational use could be to concentrate use in areas to which displaced users move. Concentrated use may increase environmental effects as well as decrease the quality of people's experiences. Crowding imposes costs on existing users in those areas by diminishing the benefits they received from their recreational use because of the inflow of displaced users from areas affected by decreased road access.

Anything that affects the demand for and benefits received from recreation and other uses of Forest Service land has subsequent economic effects, and it may alter development because land uses drive local economic activity. Forests and local economies will be affected differently, depending on the mix of local activities.

Building or removing Forest Service roads and maintaining existing roads can help mitigate ecosystem degradation associated with roads. Note that the tradeoffs are between the expense of minimizing or eliminating environmental degradation associated with Forest Service roads and access to Forest Service lands with associated economic activity.

Many roads are or have been funded by the timber program. Benefits accrue from use of those roads beyond timber, largely for recreation. This contrast presents a classic problem of joint cost allocation, and the accounting problem of attributing cost should not be used as an excuse for looking only at specific programs or components of the Forest Service mission.

The jobs and other economic activity supported by building and maintaining roads must be balanced against the cost of building and maintaining those roads, including costs resulting from choosing not to maintain selected roads. The question is, do the benefits associated with the roads, both direct and indirect from all sources, justify the cost incurred by society, including costs of increased ecosystem degradation from deferred or inadequate maintenance? Reports like this one can provide information on a wide variety of benefits and costs, but answering the question just posed is a policy decision.

Reliability, confidence, and generalizability—Analyses done for the 1995 RPA program provide a broad picture of national effects that can be expressed as averages and rates per unit of activity. They are not site-specific studies, and they do not estimate the effects on local areas. A few recreation-demand studies based on specific sites and regions provide corroborating evidence of the qualitative results (English 1997, McCollum and Miller 1994, Miller and McCollum 1997). The transportation literature contains some studies on roads and development (Berechman 1994, Broder and others 1992, Rephann 1993, Rietveld 1994), but those studies are mainly about highway systems, and though we expect their conclusions to be qualitatively relevant to the types of roads administered by the Forest Service, some attributes of Forest Service roads are so different that creating a complete picture is impossible. A primary gap in knowledge is understanding the links between policy or management actions and their effects on forest-based activity (both in the amount of activity undertaken by users and in the benefits they receive), especially for recreational and noncommodity uses. Changes in road availability and quality affect whether and how much users access the forest in particular areas. Road availability and quality also affect the quality of users' experiences, and thereby affect the benefit they receive. No access or access on a poorly maintained road, for example, could decrease benefit for some activities but have little or no effect on others. We did not find any activity-specific studies documenting the direction and size of such effects. Those factors are relevant because they drive demand for access to Forest Service land and the local economic activity associated with use of these lands.

Further gaps in knowledge exist on the distributive effects of new or improved and degraded or removed roads on forest use in local areas and on local economic activity. To what extent do the existence or lack of Forest Service roads, and their condition, attract or drive away users pursuing particular activities? The general development literature provides some insights and qualitative expectations for Forest Service roads, but empirical findings on the likely size of the effects are absent.

Conclusions—Empirical estimates are not available to document the size of the economic contribution of recreation-dependent commercial activities like guides, outfitters, and rafting permittees. Also missing are empirical estimates of benefits received from and economic activity supported by specific recreation activities in specific areas. Estimates are often obtained from national studies or site-specific studies in other areas and blindly applied to areas being analyzed.

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Appendix

This section draws from the analysis in the main document, with interpretations relevant to roadless and unroaded areas.

Forest Service Roadless Areas: A Synthesis of Science Information

Managing and maintaining existing forest roads has not kept pace with either the shifting balance of forest users or the increased scientific understanding of the ecological effects of roads. In particular, entry into roadless areas merits consideration of both benefits derived and risk of unacceptable impacts. Thus, managing for roadless area protection consists of positive steps such as providing for habitat conservation areas, watershed protection, critical habitat protection, contingency or passive-use values, and related land stewardship objectives. It also consists of restricting actions that may contribute to deteriorating environmental integrity, such as stand-replacing fires or large-scale insect outbreaks.

Questions affecting roadless areas include:

- Are significant and important social values associated with the existence and protection of wilderness and roadless areas?
- Does a road network in itself pose a risk to the integrity (as defined in the interior Columbia River basin study) of roadless forested ecosystems?
- Do roadless areas make substantial contributions to maintaining biodiversity and desirable habitat characteristics?
- Can roadless areas stay intact without management efforts that are facilitated by roads (for example, fire prevention, disease and pest control)?
- Does creating new roads in roadless areas have overriding benefits that outweigh the potential ecological costs?

Existing and perhaps new science information may be needed to assess some or all of the questions posed. In addition, methods from the social sciences are available to conduct surveys and assessments of public perceptions, values, and beliefs to determine the values that roadless areas hold in the mind of the public. This summary of existing information is an attempt to identify the ecological and biophysical characteristics of large nonroaded blocks of the forest and rangeland ecosystems that would permit conclusions about the value of maintaining such landscape features, and to examine the scientific aspects of a possible rationale for road building in currently roadless areas.

Ecological and biophysical aspects of roadless areas—An approach for providing the scientific basis of ecological and biophysical value is to summarize the known information on roadless areas at the landscape or large basin scale and proceed to smaller spatial scales. Questions that may be asked at the larger scale include the following:

- Is retention of existing roadless areas an important as part of a conservation strategy?
- Does the distribution of roadless systems affect the success of conservation strategies?
- Does the size of individual roadless areas affect the success of conservation strategies?

One of the few examples of landscape-scale analysis of road influences is the interior Columbia River basin environmental assessment. Analysis of fish distribution and status data for seven species of anadromous and resident salmonids in the Columbia basin showed that frequency of strong populations generally declined with increasing road densities. Additional analyses of road effects focused on four non-anadromous species, because effects of roads and other land uses on anadromous species may be masked by migrational and ocean-related factors (for example, dam passage, predation, and harvest). Three species showed significant effects from roads, either when occupied spawning and rearing areas were distinguished from unoccupied areas or when strong status was differentiated from depressed. The analysis suggested a decreasing likelihood of occupancy—or a decreasing likelihood of strong status if occupied—with increasing road density. No other variables except ground slope showed the consistent patterns across all species shown by the road density measures.

The investigation of the influence of roads on population status clearly showed an increasing absence and a decreasing proportion of strong populations with increasing road density for several subgroups. Additional evidence suggests that the lowest mean road density values (number of road miles per unit of area) always are associated with strong population status.

Based on the synthesis reported in the main body of this document, this trend is apparent for Yellowstone cutthroat trout, even though it was the only subgroup not showing a significant road effect in a logistic regression analysis. The lack of statistical significance in the face of apparent trends, however, points to complex interactions among the explanatory variables not adequately addressed in the relatively simple logistic model. Consistent, significant effects for other species may be further testament to the presence and pervasiveness of the effects. Strong relations between roads and the distribution and status of these species were detected despite the potential confounding effects of other variables (such as harvest, non-native introductions, and other habitat factors).

These results show that increasing road densities and their attendant effects are associated with declines in the status of four non-anadromous salmonid species. These species are less likely to use highly roaded areas for spawning and rearing and, if found, are less likely to have strong populations. This consistent pattern is based on empirical analysis of 3,327 combinations of known species' status and subwatershed conditions, which were limited primarily to forested lands administered by the Forest Service and the Bureau of Land Management. We would not expect the relation to be as strong on the nonforested, lower gradient lands administered by BLM. Of the four species examined, the redband trout is the only one supported by the low-gradient lands. Only in forested, high-elevation areas could redband trout status be clearly associated with road density changes.

Most aquatic conservation strategies acknowledge the need to identify the best habitats and most robust populations to use as focal points; from these, populations can expand where adjacent habitat can be usefully rehabilitated or the last refugia of a species can be conserved. These strategies also provide necessary experimental controls for evaluating the effects of land management activities in other areas. The ecological importance of unroaded areas has been highlighted in the Columbia basin assessment and in other reports cited in the main body of this paper.

The overlap of unroaded areas within and outside designated wilderness areas with stronghold watersheds for fish and other important conservation watershed efforts in the Columbia basin also was examined. Designated wilderness and unroaded areas are important anchors for strongholds throughout the basin. Unroaded areas occupy

41 percent of area with known and predicted strongholds in the east-side EIS area. One-third of this area is outside wilderness. Sixty-eight percent of known and predicted strongholds in the upper Columbia basin EIS area are unroaded, of which 37 percent are outside of wilderness.

Aquatic integrity in the Columbia basin was analyzed in relation to road densities and integrity ratings for other resources (forest, range, hydrology). Forest clusters with the highest integrity ratings for aquatic organisms were associated with low road densities; low integrity ratings corresponded with moderate or higher road densities. The range cluster having the highest aquatic and composite integrity also had mostly low road densities. The relations between road densities and integrity ratings for other range clusters were more variable, however (FEMAT 1993, Henjum and others 1994, Lee and others 1997). The correlation of basin or subbasin integrity is not total, suggesting the variables and interesting mechanisms are complex and nonuniform. Such data suggest that criteria be developed to examine the role of roadless areas in conservation strategies and permit assessing the risks taken when roadless blocks that are significant features at the landscape level are further intersected by roads.

- Does the distribution of roadless areas contribute to the ecological integrity of forested ecosystems?
- Does a conservation strategy that includes roadless areas need to be spatially explicit?

The distribution and the desirability of having well-distributed roadless area systems pose interesting scientific challenges. Historical trends significantly influenced the extent and distribution of roadless areas. Logging progressed from easily accessible, low-elevation forests to more difficult, high-elevation terrain; thus the remaining road-less areas tended to be at high elevations. We are unaware of a systematic analysis of this issue. Criteria that include assessing how well some roadless areas represent certain native ecosystems should be considered. This is especially the case at lower elevation sites that historically have seen the greatest harvesting effort and attendant road building. If the goal is to have a system of reserves consisting of representative, relatively undisturbed habitats, then roadless areas and the habitat types within them should be distributed over major ecoregions and be derived logically.

- Do corridors connect the high-quality roadless areas?

Biodiversity is, in simplest terms, the variety of life and its processes (Keystone Center 1991). Recent syntheses (Heywood and Watson 1995) emphasize the reciprocal relation between biodiversity—conceived as genetic and species diversity—and ecosystem function. The many species representing the biodiversity of an area play roles necessary for ecosystem function and, importantly, are the source of the variation enabling an ecosystem to adapt to change. The processes of a healthy, functioning ecosystem in turn support the many species. Appreciating the reciprocity means that biodiversity can be taken as a natural measure of the ecosystem as a whole and thus can integrate the many concerns listed.

Some species may play more important roles than others in the normal functioning of an ecosystem. Keystone species, for example, may define the major structural elements of an ecosystem as Douglas-fir does for forests in the Pacific Northwest, or they may—by virtue of their position in a complex trophic structure—act to maintain the diversity as keystone predators do for herbivores. The many species that do not seem to serve an important role in an ecosystem constitute a reservoir of potential adaptation to change.

Because an ecosystem cannot predict change, the very diversity of species acts as a hedge against it. Thus, biodiversity is important to long-term ecosystem function, and human activities that decrease biodiversity can impair it. Our working hypothesis is, then, that measures of biodiversity provide the best integrative assessment of the effects of roads on ecosystems.

Forest roads create corridors that not only permit invasion of alien, weedy species, but also permit entry of predators, including humans, to the forest environment and affect wildlife populations. Limited studies have shown that roads allow exotic species into areas where they historically have been absent or where appropriate habitat was not available (Parendes, 1997). Clearly, these secondary effects are promoted by the existence of roads but are not due to the roads themselves; however, the increase in human access to remote areas allowed by roads has a far more significant effect on native populations. High road densities are associated with a variety of negative human effects on some wildlife species. Black bear populations are inversely related to road density in the Adirondacks (Wisdom and others 2000). Increases in hunting pressure, particularly illegal hunting, have the potential to impact populations. Moose and caribou are particularly vulnerable to this kind of predation (Scott and Servheen 1985). Such connectivity will be important for endangered species where the gene pool is already limited, such as in the case of the Florida panther (*Puma concolor corgi*), and where gene exchange between populations in adjacent habitat may help species viability (Shrader-Frechette 1995). Connectivity also is important for species having large home ranges, and road avoidance or risk from road related mortality constitutes an additional threat to the populations, or may lead to undesirable, even dangerous animal-human interaction, as may be occurring with mountain lion (*Felis concolor*) populations in southern California.

Whenever forest roads are built, modified habitat and changes in animal behavior will lead to changes in risk to viability and distribution and even local extirpation in wildlife populations. Road avoidance behavior is characteristic of large mammals such as elk, bighorn sheep, grizzly bear, caribou, and wolf. Avoidance distances of 100 to 200 yards are common for these species. Road usage by vehicles and humans has a significant role in determining road avoidance behavior. In a telemetry study of black bear movements, interstate highways were almost never crossed, and roads with low traffic volume were crossed more frequently than roads with higher traffic volumes (Wisdom and others 2000.). It appears that in some cases, male bears may actually be using roads as travel corridors (Young and Beecham 1986, Zager 1980). Wolves in Wisconsin are limited to areas with overall mean road densities of 0.07 miles per square mile. Some studies have shown that the existence of a few large areas of low road density, even in a landscape of high average road density, may be the best indicator of suitable habitat for large vertebrates (Wisdom and others 2000.).

- Are roadless areas important to the conservation of high-quality aquatic and terrestrial habitats?

Again drawing on the Columbia River basin assessment, fish with strong populations occurred more frequently in areas with lower road densities. Supplemental analysis further showed that increasing road densities and their attendant effects were associated with declines in the status of four non-anadromous salmonid species. Fish seem to be less likely to use highly roaded areas for spawning and rearing and, where found, are less likely to have strong populations. Patterns based on empirical analysis of 3,327 combinations of known species status and subwatershed conditions are consistent and unmistakable, though limited primarily to forested lands administered by the Bureau of

Land Management and Forest Service. Although unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds. Possible reasons for this coexistence are that, in general, increased short- or long-term watershed and ecological risks are associated more with entering an unroaded area than with proceeding continuously with management activities in roaded areas to upgrade, maintain drainage, or close or obliterate existing roads (Lee and others 1997). The empirical evidence is correlational and, when the causes for the above observations are fully established, a more complex picture is likely to emerge.

At a more local scale, hydrologic and geomorphic interactions are a potential consequence of road building and presence that can involve altered flow regimes, increased sedimentation, local failures with local and "downstream" consequences for streams, riparian areas, and vegetation cover. For example, the FEMAT (1993) analysis stats, "Management activities in roadless areas will increase the risk of aquatic and riparian habitat damage and potentially impair the capacity of Key Watersheds to function as intended...[while]...most timber-suitable roadless acreage can be harvested either directly from existing roads or from helicopters." Further, "if all timber-suitable roadless remains unroaded in Option 9, then the estimated reduction for the total regional probably sale quantity is less than 0.2 percent." In terms of aquatic effects, the Columbia basin assessment summaries include the following statements: "Roads provide access, and the activities which accompany access magnify the negative effects on aquatic systems beyond those solely due to roads." Among other findings, the assessment "...subwatersheds supporting strong populations were found on Forest Service administered lands (75 percent) and a substantial number (29 percent) are located within designated Wilderness areas and National Parks." Thus, the data "...clearly show increasing absence and decreasing proportion of strong [fish] populations with increasing density for some subgroups" (FEMAT 1993). Other studies found that the length of road segments connected to the stream network at stream crossings or gully-debris slide tracks amounted to a 40-percent extension of the stream network length in a Cascade Range watershed (Jones and others, in prep; Wemple 1999).

High-quality terrestrial habitats may be affected by the potential for invasion of exotic plants and animals that can displace or threaten native populations; that is, affect biodiversity, which can be increased by roads. Migrating populations of rare amphibians may be killed during road use; disease and pathogens are spread more rapidly and widely if roads are present (Kiester and Slatkin 1974). The preponderance of the negative findings in many scientific studies also suggests that the potential for ameliorating or minimizing the unwanted effects exists, even if it has not been made a prime objective historically. Lastly, some positive ecological results may follow (though they are proportionately less significant) that roads create edge environments exploited by small mammals, can sustain some desirable species, and provide useful niches. Maintaining an optimum balance is a function of the long-term magnitude of road networks; for the present system, the need for additional niches and habitats is difficult to demonstrate.

A full scientific view of the data on roadless areas cannot stop at the local scale, but must ultimately view the presence of roaded and roadless areas in a landscape context and be able to draw the distinction between a large road network and small roadless areas or large roadless areas and a small road network. Again drawing on the Columbia basin assessment, we note that "while unroaded areas are significantly more likely to support strong populations, strong populations are not excluded from roaded watersheds.... the scale of the subwatershed (8000 ha on average) at which strong populations are identified may mask potential disconnects between the real locations of strongholds and roads. The significance of the impacts and benefits will be affected and must withstand rigorous scientific approaches over a spectrum of possibilities and of scales" (Lee and others 1997).

Social, aesthetic, and economic values of roadless areas—The interaction between roadless areas and people's aesthetic and spiritual beliefs about the landscape probably affects people's perceptions in many different ways. We know that passive or "nonuse" values include "existence" and "bequest" value. Existence value pertains to things, places, or conditions people value simply because they exist, without any intent or expectation of use. Bequest value pertains to a desire people may have to allow others, such as future generations, to receive benefit from a resource (Peterson and Sorg 1987, Randall and others 1979). The issues are as follows:

- People assign significant passive-use value to national forest landscapes or attributes.
- Forest Service road policies or management actions affect passive-use values.

People do assign passive-use (nonuse) value to natural resources, and passive-use value may exceed the active-use value served by road access to the resource. Invasion of roads will reduce some aspects of passive-use value in natural areas. Likewise, obliteration of roads may increase such value. Building roads into roadless areas may, however, serve values that require access, and obliterating roads may obstruct values and uses that require access, so tradeoffs need to be considered. Though not universally shared, a strong value is doubtless attached to the continued existence of wilderness and roadless areas, including those in national forests.

The relation between roadless areas and recreation on national forests is highly complex. Research findings are limited and uneven on the issues of direct, indirect, and secondary effects on recreation of altering the national forest road system. Indirect evidence and related research provide the following insights:

- Roads provide corridors of access to various national forest sites, settings, and visual and aesthetic experiences; in fact, almost all recreation in national forests depends to some degree on road access.
- Roads provide access to remote areas and wilderness but at the same time can reduce opportunities for solitude elsewhere.
- The amount of roading and the amount of recreation use are positively correlated, sometimes leading to heavy concentrations of use, and roads may be the only means of enjoyment for persons with some forms of disability.
- Demand for forest recreational opportunities continues to grow regionally and nationally.
- Placement, scale, class, and setting of roads can greatly affect the quality of scenic views of national forests and access to outstanding vistas.

The three most highly ranked uses of lands administered by the Forest Service and Bureau of Land Management in the basin today are timber, fishing, and hunting. Projected uses by 2045 will be motor viewing and day and trail use; this for an area where 70 percent of the unroaded areas of >200,000 acres occurs in the lower 48 states (Cordell and Bergstrom 1991, Tarrant and others 1999).

- Does a roadless area preclude needed access for public services and resources as well as conservation management?

Roadless areas not already congressionally withdrawn (for example, as a designated wilderness area) total about 34 million acres in national forests. Of these, 9 million acres have been identified as suitable for timber production. Management practices and natural resource use may suggest strong reasons for entry into the 9 million acres (Coghlan and Sowa 1997). Timber harvesting using roadless approaches in these areas would lead to greater reliance on helicopter logging systems, which increase logging costs. The FEMAT study (1993) suggests that in key watersheds, the reduction in timber volume would be about 0.3 percent, and reduction by prohibiting entry into existing roadless areas not congressionally withdrawn in all areas considered by FEMAT (that is, the range of the northern spotted owl) would be 6 percent.

For the interactions of grazing rights, grazing access, and roads, essentially no scientific information exists analyzing the ecological, administrative, or economic effects of roads on administering the Forest Service range management program, and the synthesis in the main report did not uncover data specific to the relation of roadless areas and grazing practices (Peterson and Sorg 1987).

That improved road access leads to increased efficiency and effectiveness of fire suppression activities is a long-held tenet of fire fighting. Much of the effectiveness of past fire suppression policies probably can be attributed to increased access for ground crews and equipment, particularly under weather and fuel situations where fire behavior is not severe. Under the severe conditions associated with intense, rapidly spreading fires, the value of forest roads for access or as fuel breaks is likely to be minimal. However, quantification of these effects in published research in the United States is minimal. But it should be noted that indirect effects of increased access have increased the role of human-caused ignitions, and this is particularly true in areas of expansion of urban and rural development into wildland interfaces.

Roadless areas: conclusions—The scientific literature provides a framework of general principles regarding the nonuse values of present roadless areas and may even be extended to apply to areas where road decommissioning may recreate roadless areas. Such values include areas (1) having significant amounts of interior habitat for many forest species now being observed under the “survey and manage” concept of the Northwest Forest Plan, (2) maintaining connectivity of habitat for species having large home-ranges, (3) valuing the existence of forest “reserves” that permit the continued functioning of representative habitat types in a state of least human disturbance, and (4) becoming aware that forest-stream interactions seem to confer somewhat stronger fish viability in areas of low to no road densities. At present, no science-based analytical models, formulas, tables, or handbooks are available that the manager can use to apply the general principles to specific decisions, though pilot efforts are now underway by the USDA Forest Service to develop such tools. Such tools will provide methods that permit judgments about offsetting benefits and impacts from road building and usage, which suggests that we will have the means at hand to decide on an agreed on mix of roaded vs. roadless areas in national forests.

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Hermaphroditic, Demasculinized Frogs after Exposure to the Herbicide Atrazine at Low Ecologically Relevant Doses

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Hermaphroditic, demasculinized frogs after exposure to the herbicide atrazine at low ecologically relevant doses

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Atrazine is the most commonly used herbicide in the U.S. and probably the world. It can be present at several parts per million in agricultural runoff and can reach 40 parts per billion (ppb) in precipitation. We examined the effects of atrazine on sexual development in African clawed frogs (*Xenopus laevis*). Larvae were exposed to atrazine (0.01–200 ppb) by immersion throughout larval development, and we examined gonadal histology and laryngeal size at metamorphosis. Atrazine (≥0.1 ppb) induced hermaphroditism and demasculinized the larynges of exposed males (≥1.0 ppb). In addition, we examined plasma testosterone levels in sexually mature males. Male *X. laevis* suffered a 10-fold decrease in testosterone levels when exposed to 25 ppb atrazine. We hypothesize that atrazine induces aromatase and promotes the conversion of testosterone to estrogen. This disruption in steroidogenesis likely explains the demasculinization of the male larynx and the production of hermaphrodites. The effective levels reported in the current study are realistic exposures that suggest that other amphibian species exposed to atrazine in the wild could be at risk of impaired sexual development. This widespread compound and other environmental endocrine disruptors may be a factor in global amphibian declines.

In the last 10 years, a great deal of attention has focused on the global presence of endocrine-disrupting contaminants in the environment (1, 2). Similarly, a great deal of attention has focused on global amphibian declines (3, 4). In the case of amphibian declines, efforts focus on identifying causes (5), whereas for endocrine disruptors, the "causes" have been identified and studies focus on identifying effects of endocrine disruptors in the environment (6–11).

Atrazine (2-chloro-4-ethytlamino-6-isopropylamine-1,3,5-triazine) is the most commonly used herbicide in the U.S. and probably the world. The U.S. Department of Agriculture reports that more than 30,000 tons (60 million pounds) are used annually in the U.S. alone (12). Atrazine has been used for over 40 years and currently it is used in more than 80 countries. Despite its widespread intensive use, atrazine is considered safe because of its short half-life and negligible bioaccumulation and biomagnification (13). Also, atrazine seems to have very few effects on adults and reportedly induces abnormalities and deformities only at very high doses. As a result of the high doses required to produce deformities, it has been suggested that "direct toxicity of atrazine is probably not a significant factor in recent amphibian declines" (14). Here, we test the hypothesis that atrazine may interfere with metamorphosis and sex differentiation at ecologically relevant low doses via endocrine-disrupting mechanisms.

Materials and Methods

Animal Breeding and Larval Care. We report results from two experiments that used frogs from two separate sources. Adults from Exp. 1 were from a long-term captive colony maintained at the University of California, Berkeley, whereas adults from Exp. 2 were obtained from Nasco (Fort Atkinson, WI). In both experiments, three females and three males were injected with human choriogonadotropin (1,000 international units) 6 h before harvesting gametes. Eggs were manually stripped from the female and fertilized *in vitro* in 0.3 × modified mammalian Ringer's solution by using the sperm obtained from the dissected testes of the three males. The embryos were allowed to hatch. After 4 days, the larvae were all mixed and netted into tanks 5 at a time repeatedly, until all tanks contained 30 larvae. Larvae were reared in 4 liters of aerated 10% Holtfreter's solution (15) and fed a solution of ground Purina rabbit chow daily. Food levels were adjusted as the animals grew to maximize growth.

Dosing. In Exp. 1, we exposed larvae to atrazine at nominal concentrations of 0.01, 0.1, 1.0, 10.0, and 25 parts per billion (ppb), whereas the second experiment used 0.1, 0.4, 0.8, 1.0, 25, and 200 ppb atrazine. Concentrations were confirmed by two independent laboratories (PTRL West, Richmond, CA, and the Iowa Hygienic Laboratory, Univ. of Iowa, Iowa City, IO). All stock solutions were made in ethanol (10 ml), mixed in 15-gallon containers, and dispensed into treatment tanks. Controls were treated with ethanol such that all tanks contained 0.004% ethanol. Water was changed and treatments were renewed once every 72 h. Each treatment was replicated 3 times with 30 animals per replicate (total of 90 animals per treatment) in both experiments. All treatments were systematically rotated around the shelf every 3 days to ensure that no one treatment or no one tank experienced position effects. Experiments were carried out at 22°C with animals under a 12-h/12-h light/dark cycle (lights on at 6 a.m.). Animals were exposed throughout the entire larval period, from hatching [Niewkwoop–Faber (NF) Stage 48 (16)] until complete tail reabsorption (NF Stage 66). In all experiments, all treatments and analyses were conducted blindly with color-coded tanks and treatments and number-coded specimens.

Gross Measurements. At metamorphosis (complete tail reabsorption—Niewkwoop–Faber Stage 66), the date was recorded for each animal. Each animal was weighed to the nearest 0.002 g on a Mettler AT 261 Delta Range balance and its total length was measured to the nearest 0.5 mm. Animals were anesthetized in 0.2% benzocaine (Sigma), assigned a unique identification number, fixed in Bouins' fixative, and preserved in 70% ethanol until further analysis.

Gonadal Analysis. Initially, the sex of all individuals was determined based on gross gonadal morphology (Fig. 1). Sex identification was confirmed by histology for 10 animals per tank. Further, histological analysis was conducted on all animals for which the sex was ambiguous when determined by gross mor-

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Abbreviation: ppb, parts per billion.

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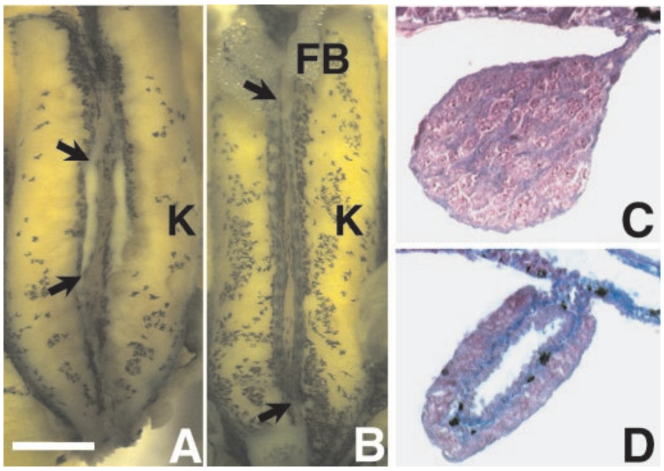

Fig. 1. Gonads of a control postmetamorphic male (*A* and *C*) and female (*B* and *D*) *X. laevis*. *A* and *B* show the entire dissected kidney–adrenal–gonadal complex preserved in Bouins' fixative. *C* and *D* show 8 μm of transverse cross-sections through the animals' right gonad stained with Mallory's trichrome stain. [Bar = 0.1 mm (*A* and *B*) and 10 μm (*C* and *D*)]. FB, fatbody; K, kidney. Arrows (in *A* and *B*) show the anterior and posterior ends of the animals' right gonads. The yellow color in *A* and *B* is a result of fixation in Bouins' fixative. Without fixation, the gonad is transparent. The ovary is distinguished by its greater length, lobed structure, and melanin granules. Although some specimens' ovaries lack pigment (especially atrazine-treated animals), testes never have melanin in this species. Histologically, the ovary is distinguished by the ovarian vesicle (hole in the center) along its entire length and the internal ring of connective tissue (in blue). Note the melanin granules (black) in the connective tissue in *D*.

phology. All histology was conducted according to Hayes (17). In brief, tissues of interest were dissected and dehydrated in graded alcohols, followed by infiltration with histoclear and paraffin. Sections were cut at 8 μm and stained in Mallory's trichrome stain.

Laryngeal Size. Serial transverse histological sectioning was conducted on the larynges of 10 males and 10 females from each replicate from all treatments in both experiments. Histology was conducted as described above. To estimate the size of the larynx, the *M. dilator laryngis* was measured. We used the largest cross-sectional area (transverse section) as a measure of muscle size. Initially, 10 sections were taken from 100 animals (distributed over all treatments from Exp. 1) until a region approximately one-third through the larynx was repeatedly determined to be the largest section. For the final analysis this region was identified by shape. Thus, similar sections were measured for each individual. Images of this section from each animal were recorded with a Sony DKC-5000 and analyzed with METAMORPH software (version 2.75, Universal Imaging, Media, PA).

Adult Treatments. Newly metamorphosed animals were too small to obtain enough plasma to measure hormone levels. Thus, studies of effects of atrazine on hormone levels focused on adults. For adult studies, males and females were obtained from a long-term captive colony at University of California, Berkeley. Adults were maintained under the same light and temperature cycles as described for larvae. Animals were acclimated in 10% Holtfretter's solution for 5 days and then exposed to 25 ppb atrazine. Water was not aerated, animals were fed Purina trout chow daily, and water was changed and treatment renewed every 72 h. Animals were treated for 46 days. At the end of the exposure, animals were killed by decapitation, and the blood was collected. Plasma was collected and stored frozen until analysis.

RIA. For testosterone analysis, plasma was extracted with diethyl ether and dried under nitrogen. All samples were reconstituted in PBS with gelatin (PBS-g). Hormone assays were conducted as described in Hayes and Licht (18). Testosterone antisera were obtained from Endocrine Sciences (Calabasas, CA) and were validated for several species including *Xenopus laevis*. Plasma from controls and treated animals was assayed in the same assay at 3 doses and the assay was repeated 3 times. Intraassay variation was 1.0%, and interassay variation was 1.3%.

Statistical Analysis. Statistical analysis was conducted with the aid of SYSTAT software (SPSS, Chicago). Sex ratios were analyzed by using the G test with Wilkin's g- adjustment as described in Hayes and Menendez (19). Similarly, mortality was analyzed by using the G test. Time to metamorphosis and size (length and weight) at metamorphosis were analyzed by using ANOVA with treatment, tank, and sex (sex nested within tank and tank nested within treatment) as independent variables. In addition, we conducted correlational analyses to determine whether laryngeal size correlated with time to metamorphosis, size, or atrazine dose. Also, we scored all animals as to whether they were greater or less than the mean laryngeal size for controls and then conducted a G test to determine whether the number of affected animals in the treatment group changed with atrazine treatment. Finally, we used Kendall's ranked coefficient to determine whether the percentage of below-average animals varied with the dose of atrazine.

Results

Mortality, Development, and Growth. At the doses tested, atrazine exposure had no effects ($P > 0.05$) on mortality, time to metamorphosis, length, or weight at metamorphosis (not shown).

Effects on Primary and Secondary Sex Differentiation. Males and females were sexually differentiated at metamorphosis based on gonadal morphology and histology (Fig. 1). At all doses tested (except 0.01 ppb), atrazine produced gonadal abnormalities. Up to 20% of the animals (16–20%) had multiple gonads (up to 6 in a single animal) or were hermaphrodites (with multiple testes and ovaries; Fig. 2). These abnormalities were never observed in control animals in the current experiments or in over 10,000 observations of control animals in our laboratory over the last 6 years.

Control males had larger larynges than females at metamorphosis, but males exposed to atrazine ($\geq$1 ppb) had reduced larynges (both studies; Fig. 3 *A* and *B*). When we examined the proportion of "below-average" animals against dose, we found a threshold effect at 1 ppb (both studies; Fig. 3*C*), but Kendall's rank coefficient suggested a dose effect with increasing proportions of affected males associated with increasing atrazine doses ($P < 0.01$; Fig. 3*D*).

We hypothesized that the effects of atrazine were caused by a disruption of steroidogenesis (20–27). Further, we showed that sexually mature males suffered a 10-fold decrease in plasma testosterone (Fig. 4).

Discussion

Although data from two experiments are reported here, these studies have been repeated four times, including an unpublished report and a study submitted to the U.S. Environmental Protection Agency (28). In total, atrazine exposure at these levels has been replicated 51 times by our laboratory with similar results. We chose *X. laevis* for these studies, because it is a well studied laboratory model for which the effects of sex steroids are well known. Exposure to exogenous estrogen in this species results in 100% females (29, 30), whereas androgens increase laryngeal growth but do not affect gonadal differentiation (30,

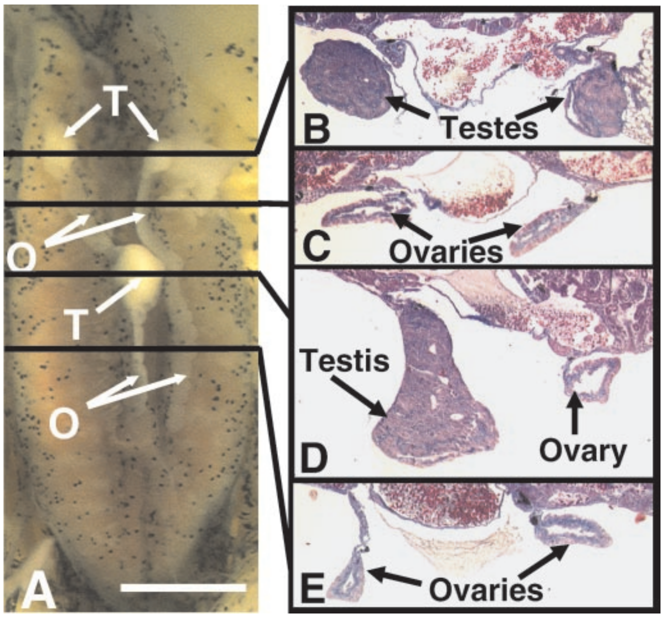

Fig. 2. An atrazine-treated hermaphrodite. The specimen shown was treated with 1 ppb atrazine. *A* shows the entire dissected kidney–adrenal–gonadal complex. *B–E* show 8 μm of transverse cross-sections (stained with Mallory's trichrome stain) through the areas indicated by the lines in *A*. [Bar = 0.1 mm (*A*) and 25 μm (*B–E*)]. FB, fatbody; K, kidney; O, ovary(ies); T, testis(es). Note the absence of pigment in the ovaries, which was typical of hermaphrodites.

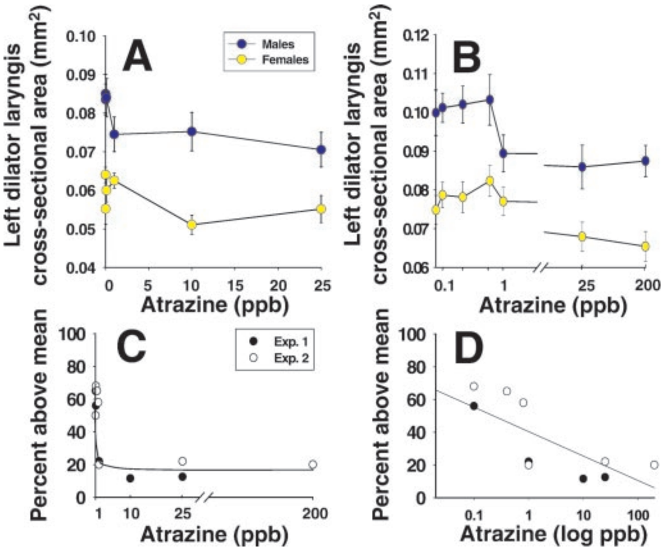

Fig. 3. Results of measurements of the left laryngeal muscle (*M. dilator laryngis*) in control males and females compared with atrazine-treated animals. In Exp. 1 (*A*), atrazine ($\geq$1 ppb) reduced laryngeal size in males but did not affect females. Doses of 0.01 and 0.1 ppb did not have a significant effect. In Exp. 2 (*B*), 0.1–0.8 ppb atrazine did not have a statistically significant effect on laryngeal size but again, exposure to $\geq$1 ppb atrazine significantly reduced laryngeal size in males ($P < 0.05$). Laryngeal size was greater in animals from Exp. 2 compared with Exp. 1, suggesting a population difference in the absolute size of the larynges, but the relative sizes (male to female and atrazine-treated compared with controls) were similar within each experiment. *C* and *D* show two interpretations of the data by using analysis of the proportion of above-average males for both experiments. Atrazine exposure ($\geq$1 ppb) significantly decreased the proportion of males that were at or above the mean for control males (G test; $P < 0.05$) and suggested a threshold effect at 1.0 ppb in which 80% of the exposed males were below average (*C*). Kendall's rank coefficient analysis ($P < 0.01$), however, suggested a relationship between dose and the proportion of affected males with a decrease in the proportion of normal males with increased dose (*D*). Note that control males were normally distributed with exactly 50% of the individuals above the mean in both experiments.

31). Thus, endpoints for detecting sex steroid-like or antagonistic effects are well defined for this species. The current findings suggest that atrazine inhibits testosterone and induces estrogen secretion.

Previous studies have suggested that atrazine is an endocrine disruptor, but these effects have been observed in a single strain of rat or were produced only at high doses (32–38). In fact, no published studies have addressed effects of atrazine at concentrations considered safe in drinking water or safe for limited human exposure—3 and 200 parts ppb, respectively (39). Also, until now, the potential endocrine-disrupting effects of atrazine have not been examined in amphibians, although teratogenesis, mortality, and growth effects have been examined at high doses (14, 40–45). In the cited amphibian studies, deformities, acute toxicity, or physiological impairments were not detected below atrazine doses of 47.6 ppm.

Disruption of steroidogenesis by atrazine has been reported in mammals (20–26) and reptiles (27), however. Several of these studies reported the induction of aromatase and an increase in estrogen. Here, we suggest that the same mechanism may explain the effects observed in *X. laevis*. An induction of aromatase may result in the decrease in androgens (as androgens are the substrate for aromatase). The loss of masculine features, such as the decreased laryngeal size, may be a result of the decreased androgens, whereas the induction of ovaries may be a result of increased estrogen synthesis and secretion. The possible common mechanism underlying the abnormal sexual development in the current study and reproductive abnormalities in reptiles and mammals has significant implications for environmental and public health. The effects observed in mammals were dismissed as a concern for public health because the exposure levels were very high (20–26, 32–38). The effective doses in the current study, however, demonstrate the sensitivity of amphibians relative to other taxa, validate the use of amphibians as sensitive environmental monitors/sentinels, and raise real concern for amphibians in the wild. The effects on the gonads in the current study were produced at 0.1 ppb, which was more than 600 times lower than the dose required to induce aromatase in human adrenocortical carcinoma (25) and placental choriocarcinoma studies (25–26) and 30,000,000 times lower than the dose required to produce reproductive effects in rats (24).

Furthermore, the current data demonstrate the importance of considering endocrine-regulated endpoints in assessing the potential impact of pesticides on amphibians. Reported teratogenesis, growth inhibition, and mortality in amphibians in response to atrazine were not considered environmental concerns because of the high doses required to produce these effects (40). Effects in the current study, however, occurred at levels 10,000 times lower than the dose required to produce effects in amphibians in these previous studies (40–45). Allran and Karasov (14) reached the conclusion that atrazine was probably not a significant factor in amphibian declines based on their studies of toxicity, deformities, and effects on feeding and ventilation in leopard frogs that did not produce noticeable effects below 3 ppm. The current data show that negative effects on sex differentiation occur at doses 30,000 times lower than effective doses reported by Allran and Karasov. The Allran and Karasov study, however, examined a different species and different endpoints.

The current data raise new concerns for amphibians with regards to atrazine. Effective doses (0.1 ppb for the production of hermaphrodites and 1 ppb for reduction in laryngeal size) are ecologically relevant. The recommended application level of

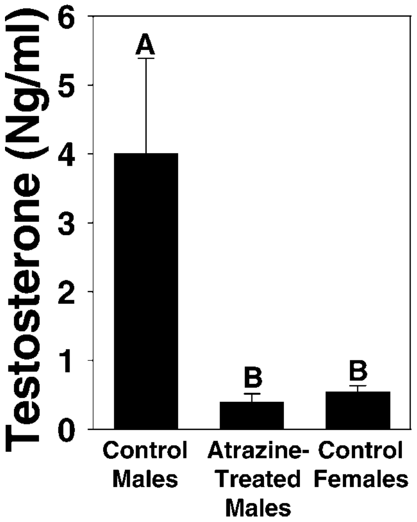

Fig. 4. Effect of 46-day exposure to atrazine on plasma testosterone levels in sexually mature male *X. laevis*. Sexually mature males were housed individually. Experimental animals were treated every 3 days with 25 ppb atrazine, and controls were treated identically except without atrazine exposure. Control females are shown for comparative purposes. Letters above bars show statistical groupings (ANOVA, $P < 0.05$).

atrazine ranges from 2,500,000–29,300,000 ppb (46), the allowable contaminant level for atrazine in drinking water is 3 ppb (39), and short-term exposures of 200 ppb are not considered a health risk. Atrazine can be as high as 21 ppb in ground water, 42 ppb in surface waters, 102 ppb in river basins in agricultural areas, up to 224 ppb in Midwestern streams, and up to 2,300 ppb in tailwater pits in Midwestern agricultural areas (47, 48). Atrazine can be found in excess of 1 ppb in precipitation in localities where it is not used and up to 40 ppb in rainfall in Midwestern agricultural areas (49–51). Further, Davidson *et al.* (52) recently reported that at least one species (*Rana aurora*) may be affected by aerial transport of agrichemicals. They showed that declines and extirpations of *R. aurora* populations were strongly correlated with areas that were downwind of agricultural activity. Furthermore, Cory *et al.* (53) showed that agrichemicals can be transported aerially and accumulated in amphibians' tissues. Thus, the likelihood that wild amphibians are exposed to 0.1 ppb or even 1 ppb atrazine is extremely high.

Furthermore, atrazine is typically applied when the soil is tilled, such that levels are highest during spring rainfall (13). This pattern of use puts amphibians at great risk, because the highest atrazine levels coincide with the breeding season for amphibians. Throughout areas where atrazine is used, atrazine levels peak while larval amphibians are at critical developmental stages. Also, depending on the species, amphibians breed in every possible freshwater microhabitat—from temporary pools, irrigation ditches, and flooded fields, to streams, rivers, lakes, and other permanent sources of water. The current data raise the question of the threat of atrazine, in particular, and of pesticides, in general, to amphibians in the wild. Low-dose endocrine-disrupting effects, which have not been addressed extensively in amphibians, are of special concern in this regard. If such effects do occur in the wild in other species, exposed animals could suffer impaired reproductive function. The described effects are all internal and may go unnoticed by researchers—unlike mortality and external malformations. Thus, exposed populations could decline and even go extinct without any recognition of the developmental effects on individuals. Already, it has been suggested that pesticides may play a role in amphibian declines (3, 52, 54, 55). Also, Reeder *et al.* (56) found that atrazine exposure may be associated with intersexual cricket frogs in the wild in the Illinois. Because the *P* value in the Reeder *et al.* study was 0.07 and because no laboratory data were available, they concluded that "[w]hether atrazine accounts for findings of intersexuality is less clear" (ref. 56, p. 265). We believe that the current data strongly suggest a connection between atrazine exposure and intersexuality. Combined with the decreases in dissolved oxygen, pH, and available food sources (phytoplankton, periphyton, and macrophytes) caused by atrazine (45), this common contaminant could be a contributing factor in amphibian declines. Ongoing investigations of the effects of atrazine on other species and amphibians in the wild will assess the realized role of this widespread compound in amphibian declines.

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Effectiveness of Road Ripping in Restoring Infiltration Capacity of Forest Roads[1]

Charles H. Luce[2]

Abstract: Many forest roads are being closed as a step in watershed restoration. Ripping roads with subsoilers or rock rippers is a common practice to increase the infiltration capacity of roads prior to closure. When considering the effectiveness of ripping for reducing runoff and erosion and the potential reduction in slope stability by saturating road fills, it is important to know how ripping changes the infiltration capacity of forest roads. Hydrographs from simulated rainfall on 1-m x 1-m plots were analyzed to find the saturated hydraulic conductivity, an indicator of infiltration capacity. I examined saturated hydraulic conductivity for three treatments on two different soils. One road was built in a soil derived from the metamorphic belt series geology of northern Idaho, a soil noted for its high rock fragment content. The second road was built in a sandy soil derived from decomposed granitics of the Idaho batholith. On each soil, five plots were installed on a road prior to ripping, and nine plots were installed on the same road segment following ripping, four covered with a heavy straw mulch and five without. Three half-hour rainfall events with intensities near 90 mm/hr were simulated on each plot. Results show that ripping increases hydraulic conductivities enough to reduce risk of runoff but does not restore the natural hydraulic conductivity of a forested slope. The unripped road surfaces had hydraulic conductivities in the range of 0-4 mm/hr, whereas ripped roads were in the range of 20-40 mm/hr after the second event. Surface sealing and tilled soil subsidence processes are important in reducing the hydraulic conductivity of the soils with repeated wetting. Subsidence appears to be important on the granitic soil, whereas surface sealing was more important on the belt series soil.

Key Words: road closure, infiltration, runoff

Introduction

Natural surface runoff in most forests is restricted to channels and nearby areas. Construction of forest roads substantially alters the hillslope hydrology by causing surface flow in areas far from established channels. Overland flow from forest roads can carry sediment eroded from the road surface, extend channel systems (Montgomery, 1994, Wemple, 1994), and increase the probability of landslides (Sidle et al., 1985). Watersheds with dense road networks commonly experience increased sedimentation and peak flows.

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To reduce watershed degradation by roads that are no longer needed, many roads are being closed and obliterated. Methods to carry this out vary from simple closure with an earth berm at the road entrance to complete recontouring of the surface. One of the more popular methods is ripping, where a bulldozer drags rock rippers or subsoilers through the road to break up the compacted layers forming the road. This method produces a result similar in appearance to plowing and is meant to enhance infiltration to reduce runoff and flow concentration. The loosened surface deters further vehicle access, and the improved seedbed enhances revegetation. Vegetation, in turn, maintains infiltration capacity, stabilizes the road prism, and protects against erosion.

Several researchers have examined the problem of surface soil compaction in reclaiming roads, mined lands, and degraded rangeland. Some of the early research on roads examined plant densities following various treatments. Kidd and Haupt (1968) examined effectiveness of scarification to a 12-inch depth and other factors on growth of grass species on recently closed logging roads. Scarification followed by seeding increased the number of surviving plants per unit area but did not change the percentage ground cover relative to no scarification.

Ripping and related activities are an important part of reclaiming mined lands, and there is substantial literature on the effectiveness of various treatments for mine reclamation. Most studies have examined the effectiveness of soil amendments and plant selection in achieving appropriate plant cover (see for example Farmer et al., 1974). Ripping is considered so fundamental that few studies have addressed it directly. Verma and Thames (1978) point out that deep chiseling (a specific tillage technique similar to ripping but shallower) is effective in preventing runoff and erosion on relatively flat slopes. They also point out that the effect is temporary, usually less than one year, and that chiseling must be used in conjunction with other treatments.

Gifford (1975) reviewed a few studies on the effectiveness of ripping in decompacting rangeland soils. The articles reviewed there showed that deep ripping could greatly decrease runoff from natural events, while shallow ripping with little surface disturbance had little effect. The papers reviewed also suggested that effectiveness of ripping treatments on rangelands decreases over time.

Agricultural operations and the reactions of soil to tillage have been under scrutiny for centuries. Two processes examined by contemporary researchers, soil crusting (also called surface sealing) by rainfall and tilled soil subsidence, are important to freshly tilled mineral road soils. Among authors who have measured the development of crusts on soils during rainfall are Mohammed and Kohl (1987), Loch and Foley (1994), Sharma et al. (1981), Jennings et al. (1987), and Bosch and Onstad (1988). High-energy raindrop impact drives the process by breaking apart aggregates on the surface and redistributing the fine particles to fill pores, thereby reducing the saturated hydraulic conductivity of the surface layer. This process seems most important in soils with enough clay content to form aggregates that are otherwise water stable. To study the effects of soil settlement on four freshly tilled soils, Onstad et al. (1984) examined bulk density, saturated hydraulic conductivity, and random roughness height while applying 15.2 cm of water with no raindrop impact energy (using furnace filters). They found that dry bulk density rose quickly, and saturated hydraulic conductivity dropped quickly, as water was added. Soil settlement is a well-known problem in regions of silty and sandy soils because of their low cohesion under saturated conditions.

Tilled agricultural soils typically have saturated hydraulic conductivities in the range of 2-30 mm/hr, whereas mildly disturbed forest soils (bare mineral soil, no compaction) have conductivities in the range of 60-80 mm/hr (Luce, 1995). These figures suggest that ripping may not entirely restore the hillslope hydrology. Roads generally have saturated hydraulic conductivities less than 4 mm/hr, so a tilled soil still represents an improvement.

Given these insights, one must seriously question the degree of hydrologic recovery provided by ripping. If the purpose of the ripping is, in part, to prevent surface runoff, it must increase the infiltration capacity of the soil. Infiltration capacity at a particular time is a function of soil properties and the soil moisture content at that time. Saturated hydraulic conductivity is a reasonable standard for comparing infiltration capacity among soils or treatments, as it is independent of soil moisture and represents the infiltration capacity of a soil near saturation. This study examined the response of saturated hydraulic conductivities of roads to ripping.

Methods

Saturated hydraulic conductivities were measured on an unripped road, a ripped road, and a ripped road with a heavy straw mulch application, on two soils during three sequential simulated rainfall events. The three sequential simulated rainfall events were used to determine hydraulic conductivity changes with added water and rainsplash. The first soil was derived from metasedimentary belt series parent materials. Belt series soils have high rock fragment content and high fine content (Figure 1). Plasticity indexes for road soils in the belt series soils range from 5-10%, and clay content ranges from 18-25% of the fraction finer than 4 mm (unpublished data, Idaho Panhandle National Forests). The second soil was derived from Idaho batholith granitics. Most of the road material came from alluvial deposits of the South Fork of the Salmon River. These materials were sandy with some rock fragments and low fines (Figure 1). Plasticity indexes for road soils on the South Fork Salmon River road range from 0-10% (unpublished data, Payette National Forest). The mulch was added to reduce the raindrop kinetic energy impact important to the surface sealing processes and to determine the result of wetting only on soil consolidation. This provided a control to see whether surface sealing contributed to a decline in saturated hydraulic conductivity following ripping.

Ripping on the belt series road was accomplished using a Caterpillar D-9 bulldozer with three 4-foot ripping teeth spaced 2.5 feet apart. Small, 8-inch “wings” were welded to the

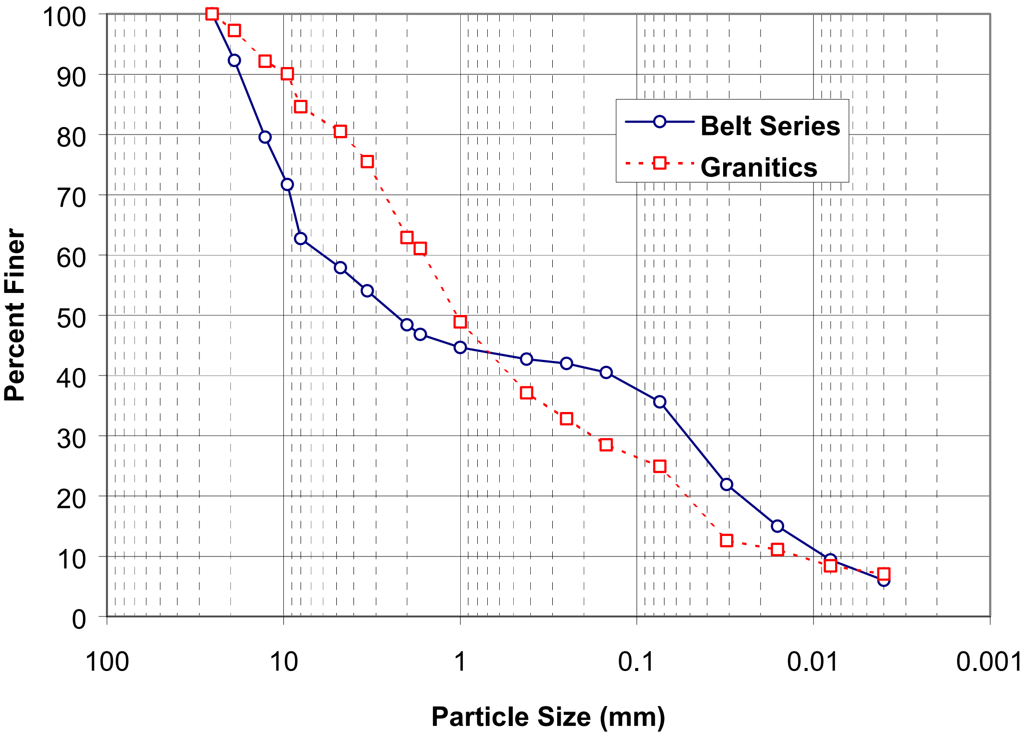

Figure 1: Particle size distribution for the two soils for all particles smaller than 20 mm. Steeper parts of the graph indicated areas of high relative frequency. The belt series soil shows a bimodal distribution, peaking in gravels (> 2 mm) and silts (< 0.05 mm). The granitics are sandier, peaking in the coarse sand to fine gravel region (< 3 mm and > 0.5 mm).

bars about 1 foot from the tip to promote fracture in the soil. The resulting ripped soil was well fractured and turned over to a depth of roughly three feet. On the granitics, a large Gallion road construction grader with two-foot ripping teeth with two foot spacing was used. The resulting ripped soil had large, flat clods, between 3 and 4 inches thick and 8-14 inches wide jumbled in a well tilled sand. The clods were formed from the original road surface, and the material underneath fractured more completely. The depth of tilled material was between 2 and 2.5 feet. Saturation overland flow was not observed from the rainfall simulations used here, so the lack of depth did not affect the results printed here. Mulch was added such that the soil underneath was not visible, a loading much greater than typically applied for erosion control.

The hypothesis was that the belt series soil would exhibit a modest increase in hydraulic conductivity with ripping; that the protection of the rock fragments would yield only a minor decrease over the course of the rainfall due to surface sealing; and that the matrix of cohesive fines would prevent tilled soil settling. It was further expected that the granitic soil would increase modestly in hydraulic conductivity with initial ripping; decrease slightly with increased rainfall due to settling, and show no difference in decrease when mulch was added.

For each treatment, 1-m by 1-m plots were constructed with sheet metal boundaries and a trough at the downslope end to collect runoff. On each soil, five plots were installed on a road prior to ripping, and nine plots were installed on the same road segment following ripping, four covered with a heavy straw mulch and five without.

A modified Purdue rainfall simulator provided rainfall for the sprinkling infiltrometer plots. The rainfall simulator oscillates a downward-pointed irrigation sprinkler nozzle through a small arc to cover the plot and immediately surrounding area with spatially uniform rainfall. Measurements of rainfall energy under a similar simulator (Foltz et al., 1995) suggest that the rainfall kinetic energy was about half that of natural rainfall. However, Mohammed and Kohl (1987) successfully used a similar design and nozzle to observe surface sealing on agricultural soils. Rainfall was applied at approximately 90 mm/hr to each plot during three 30-minute rainfall events. The first event was conducted under existing soil moisture conditions, the second event was carried out roughly 20 hours later, and the third event was typically started within 45 minutes of completing the second event. The high rainfall intensity and short interval between storms are not meant to simulate potential storm occurrences but are used to find the average saturated hydraulic conductivity of the plot. Sprinkling infiltrometers require that the precipitation intensity exceed the highest saturated hydraulic conductivity on the plot. Past experiences with forest soils and the potential of a ripped road to meet these hydraulic conductivities suggested that 90 mm/hr would be appropriate. Actual rainfall intensity for each event was measured at the beginning and end of each event using a sheet metal plot cover. Timed runoff samples were collected in 1,000-ml bottles.

Runoff hydrographs were analyzed by the method of Luce and Cundy (1994) to find infiltration parameters for Philip's (1969) equation, including saturated hydraulic conductivity and sorptivity. The method is essentially a curve fitting procedure for a kinematic wave model of Hortonian overland flow (Cundy and Tento, 1986; Luce and Cundy, 1992). The curve fitting procedure uses a genetic algorithm, which robustly finds the optimum fit for three variables simultaneously and makes it possible to detect changes in saturated hydraulic conductivity independent of routing effects and changes in moisture content and depression storage that occur over time.

Bulk density and moisture content were measured with a calibrated nuclear densiometer. Between two and four measurements were taken on the surface within or adjacent to each plot before and after each rainfall event.

The treatments, plots, soils, and events constitute a full-factorial ANOVA design. There were three treatments, two soils, with three events as repeated measures and five replicates (plots) for each cell in the design. Analysis consisted primarily of planned comparisons within this ANOVA design.

Results and Discussion

Hydraulic conductivities for all soils, treatments, and events are summarized in Figure 2.

Figure 3 shows the variation within the treatment for each soil. Prior to ripping, the roads were nearly impervious. Saturated hydraulic conductivities for the roads fell in the range of 0-12 mm/hr (Figure 3). These values agree well with other observations (Reid, 1981; Luce and Cundy, 1994).

The saturated hydraulic conductivity of a ripped road following three rainfall events was significantly greater than that of the road surface prior to ripping ($p(K_p=K_{r3}) = 0.005$ for granitics, and $p(K_p=K_{r3}) < 10^{-6}$ for the belt series). Results varied greatly from plot to plot, but most saturated hydraulic conductivities after the third rainfall event on a ripped road were in the range of 22-35 mm/hr for the belt series and 7-25 mm/hr for the granitics. These conductivities are modest compared to the saturated hydraulic conductivity of a lightly disturbed forest soil of 60-80 mm/hr (Luce, 1995). The increase in conductivity probably represents significant gains in terms of reducing runoff, however. For example, snowmelt, which was observed ponding on the South Fork road prior to ripping, would most likely infiltrate with the road in a ripped condition because

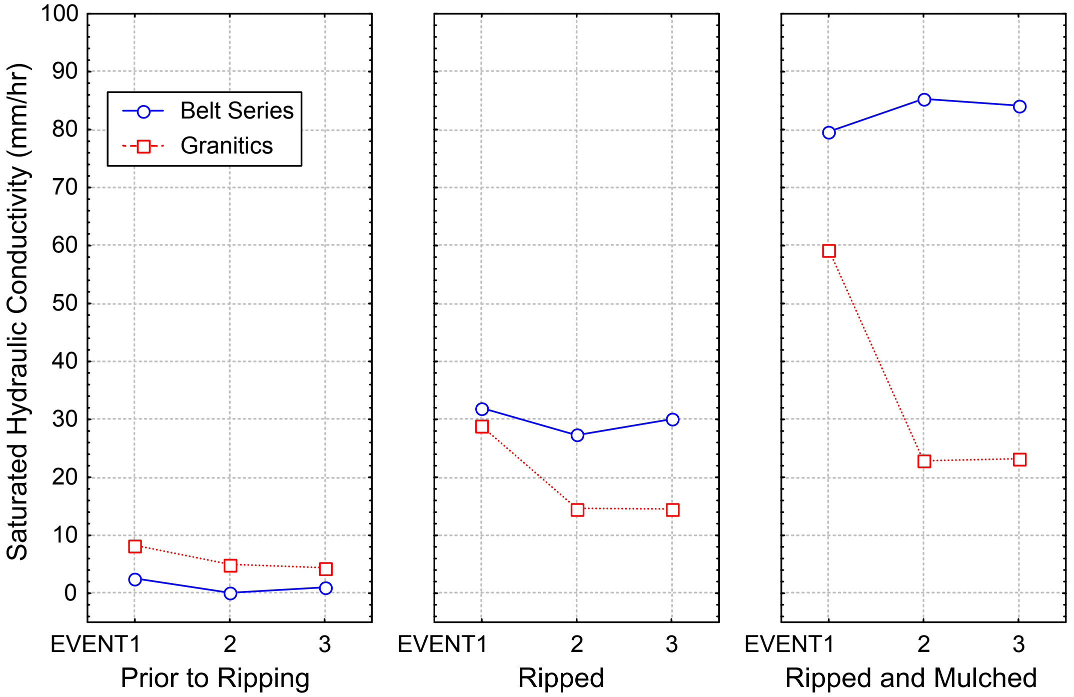

Figure 2: Average hydraulic conductivities by treatment and rainfall event. Prior to ripping there is no statistical difference between the granitics and belt series. The increase in hydraulic conductivity following ripping was statistically significant and significant relative to probable rainfall. Mulching following ripping protected the belt series soil from surface sealing, but did not prevent the collapse of the granitic soil. Following collapse, the differences between the ripped and ripped and mulched granitic soil are not statistically significant.

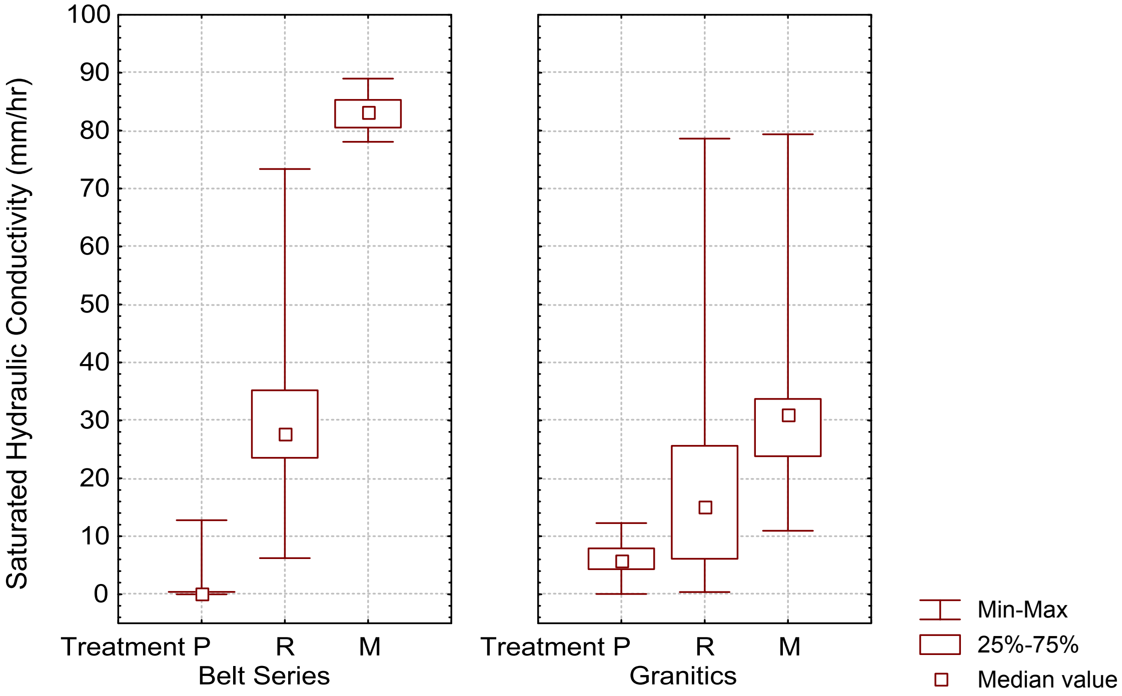

Figure 3: Distribution of saturated hydraulic conductivities by treatment across all events. P = prior to ripping, R = ripped, M = ripped and mulched.

snowmelt rates seldom exceed 15 mm/hr. Snowmelt rates measured during the peak snowmelt season of 1986 at the Central Sierra Snow Lab never exceeded 4 mm/hr (Tarboton et al., 1995). Precipitation-duration-frequency information for northern and central Idaho show that the 1-hr 100-yr event is between 25 and 33 mm/hr (Miller et al., 1973). This indicates that ripping provides some protection for rare events as well.

Hydraulic conductivity values for the ripped treatment on the granitic soil decreased about 50% with added rainfall ($p(K_1=K_2) = 0.00015$). This corresponded to field observations of soil settlement and large clods of soil created by the fracture of the road surface dissolving under the rainfall. Figure 4 shows bulk density responses to treatments and rainfall, including the large increase in bulk density with the first rainfall event on the granitics ($p(\rho_{rg0} = \rho_{rg45}) = 0.000094$). Both results compare well with those of Onstad et al. (1984).

While not evident in Figure 2 or the statistics, the saturated hydraulic conductivity of the ripped belt series soils also dropped from its initial value. Initially, and for much of the first event, the ripped plots on the belt series soil showed no runoff. During these periods, runoff from higher areas flowed to low areas and into macropores. On some plots, runoff from nearly the entire plot could be seen draining into a single macropore for short periods. The macropores were formed during the ripping by fracturing of large, weak, brittle boulders. Trenching revealed that the ripping process had changed the soil

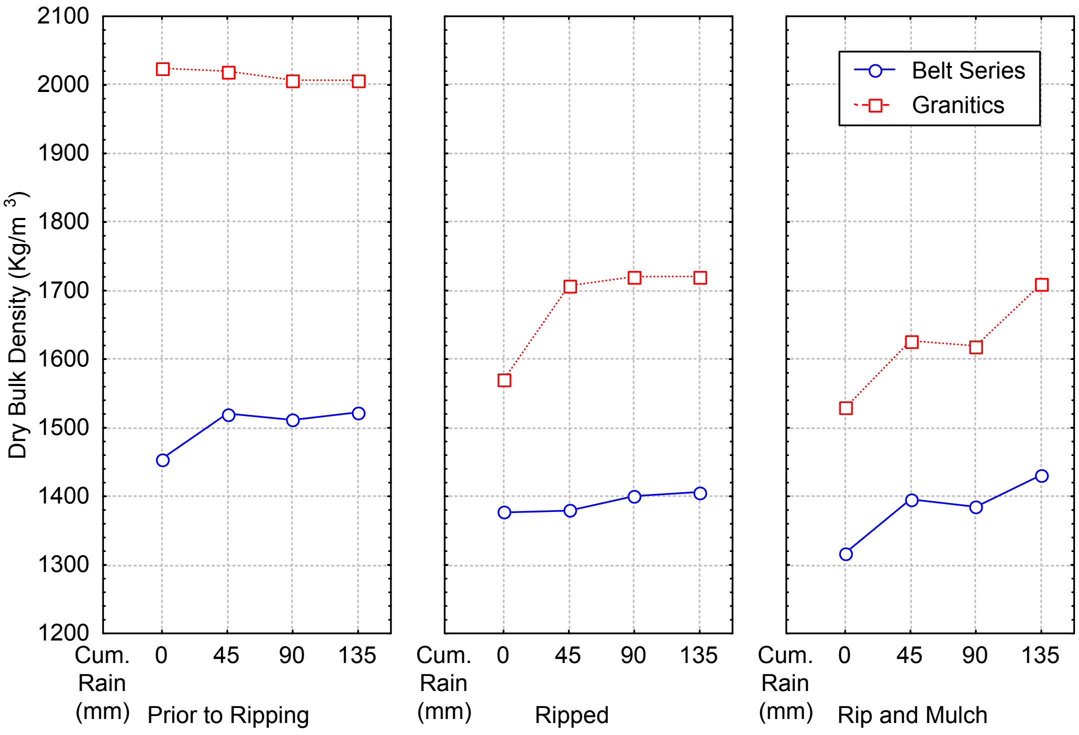

Figure 4: Dry bulk density of soil near the surface by treatment and cumulative rainfall applied to the plot. Density is statistically constant prior to ripping on both soils. Following ripping the belt series maintained a constant density, while the density of the granitics rose after the first event. After mulching the density increase of the granitics was again statistically significant, and the increase in density for the mulched belt series soils was only marginally significant.

from a matrix and clast-supported fabric to a partly open work fabric (Selby, 1993) leaving a few large voids. Erosion of fine sediment and small gravel eventually clogged these macropores. Most macropores clogged within the first 30-minute rainfall. This is one process described by Mohammed and Kohl (1987) as important in surface sealing. Because of the nearly binary response of the plot runoff to macropores at this scale, the fitting algorithm interpreted this process as a high depression storage and fit the hydraulic conductivity to the final 10-15 minutes of high flow on the hydrograph. The hydraulic conductivity during the last 10-15 minutes differed little from the hydraulic conductivity for the next rainfall event. Comparison of the amount of rainfall applied in 15 minutes, about 0.04 m to the porosity times the ripped depth, 0.40 m, suggests that saturation overland flow was not observed, and that the response was due to a change in the infiltrating surface. Saturation overland flow would yield runoff rates close to the rainfall rate; this situation was not recorded.

Examination of what happens when the mulch is applied supports these observations. The straw mulch absorbs the kinetic energy of the raindrops and prevents splash erosion. Under the mulch one would expect little transport of sediment and little surface sealing, but soil settlement should be the same as under unprotected conditions. The effect on hydraulic conductivity is striking. The belt series soil responded by maintaining a high hydraulic conductivity through all three events ($p(K_1 = K_2 = K_3) = 0.51$). The hydraulic conductivities for the granitic soil dropped to values similar to those of the ripped condition without mulch for the second and third events ($p(K_{r(2,3)} = K_{m(2,3)}) = 0.203$). The bulk density of the granitics under a heavy mulch increased significantly ($p(\rho_{mg0} = \rho_{mg45}) = 0.03$), much as it did without the mulch. A less significant rise in bulk density also occurred in the belt series soil ($p(\rho_{mb0} = \rho_{mb45}) = 0.07$), but it is not clear why. Only minor settling was observed in the field.

Although it was not quantified, water flowing from the mulched plots was visibly cleaner than that flowing from the ripped plots. This observation fits with conclusions of many other studies (e.g., Burroughs and King, 1989) that reduction of the rainfall impact reduces erosion.

Little information exists on the durability of infiltration increases beyond these few initial rainstorms. Gifford (1975) reviewed several studies where the effect of ripping compacted rangelands decreased over a period of years, and similar behavior would be expected for roads. Anecdotal observations of roads ripped in earlier years revealed that after one winter, the surfaces were nearly as solid and dense as the original road surfaces. Near the South Fork Salmon River plots, dry bulk densities of a road ripped one year earlier were similar to the final (after third event) densities measured on the ripped and mulched plots. At this site, tree planters had difficulty inserting hoedads, normally an effective instrument, to dig small holes. Hand watering was necessary to keep the trees alive because of the low infiltration capacity and porosity.

Where a contractor had inadvertently incorporated some of the organic layer from the surrounding forest soil during the ripping operation, the ripped road retained its looseness. In mining and rangeland rehabilitation, endeavors similar to road rehabilitation, soil amendments are commonly used to increase soil organic content. Several studies (Skujins and Richardson, 1984; Hudson, 1994; Page-Dumroese et al., 1990; Sidle et al., 1993; Aguilar, 1992) highlight the importance of organic matter content for soil productivity, structure, and erosion protection. In those studies, organic matter was amended as topsoil, sludge, or surface mulch that later decomposed. Direct incorporation of composts may be necessary to prevent tilled soil settlement.

Conclusions and Recommendations

These results support the hypothesis that both soils increased in hydraulic conductivity immediately following ripping. I hypothesized that the belt series soil would retain most of this initial increase, whereas the granitic soil would lose hydraulic conductivity over time. However, the hypothesis that the rock fragments in the belt series soil would prevent surface sealing was not supported. Fines eroded from between fragments were sufficient to clog macropores. The combination of cohesive fines and large voids supported by fragmented clasts yielded little soil settlement. On the granitics, the hypothesized soil settling was observed under both mulched and unmulched conditions, which led to a decrease in hydraulic conductivity following the initial increase, as expected.

Ecological restoration of forest roads and watersheds requires improved vegetative cover and improved infiltration for forest road surfaces. These findings suggest that ripping can be a reasonably effective step in the restoration process. Even considering the effects of settling and surface sealing, ripping increases hydraulic conductivities modestly -- enough, perhaps, to prevent runoff and erosion from most rainfall and snowmelt events. These increases do not represent “hydrologic recovery” for the treated areas, however, and a risk of erosion and concentration of water into unstable areas still exists. These continuing risks must be considered in the design of the restoration project so that runoff does not drain to streams or unstable hillslopes. While the roughness of the seedbed is increased and traffic reduced, the increases in porosity are slight enough that only very hardy plants may initially take advantage of the improved surface.

The findings on soil settlement and surface sealing highlight the fact that freshly tilled road soils are sterile and poorly structured. Ripping and subsoiling alone provide only temporary and marginal improvements. Amended organic matter would likely enhance both the short-term effectiveness and durability of gains in porosity and infiltration capacity, greatly accelerating restoration of the road’s hydrologic and ecological function.

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Research Article

Silviculture

Composition and Structure of Reproduction in Group Selection Openings after 20 Years in a Southern Appalachian Mixed-Hardwood Forest

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Abstract

Following harvests by even-aged methods in mixed-hardwood forests, desirable oak (*Quercus* spp. L.) reproduction can be quickly overtopped by shade-intolerant pioneer species. In a long-term, operational-scale study of uneven-aged management by group selection, we inventoried 10- and 20-year-old reproduction following two harvest entries in a mature, dry-mesic southern Appalachian upland hardwood stand. Our study objectives included evaluating the species composition and structure of reproduction in relation to opening size (0.05 ha to 0.41 ha), position in openings (center or edge), uneven-aged diameter structure, and evidence supporting the delayed oak dominance hypothesis. Opening size had no effect on reproduction from either entry. After 10 years (but not 20) yellow-poplar (*Liriodendron tulipifera* L.) stem density and basal area were greater in opening centers; oaks were greater at edges. Position did not affect red maple (*Acer rubrum* L.) or other tolerant species. Diversity significantly increased between 10 and 20 years, suggesting evidence supporting the delayed oak dominance hypothesis. Early results from this study suggest that small openings can be used to regenerate desirable midtolerant and intolerant species in Appalachian mixed-hardwood stands on intermediate quality sites where uneven-aged stand structure is important for timber management and other goals such as visual appearance and early successional habitat.

Study Implications: Periodic harvests of small groups (0.05 to 0.41 ha) of mature trees in dry-mesic Appalachian oak-dominated, mixed-hardwood stands, followed by site preparation and competition release treatments to control undesirable shade tolerant species, can result in openings stocked with desirable shade midtolerant and intolerant reproduction after 20 years. Centers of openings will be dominated by yellow-poplar, but oak reproduction can be most prevalent around the periphery. Group selection is a flexible method of uneven-aged management that can be used to meet regeneration objectives and related goals such as early successional habitat and visual quality.

Keywords: dominance hypothesis, group selection, opening size, *Quercus*, self-thinning

Gap-based partial cutting practices have been proposed as an appropriate silvicultural system to address multiple management objectives of visual quality, habitat, regeneration, resiliency, biodiversity, and sustainability of ecosystems in many forest types in North America (Coates and Burton 1997), which includes the Central Hardwood Region (CHR) of the eastern United States. There, oaks are both characteristic and keystone canopy species with high economic and habitat value in the mixed-hardwood forests (Fralish 2004). However, recruitment and development of midtolerant oak reproduction following harvests has decreased in the CHR likely because of altered historical disturbance regimes during the 1900s (such as suppression of wildland fire) that have favored tolerant species (Abrams 2003, Nowacki and Abrams 2008) and stand regeneration by clearcutting that favors fast-growing intolerant species, such as sweetgum (*Liquidambar styraciflua* L.) (Johnsen and Krinard 1988) or yellow-poplar (Brashears et al. 2004). The regeneration ecology of oaks is well known (Johnson et al. 2002); advance oak reproduction must develop a large root system before meaningful height growth is initiated and maintained (Sander 1971). The multilayered forests of the southern Appalachians section of the CHR consist of nearly 100 predominantly cold-deciduous hardwood species that can be classified in two groups: desirable, which are large, shade intolerant canopy species that are well formed and can have high commercial and/or wildlife habitat value, or undesirable, which are typically small, shade tolerant species that are poorly formed and/or have little commercial value, but may provide soft mast for wildlife (Della-Bianca and Beck 1985). Depending on available light resources admitted through canopy openings to the forest floor, advanced reproduction of desirable oaks cycles through alternating phases of growth, top-dieback, and basal resprouting (Runkle and Yetter 1987). The combination of small canopy openings resulting from death of single trees, combined with the middle canopy of shade tolerant species, hinders development and growth of desired advance reproduction of intolerant species into the canopy on dry-mesic sites (Della-Bianca and Beck 1985).

It has long been known that small canopy openings resulting from single-tree selection favors regeneration by shade tolerant species in the southern Appalachians (Frothingham 1931, Della-Bianca and Beck 1985). Frothingham (1931),

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however, first suggested that larger openings resulting from group selection harvests could favor intolerant species. Group selection has been studied extensively in central and northern parts of the CHR and elsewhere to investigate effects of gap size, orientation, substrate, and age on composition and structure on reproduction of oaks and other desired species (Zhu et al. 2014, Kern et al. 2017). Studies of group selection have not been conducted in the southern Appalachians, however, which has environmental conditions and species composition different than elsewhere in the CHR (Cook et al. 1998).

Group selection research has shown that small openings (<0.05 ha) promote shade tolerant species and large openings (>0.40 ha) favor intolerant species; oak reproduction, which is semitolerant of shade as juveniles, occurs most frequently in intermediate-sized openings (Lhotka 2013). In intermediate and large openings, rapidly growing pioneer species present the primary source of competition to slowly growing advance oak reproduction following harvests in much of the CHR (Jenkins and Parker 1998, Lhotka 2013). Oliver (1978) observed that second- and third-decade self-thinning in pioneer-dominated stands can allow emergence of slower growing, intermediate oaks as codominants, and proposed the delayed oak dominance hypothesis. The hypothesis has been specifically tested by Steiner et al. (2018) and observed from inventory data in bottomland (Johnsen and Krinard 1988) and upland stands (Hilt 1985), but little studied elsewhere.

The purpose of this article is to summarize 10- and 20-year responses of reproduction resulting from two entries in a 1988 study of group selection harvests in a southern Appalachian mixed-hardwood forest. Objectives of this study were strongly influenced by results from a long-term study of single-tree selection in an adjacent stand showing lack of desirable reproduction of mid and intolerant species (Della-Bianca and Beck 1985) and favorable results from a group selection study in northern hardwoods (Leak and Filip 1977). Initial objectives of this study were associated primarily with questions regarding uneven-aged management, particularly in relation to arguments by Roach (1974) concerning application and sustainability of the group selection method in hardwood forests: (1) Will group selection result in a balanced (i.e., reverse J-shaped curve) stand structure? (2) What size opening is needed for regenerating intolerant species, particularly oaks? Two additional study objectives were included during the second harvest entry: (3) Does position in openings affect composition and structure of reproduction? (4) Is evidence available in support of the delayed oak dominance hypothesis?

Methods

Site and Vegetation Description

The Frothingham group selection study was installed in the Boyd Branch watershed of the Bent Creek Experimental Forest, a special use area in the Pisgah Ranger District of the Pisgah National Forest, in western North Carolina (35.489°N, –82.645°W) (Figure 1). The subtropical/continental climate consists of short, mild winters (mean January temperature: –14.0°C) and warm, humid summers (mean July temperature: 22.3°C). Precipitation averages 1,220 mm annually and is distributed uniformly among seasons; snowfall averages 280 mm annually. The 29.9 ha study area occupies a low-elevation (720–850 m) shallow, east-facing cove with soil moisture regimes ranging from submesic to subxeric. Soils are deep (>100 cm), predominantly Ultisols (Evard, Cowee, Toecane series) in residuum on gentle slopes and Inceptisols (Tusquitee series) in colluvium on steep slopes and in narrow bands adjacent to perennial streams. The study area is part of a larger tract previously used for subsistence farming by the Boyd family during the mid to late 1800s (Nesbitt 1941). Although none of the study area had been cultivated, it was likely an open-canopy woodlot with an herbaceous layer of native grasses and tree and shrub basal sprouts, which was burned periodically to promote browse for grazing by livestock. Except for the regional demise of American chestnut (*Castanea dentata* [Marsh.] Borkh.) in the 1920s, resulting from a blight fungus (*Cryphonectria parasitica* [Murrill] M.E. Barr.), there are no records of other human-related disturbances, such as commercial timber harvests. Predominant vegetation on middle and lower slopes consists of a high canopy (30 m) of merchantable mixed hardwoods primarily of midtolerant to intolerant oaks (black [*Q. velutina* Lam.], chestnut [*Q. montana* L.], scarlet [*Q. coccinea* Muenchh.], white [*Q. alba* L.], and northern red [*Q. rubra* L.]) and hickories (mockernut [*Carya tomentosa* Sarg.] and pignut [*C. glabra* Miller]); intolerant yellow-poplar occurs primarily on mesic soils along streams and also as scattered individuals throughout. Middle and lower canopies are dominated by small, shade tolerant, generally unmerchantable mixed hardwoods including red maple, sourwood (*Oxydendrum arboretum* L.), flowering dogwood (*Cornus florida* L.), blackgum (*Nyssa sylvatica* Marsh.), and witch hazel (*Hamamelis virginiana* L.). Common tall (>2 m) evergreen shrubs include rosebay rhododendron (*Rhododendron maximum* L.) on moist sites and mountain laurel (*Kalmia latifolia* L.) on dry sites.

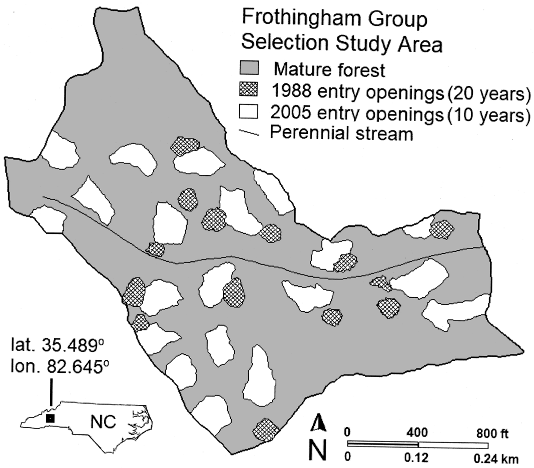

Figure 1. The Frothingham group selection study area showing openings for the first entry (1988) and the second entry (2005). (Map created by H. McNab using MicroSoft Paint software.)

Preharvest Stand Sampling

In winter 1988, advance reproduction (height < 1.37 m) and small trees (diameter breast height (dbh) ≥ 1 cm to 15.2 cm) were inventoried by species and diameter on 0.004 ha sample plots systematically distributed throughout the stand at the rate of 2.5 plots per ha. All large trees (>15.2 cm dbh) were inventoried individually by species and dbh in a total cruise of the study area. Tree diameters ranged to a maximum of 89 cm and approximated the balanced distribution

Table 1. Species subgroup, species group (Spec. Group), shade tolerance class (Tol. Class), reproduction density (Repro. Density, height <1.37 m), tree stem density (≥1 cm diameter breast height), basal area (BA), quadratic mean diameter breast height (dbhq), and importance value (IV, relative stem density + relative basal area) by species subgroups and overall subgroups of the mature, mixed-hardwood stand occupying the study area before the first harvest entry in 1988.

Species Subgroup	Spec. Group	Tol. Class	Repro. Density	Tree Density (n/ha)			BA m²/ha	dbhq (cm)	IV 200
Tree Species				1–10 cm	>10 cm	Sum			
Red oak									
Quercus spp.*	D	M	2,406	6.5	29.1	35.6	2.50	29.9	10.3
Northern red oak	D	M	8,190	6.5	3.9	10.4	0.38	21.7	1.8
Overall species	D	-	10,596	13.0	33.0	46.0	2.88	28.2	12.2
White oak									
Chestnut oak	D	M	2,566	26.0	58.8	84.8	5.54	28.8	23.2
White oak	D	M	5,898	6.5	21.1	27.6	2.21	32.0	9.0
Overall species	D	-	8,464	32.5	79.9	112.4	7.75	29.6	32.2
Yellow-poplar									
Yellow-poplar	D	I	2,097	26.0	133.8	159.8	9.00	26.8	38.9
Other desirable									
Sweet birch	D	I	291	110.5	21.1	131.6	0.76	8.5	9.3
Black locust	D	I	944	13.0	18.6	31.6	0.65	16.2	3.8
Hickory spp.*	D	M	504	32.5	5.4	37.9	0.27	9.5	2.9
Miscel. species (1)*	D	I	17	6.5	1.5	8.0	0.10	12.4	0.7
Overall species	D	-	1,756	162.5	46.6	209.1	1.78	10.4	16.8
Red maple									
Red maple	U	T	65,924	273.0	88.0	361.0	3.48	11.1	30.4
Other undesirable									
Miscel. species (2)*	U	T	6,355	728.0	95.2	823.2	4.19	8.1	56.5
Witch hazel	U	T	944	169.0	0	169.0	0.16	3.5	9.2
Miscel. species (3)*	U	T	3	32.5	0	32.5	0.04	4.1	1.8
Blackgum	U	T	422	19.5	2.4	21.9	0.06	5.7	1.3
Serviceberry	U	T	553	13.0	0	13.0	0.01	1.5	0.7
Overall species	U	-	8,277	962.0	97.6	1059.6	4.45	7.3	69.6
Overall groups	-	-	97,115	1469.0	478.9	1947.9	29.34	13.8	200.0

Species group: D, desirable; U, undesirable. Tol. class, shade tolerance class: I, intolerant; M, midtolerant; T, tolerant. **Quercus* spp.: scarlet oak (*Q. coccinea*) and black oak (*Q. velutina*).
*Hickory spp.: mockernut (*Carya tomentosa*) and pignut (*C. glabra*).
*Miscellaneous species (1): white ash (*Fraxinus americana*), black cherry (*Prunus serotina*), and shortleaf pine (*Pinus echinata*) combined. Miscellaneous species (2): flowering dogwood (*Cornus florida*), sourwood (*Oxydendrum arboreum*), and sassafras (*Sassafras albidum*). Miscellaneous species (3): American beech (*Fagus grandifolia*) and American holly (*Ilex opaca*).
Other undesirable: witch hazel (*Hamamelis virginiana*), blackgum (*Nyssa sylvatica*), and serviceberry (*Amelanchier arborea*).

of an uneven-aged stand, although mixed-species stands of eastern hardwoods are typically even-aged because of their stratified structure associated with shade tolerance (Oliver 1980). The prestudy inventory of advance reproduction and trees in the study area revealed a mean total basal area of 29.3 m^2/ha distributed among 20 species (Table 1). Five species of oaks (listed below) and yellow-poplar characterized the overstory. The small component of shortleaf pine (*Pinus echinata* Mill.), a conifer intolerant of shade and requiring exposed soil for best seed germination, is evidence of past land use for subsistence agriculture before this tract was acquired for inclusion in the national forest, around 1920. This species composition of mixed-hardwoods and scattered conifers corresponds well with the extensively occurring Southern Appalachian Oak Forest terrestrial ecosystem CES202.886, which occurs on mountainous landscapes from central Virginia to south-western North Carolina (NatureServe 2020). In 1988, mean age of dominant and codominant red oaks was 51 years (n = 8, range 33–86 years) and 69 years (n = 27, range 21–155 years) for white oaks. Oak site index averages approximately 24 m (range 22–27 m, 50-year base). Culmination of mean annual volume growth of dry-mesic hardwood stands averages 3.04 m^3/ha/year at approximately 90 years (USDA Forest Service 1994). Except for a narrow band of mixed mesophytic hardwoods along a perennial stream and on xeric sites along a ridge crest, the study area was relatively uniform in species composition and structure of vegetation was treated as a single stand.

Timber Harvests and Herbicide Site Preparation

The Frothingham study was established in 1988 to investigate group selection as a corollary to an adjacent long-term study of single tree selection (Della-Bianca and Beck 1985). Briefly, the study design specified management by area control with openings ranging in size from 0.10 ha to 0.40 ha. Openings were widely distributed on sites with acceptable regeneration potential, damaged canopy trees or poor stocking of mature trees; rotation length is 60 to 80 years and entry cycle is 10 to 15 years. Two entries have been made, in 1988 and 2005, when 14 and 24 openings were harvested, respectively. The 14 openings harvested in 1988 averaged 0.10 ha (range 0.05–0.14 ha); size of the 24 openings harvested in the 2005 entry averaged 0.29 ha and ranged from 0.13 ha to 0.41 ha. For both entries, all merchantable pulp wood and saw timber was cut using chainsaws and skidded to landings using either a farm tractor (1988 entry) or forestry-type wheeled skidder (2005).

Site preparation consisted of herbicide and hand tools treatment using cut stump and basal spray applications. Cut stumps were sprayed with a 50:50 ratio of triclopyr amine and water. Streamline basal spray was made with triclopyr ester mixed with mineral oil and bark penetrant additive at a 20% mixture. All residual trees between 2.54 and 20 cm dbh were cut. The cut surface of stumps of red maple, sourwood, blackgum, yellow-poplar, mountain laurel, rhododendron, vines (primarily oriental bittersweet [*Celastrus orbiculatus* Thunb.]), and exotic invasive species were sprayed with herbicide. Herbicide treatment of cut stumps was a standard method of site preparation used by national forests and necessary to control aggressive basal sprouting of tolerant species, which can quickly occupy a large proportion of growing space after harvests (Smith 1981, Fei and Steiner 2009). Within three growing seasons after harvest selected desirable species (primarily oaks) were released from competition using streamline basal spray application of 20% triclopyr ester herbicide mixed with mineral oil and an added compound consisting of wetting agent, sticker, and bark penetrant. Release treatments targeted sprout clumps of the species specified for site preparation. Site preparation and release treatments were applied by a contractor; field data were not collected on stems treated in the harvested openings.

Sample Plot Design and Data

Reproduction was inventoried once for each of the two entries: at 10 years after the 2005 entry and 20 years after the 1988 entry. Two sample plots (circular 0.004 ha, 3.6 m radius) were established at two locations (hereafter positions) in each opening: the center and edge. In a random direction from the center, each edge sample plot was placed so its boundary coincided with the boundary of the harvested opening. Edge plots were excluded and reselected that were adjacent to a logging road or that varied markedly in site conditions from the center plot, such as landform differences. Edge sample plots in the 2005 openings were not excluded if they were adjacent to an opening harvested in the 1988 entry. All tree reproduction ≥0.5 cm dbh was inventoried by species and 2.5 cm dbh classes in openings of each entry. Total height and dbh of several yellow-poplar trees in the center and edge sample plots of each opening were measured for development of a model for predicting total height of individual trees from inventory data. We selected yellow-poplar for height modeling because it is the predominant pioneer species on mesic and dry-mesic sites in the southern Appalachians and as such is the primary competitor of oaks and intolerant species (Beck and Hooper 1986). Yellow-poplar grows rapidly in height soon after seed germination and provides a measure of size that oaks must attain to be a component of the newly developing stand (Beck and Della-Bianca 1981).

Age of the 1988 preharvest stand was estimated from ring counts on stumps from trees cut in the third entry, in fall 2017. Oak stumps of various sizes were randomly selected in 12 openings and measured for diameter outside bark from the pith across the maximum and minimum stump surface dimensions. Stumps were identified by species and annual growth rings were counted in the field using a 10-power hand-held magnifier.

Species Groupings for Analysis

Tree species were classified in two primary groups (desirable or undesirable) associated with their traditional economic value (Della-Bianca and Beck 1985) and six secondary subgroups according to their habitat values (such as oaks) or focus of research interest (such as red maple, Abrams 1998) (Table 1). Desirable species are intolerant to midtolerant of shade, attain large size, form the overstory canopy and have commercial timber value; some (oak and hickory) are also important for wildlife because of their hard mast (acorn or nut) production. Undesirable species are shade tolerant, small, form the canopy midstory and have low commercial value, although many species produce soft mast used by wildlife. The desirable species group was further separated into four subgroups: red oak (subgenus *Erythrobalanus*), white oak (subgenus *Leucobalanus*), yellow-poplar, and other-desirable. The red oak subgroup included three species (ranked by affinity with decreasing soil moisture): scarlet < black < northern red; the white oak subgroup included two species (chestnut < white). The five oak species were separated into two subgroups because of their relative differences in shade tolerance as juveniles: red oak < white oak (Johnsen et al. 2002). Yellow-poplar was evaluated as a desirable subgroup because of its high commercial value and silvical characteristics as an aggressive, long-lived, mesophytic, intolerant species with wind-dispersed seeds that germinate best on mineral soil following disturbance of the forest floor. It is typically a pioneer species on heavily disturbed mesic sites where its high stem density and rapid growth can result in dense stands that retard growth of other species, but it can also persist as scattered individuals that become established in old-growth stands (Beck and Della-Bianca 1981). The other-desirable subgroup (hereafter desirable) includes minor species, primarily white ash (*Fraxinus Americana* L.), sweet birch (*Betula lenta* L.), black locust (*Robinia pseudoacacia* L.), mockernut hickory, and pignut hickory.

The undesirable species group was separated into two subgroups: red maple and other undesirable. Red maple, a light-seeded species, was evaluated separately because of its unusual ecological attributes of high shade tolerance and relatively quick growth response to increased light, thereby allowing it to capture growing space in openings following stand disturbances and persist as a long-lived species as it ascends in the canopy (Abrams 1998). In addition, red maple occurs across a range of site qualities and following harvests can present strong competition to desirable reproduction as rapidly growing basal sprouts from stumps during early stand development (Steiner et al. 2018). The other-undesirable subgroup (hereafter undesirable) includes shade tolerant species: flowering dogwood, sourwood, blackgum, witch hazel, serviceberry (*Amelanchier arborea* [F.Michx.] Fernald), American beech (*Fagus grandifolia* Ehrh.), and American holly (*Ilex opaca* Aiton.) in the midstory that are small and poorly formed and have greater value as producers of soft mast for wildlife habitat than for timber products. In review, tree composition was classified first in two groups, desirable or undesirable, and then in six subgroups for data summary and analysis: red oak, white oak, yellow-poplar, desirable, red maple, or undesirable.

Statistical Analysis

There were no imposed experimental treatments in our study. Species composition and structure of tree reproduction were examined to evaluate their response to opening size and position in openings. For the first entry (1988), openings were intentionally small (mean 0.10 ha, range 0.05–0.14 ha) to investigate minimum size required for establishment of intolerant desirable species. For the second entry (2005) openings were larger (mean 0.29 ha, range 0.12–0.41 ha) to follow operational national forest guidelines for uneven-aged management using the group selection method (USDA Forest Service 1994). Reproduction following the 1988 entry was inventoried in 13 of the 14 harvested openings; one opening was destroyed by a logging skid road constructed for the 2005 entry. Twenty-three openings were harvested in the 2005 entry, of which 14 were sampled.

Regression was used to evaluate the distribution of tree stems per hectare by 5 cm dbh classes of the preharvest stand and combined reproduction resulting from the two harvest entries for evidence of a reverse "J-shaped" (negative exponential) curve that is characteristic of uneven-aged stands (Leak and Filip 1977). Diameter distribution diminution quotients (Q) were calculated using the regression method of Leak (1963). The response of reproduction to opening size and position in openings was quantified by stem density (hereafter density), basal area and quadratic mean dbh (dbhq) of species subgroups for each entry. Quadratic mean dbh was calculated from plot stem density and basal area (Curtis and Marshall 2000). Density and basal area of subgroups were standardized as relative percent values of sample plot totals by center and edge positions and overall (pooled center and edge positions) opening. Histograms of plot mean stem densities indicated a tendency toward positive skewed distributions. The arcsine-square root transformation was used to approximate normal distributions (McDonald 2014). Untransformed means and standard deviations (sd) are presented in tables. A relationship between tree age and stump diameter outside bark was examined by oak subgroup using correlation and regression analysis. Correlation and linear regression were also used to examine relationships and test for effects of opening size on response variables of density, basal area and dbhq by species subgroup for pooled and nonpooled opening position (center, edge) by reproduction cohort age. Regression analysis was used to evaluate effects of opening size alone (disregarding center and edge sample positions) and in combination with position on mean total height of yellow-poplar at each reproduction cohort age. One-way analysis of variance (ANOVA) was used to test for significant differences of opening sample position on density, basal area and dbhq by species subgroup of reproduction resulting from each entry. Statistical differences of density, basal area and dbhq response variables among the six subgroups of species were not informative for objectives of our study and were not evaluated. The delayed oak dominance hypothesis was examined by one-way ANOVA of individual species subgroup changes of stem density between the two harvest entries and community structure change of combined subgroups using a modification of the method presented by Solow (1993) based on Simpson's index of diversity. Version 3.5.1 of R was used for data analysis (R Development Core Team 2011). Significance was determined at the $p = .05$ level.

Results

Before the first entry, stem density of advance reproduction (height <1.37 m) on sample plots in the mature stand throughout the entire study area consisted primarily of the red maple subgroup (69%), red oak (11%), and white oak (9%); yellow-poplar was 2% (Table 1). The midstory (height ≥1.37 m and dbh <10 cm) consisted primarily of shade tolerant red maple and flowering dogwood in the undesirable species groups. The overstory was dominated by species in the two oak subgroups followed by yellow-poplar. The diameter distribution of the entire pretreatment stand displayed a reverse J-shaped curve approximated by a calculated Q of 1.24 (Figure 2A). Mean age of the two oak subgroups in 1988 was 66 years (n = 34, sd = 29 years, range = 21–155 years). For the white oak subgroup, tree age was correlated with stump diameter outside bark (dob) ($n = 27$, $r = 0.669$, $p \leq .001$), expressed by the relationship

$$\text{Age (years)} = -8.35 + 1.26\,(\text{stump dob in cm}) \qquad (1)$$

with a standard error of estimate of 23.08 years. An age to stump dob relationship for the red oak subgroup could not be evaluated because the sample size was too small (n = 7).

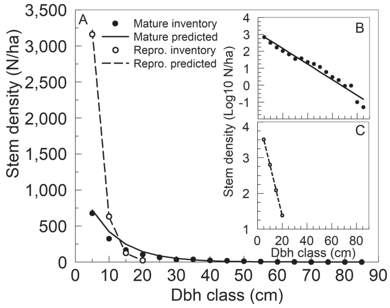

Figure 2. Size class (diameter at breast height [dbh]) distributions of tree stem densities by age cohort in the study area. (A) Inventoried and predicted stem densities for mature trees preharvest and reproduction postharvest.* (B) Log10-transformed preharvest mature stand. (C) Log10-transformed reproduction in the combined 1988 (20 years old) and 2005 (10 years old) openings. *Reproduction combines 1988 and 2005 harvest openings.

Opening Size and Position in Opening

Opening size was not significantly correlated with overall (pooled center and edge positions) mean relative density or basal area for any species subgroup in either of the two age classes of reproduction (not shown). Pearson correlation coefficients were relatively low for basal area of the 10-year-old red oak ($r = -0.20$) and white oak reproduction ($r = -0.25$) and were still lower for the 20-year-old reproduction: $r = -0.07$ for red oaks and $r = -0.01$ for white oaks. Correlation analysis of the effects of opening size based on nonpooled positions, however, indicated a significant relationship ($r = 0.61$, $p < .05$) of mean relative density for 10-year-old reproduction of the red oak subgroup in opening centers (not shown). Pearson's r was not significant for similar correlations between opening size with mean relative density and basal area by position and reproduction age class for the other species subgroups.

Position in openings alone (without the influence of opening size) significantly affected relative stem density and relative basal area of 10-year-old reproduction for the red and white oak subgroups and yellow-poplar (Table 2). Relative basal area was greater for the red oak ($p < .05$) and white oak ($p < .05$) subgroups at the edges of openings than in centers (Figure 3). In a reverse relationship, yellow-poplar basal areas were greater ($p < .05$) in centers compared with edges. Mean relative basal area between the center and edge positions were not significant different for the desirable, red maple and undesirable subgroups. In the 20-year-old openings the only significant effect of position on species was for mean basal area of red maple, which was greater ($p < .05$) at the edge than at the center of openings (Table 2). In openings of both harvest entries, basal area and dbhq of overall species subgroups were significantly ($p < .01$) greater in center compared with edge positions.

Mean height of yellow-poplar reproduction was not correlated with opening size across pooled sample positions for either the 10-year-old reproduction ($\overline{x} = 2.35$ m, n = 28, $r = 0.59$, $p = .058$) or at 20 years ($\overline{x} = 6.67$ m, n = 24, $r = 0.12$, $p = .713$) (not shown). Analysis of variance, however, indicated position in openings had a significant effect ($p < .01$) on mean height of 10-year-old yellow-poplar reproduction (Figure 4). Mean total height was 3.1 m (n = 16, sd = 1.39 m) in the center of openings and 1.3 m (n = 12, sd = 1.20) at the edge, a difference of 1.8 m. For the 20-year-old openings of the 1988 entry, mean total height of yellow-poplar was 7.7 m (n = 12, sd = 3.29 m) in center compared with 5.7 m (n = 12, sd = 3.22 m) at edge positions, but the difference of 2 m was not significant ($p = .14$).

A quadratic multiple regression model was developed to evaluate the combined effects of opening size, position in opening and their interaction on mean total height of yellow-poplar at each cohort age. For the 10-year-old yellow-poplar subgroup the best model (n = 28, $R^2 = 0.51$, $p = .0001$) included opening size ($p = .005$) and a categorical variable for position ($p = .0002$); the interaction was not significant ($p = .54$):

$$\text{Height (m)} = 1.7745 + 14.2866\,(\text{size}^2) - 1.8896\,(\text{edge position}) \tag{2}$$

where size is opening area in hectares squared and edge position is 1 or 0 (for edge or nonedge position). Variation explained by position increased the level of significance of opening size from $p = .058$ to $p < .001$ for the 10-year-old trees. Model (2) predicts mean yellow-poplar height at 10 years increases from 1.9 m to 3.9 m in opening centers as opening size increases from 0.13 ha to 0.41 ha (Figure 5). The position coefficient indicates yellow-poplar height averaged 1.9 m shorter at edges compared with centers of openings. A similarly formulated model for height of the 20-year-old yellow-poplar saplings in the small 1988 openings was not significant (n = 24, $R^2 = 0.10$, $p = .32$) and is not shown.

Species Composition and Structure

Mean total basal area of reproduction in the openings at 10 years was 8.0 m^2/ha and 14.1 m^2/ha at 20 years, an increase of 77%. (Table 2). Basal area of all species subgroups increased between 10 and 20 years, but none of the changes were significantly different. The overall percent increase of basal area between 10 and 20 years was less in opening centers (71%) compared with edges (92%).

Between the first and second entries, the overall mean basal area of yellow-poplar decreased from 67% to 47% ($p = .17$), and the desirable subgroup from 15% to 12% ($p = .73$) (Figure 6). This total decrease of 23% was matched by a total increase of 11% for the red and white oak subgroups ($p = .57$ and $p = .70$, respectively) and 12% for the combined red maple ($p = .49$) and undesirable subgroups ($p = .48$). The mean increase of white oak basal area (7%) between the two entries was nearly double that of the red oak subgroup (4%). Twenty years after the first entry, the shade tolerant red maple subgroup was less than inventoried preharvest in the mature stand (4% versus 14%), but the undesirable subgroup occupied 17% in the regenerated openings compared with 5% in the preharvest stand.

Large trees (≥10 cm dbh class) in opening centers were dominated by the yellow-poplar subgroup and lesser stem densities in the two oak subgroups (Figure 7). Also present (not shown in Figure 7) in centers were species in the desirable subgroup (primarily sweet birch) consisting of 133 trees/ha in the two smallest dbh classes. In the periphery of openings, density of the yellow-poplar subgroup (152 trees/ha) was similar to stem density of the two oak subgroups (133 trees/ha), particularly for white oaks. The undesirable subgroup of species was represented in the edge position by 38 trees/ha (primarily sourwood) in the 10 cm dbh class. All red maple stems were ≤7.5 cm dbh.

Table 2. Mean density (standard deviation in parentheses), mean basal area and quadratic mean diameter breast height (dbhq) by position in openings, reproduction age, and species subgroups following two harvest entries in the mixed-hardwood stand of the Frothingham group selection study area.

Reproduction Age, Species Subgroups	Stem Density (n/ha)			Basal Area (m²/ha)			Dbhq (cm)		
	Center	Edge	*p**	Center	Edge	*p*	Center	Edge	*p*
10-year-old openings (n = 14)									
Red oak	230 (383)	1,659 (2,012)	<.01	0.21 (0.63)	0.34 (0.48)	.04	1.12 (1.35)	1.22 (0.70)	.80
White oak	971 (1,955)	5,416 (7,969)	<.01	0.11 (0.34)	0.74 (1.32)	<.01	0.70 (0.23)	1.35 (0.64)	.01
Yellow-poplar	11,750 (6,541)	8,222 (11,855)	<.01	8.16 (5.57)	2.55 (3.68)	.04	2.88 (1.51)	2.03 (2.26)	.26
Other desirable	2,047 (2,438)	1,870 (2,022)	.71	1.98 (2.63)	0.39 (0.47)	.44	3.06 (1.70)	1.46 (0.60)	<.01
Red maple	971 (916)	2,047 (2,636)	.20	0.23 (0.36)	0.13 (0.15)	.40	1.34 (0.91)	0.81 (0.35)	.06
Other undesirable	2,382 *(3,974)*	*2,929 (3,080)*	*.41*	0.73 *(1.00)*	*0.39 (0.40)*	*.29*	*1.67 (1.11)*	*1.21 (0.63)*	.19
Overall subgroups	18,350 (6,962)	22,142 (11,168)	.29	11.42 (4.89)	4.53 (3.18)	<.01	2.94 (1.07)	1.61 (0.55)	<.01
20-year-old openings (n = 13)									
Red oak	551 (622)	646 (938)	.34	1.39 (2.83)	0.78 (2.05)	.86	2.45 (2.78)	1.72 (1.77)	.44
White oak	476 (617)	817 (1,142)	.20	1.86 (5.06)	1.51 (2.11)	.18	2.48 (3.43)	3.98 (4.45)	.36
Yellow-poplar	2,527 (1,984)	1,938 (2,334)	.26	10.26 (9.12)	3.05 (2.71)	.23	7.56 (5.05)	4.93 (4.20)	.17
Other desirable	1,539 (1,675)	1,406 (1,712)	.86	2.46 (2.80)	0.95 (1.76)	.75	3.43 (2.23)	1.93 (1.51)	.06
Red maple	1,406 (1,844)	1,121 (810)	.73	0.40 (0.80)	0.74 (1.22)	.04	1.42 (0.91)	2.06 (1.45)	.20
Other undesirable	1,121 *(965)*	1,273 *(1,192)*	.59	3.16 (6.79)	1.68 (2.62)	.73	3.47 *(3.02)*	2.89 *(1.80)*	.55
Overall subgroups	7,620 (2,844)	7,202 (3,610)	.74	19.54 (8.05)	8.70 (4.65)	<.01	5.87 (1.69)	4.10 (1.47)	<.01

**p* determined from *F* statistic resulting from one-way analysis of variance of position in opening (center versus edge) by species subgroup and reproduction age class.

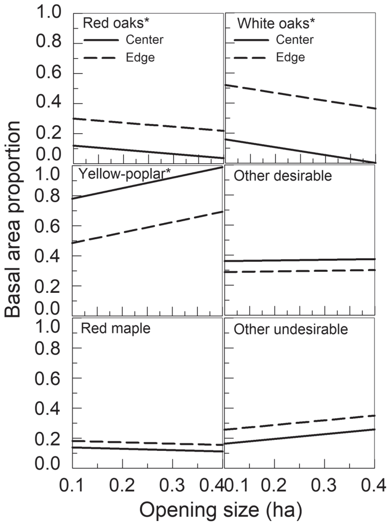

Figure 3. Effects of position in opening and trends of opening size on relative basal area of reproduction for six species subgroups 10 years after the 2005 entry. Relative basal area of species subgroups followed by an asterisk (red oaks, white oaks, yellow-poplar) were significantly ($p < .01$) influenced by position in the opening.

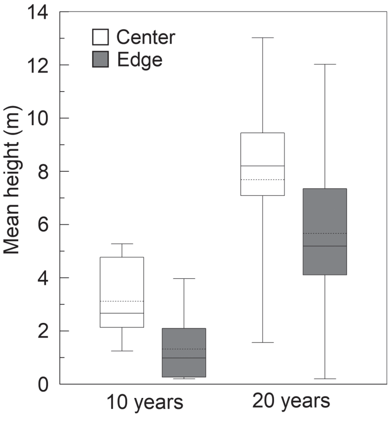

Figure 4. Box-and-whisker plots of yellow-poplar mean height of reproduction in relation to center and edge positions in openings at 10 and 20 years of age. The lower and upper whiskers of each box represent the minimum and maximum mean heights of trees in openings, the box ends are the 25 and 75 percentiles of mean heights, the solid line in the box is the median, and the dashed line is the mean tree height. Mean heights were significantly different ($p < .01$) between center and edge positions for both cohort ages of yellow-poplar reproduction.

Analysis of the two species groups combined as a community of reproduction, rather than individually as subgroups, revealed a significant change of dominance between the first and second decades of development (Table 3). Simpson's index of diversity increased (greater diversity) significantly ($p = .035$) for the openings overall, from 0.603 to 0.704. Diversity of species groups did not change at opening edges ($p = .141$) but increased significantly in centers ($p = .002$). Although the diversity index of the preharvest stand was greater than for the 20-year reproduction (0.881 versus 0.782), the preharvest and postharvest communities could not be statistically compared because of an incompatible data structure used for the mature stand inventory.

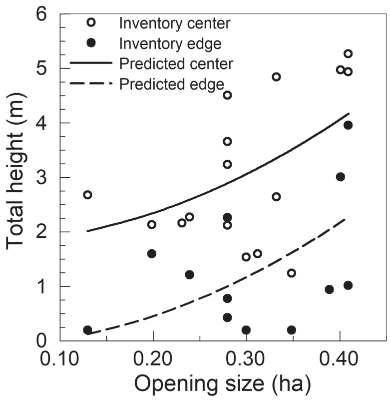

Figure 5. Inventoried and predicted mean total height of 10-year-old yellow-poplar reproduction in relation to the combined effects of opening size and center or edge positions.

Stem density of the combined reproduction subgroups in openings of both entries (Figure 2C), which attained a maximum dbh of 20 cm, was nearly six times greater than for the preharvest mature stand (Figure 2A). The regression of log10-transformed stems per hectare graphed as a function of 5 cm dbh classes for the pooled two cohorts of even-aged reproduction in the 1988 and 2005 entries displayed the linear form of a balanced distribution (Figure 2C). The regression value of Q (1.92) for the two combined age classes of reproduction was significant ($p < .01$, $R^2 = 0.999$).

Discussion

Following the failure of long-term management by single tree selection to recruit desirable reproduction in stands of upland Appalachian oaks (Della-Bianca and Beck 1985, Keyser and Loftis 2013), the Frothingham study was established in an adjacent stand to evaluate group selection harvests to achieve that goal. Before the first entry basal area composition of the mature stand consisted largely of approximately similar amounts of oaks (36%) and yellow-poplar (31%) (Table 1). Twenty years after the first entry postharvest reproduction composition in the small (0.10 ha) openings matched that present in the preharvest stand, except for absence of shortleaf pine, which was a remnant artifact of previous woodlot land use. Species basal area, however, shifted to dominance by yellow-poplar (47%) over oaks (20%), which can be partly attributed to a different disturbance regime following federal land acquisition associated with reduced woodland burning, and livestock grazing in particular. For example, the ranking of tree species favored by cattle for browsing (red maple < oaks < yellow-poplar) was likely a contributing factor resulting in reduced presence of yellow-poplar and increased establishment and survival of oaks (Biswell and Hoover 1945). Although mean stem densities ranged widely among the 13 openings of the 1988 entry and 14 openings of the 2005 entry, the diameter structure of reproduction for the combined openings displayed the (log10-transformed) reverse J-shaped curve of a balanced, uneven-aged stand (Figure 2C).

Opening sizes ranged from 0.05 ha to 0.41 ha for the 27 openings inventoried in the two entries of our study. We found no significant correlations of opening size alone with stem density or basal area of reproduction for any of the two species groups, a result that differs from findings reported in other group selection studies in mixed-hardwood forests. In a study of three opening areas (0.02, 0.16, 0.46 ha), 48 years after harvests in mixed upland oak stands, Lhotka (2013) reported greatest stem densities of yellow-poplar in the largest openings, oak in intermediate sizes and shade tolerant species in the smallest size. Minckler and Woerheide (1965) also reported greater densities of yellow-poplar in large openings (diameter greater than height of the surrounding stand) and oaks in smaller openings (diameter less than height of the adjacent stand). In a group selection study of six opening sizes (0.015–0.503 ha) in bottomland mixed-hardwood stands of South Carolina, Collins and Battaglia (2008) reported no optimum size for enhancement of oak reproduction but reported greater stem densities in the largest openings. Zhu et al. (2014), in a global metastudy, concluded there was no consistent optimum opening size favorable for oak reproduction. In agreement with Zhu et al. (2014), we were unable to determine an opening size most favorable for oak reproduction. It is evident from results of our study, however, that reproduction of desirable intolerant species, including oaks, can be obtained in openings as small as 0.05 ha in dry-mesic mixed-hardwood stands.

Position had a greater effect than opening size on density and basal area of several species subgroups, particularly oaks (Table 2). Density of oak reproduction was greater at edges of openings, compared with centers, which agrees with 10-year results from a group selection study reported by Holladay et al. (2006). In our study, stem density and basal area of white oaks were strongly associated with position and were five times greater at edges of openings than in centers. The red oak subgroup showed a similar relationship with opening position for density, but a weaker association with basal area. Shade-intolerant yellow-poplar was the dominant species in the centers of all openings. Distribution of the shade-tolerant undesirable group of species was not significantly associated with either the periphery or center of openings. Our findings agree with other studies reporting variable distribution of species in openings, likely as a response primarily to reduction of light resulting from shading by other species (Gottschalk 1994). In agreement with our results, Dale et al. (1995) and Collins and Battaglia (2008) reported greater density and basal area of oak reproduction at edges of openings than centers. As we observed and consistent with what has been observed elsewhere in the CHR, yellow-poplar can be the primary competitor to oak in group selection openings on dry-mesic sites, particularly in the center of openings (Smith 1977, Weigel and Parker 1997, Iverson et al. 2017). Trends of composition changes between 10 and 20 years (Figure 6), however, indicate decreasing density of yellow-poplar and reduction in the desirable subgroup primarily by sweet birch and black locust, as Beck and Hooper (1986) reported for a mesic clearcut site. During this 10-year interval basal area of the other subgroups approximately doubled, particularly for the undesirable subgroup of shade tolerant species. Noteworthy during this period were increases in relative basal area of the two oak subgroups, particularly by white oaks (from 5% to 12%) (Figure 6), which occurred in nearly all openings and particularly among trees of large dbhq (>10 cm) at edge positions (Figure 7). Although large oaks at edge positions were sparse (133/ha), we suggest this trend of reproduction development could be preliminary evidence in support of the

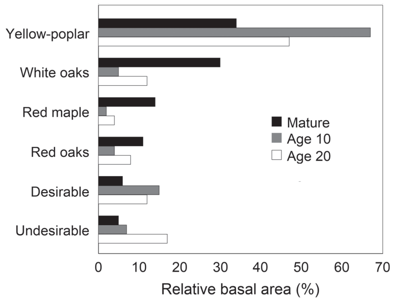

Figure 6. Proportion of total tree basal area ≥2.5 cm diameter breast height (dbh) by species subgroup in the mature mixed-hardwood stand before the first entry and in the regenerated openings after 10 years and 20 years.

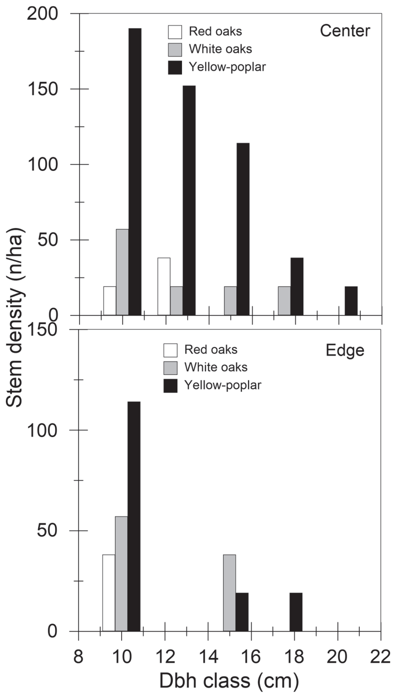

Figure 7. Distribution by 2.5 cm diameter breast height (dbh) classes of predominant species subgroups consisting of large trees (≥10 cm dbh) at the opening center and edge of 20-year-old reproduction of oak and yellow-poplar subgroups. Not included at the opening centers of the desirable subgroup were 133 stems/ha in the 10 and 12.5 cm dbh classes and at the edges of the undesirable subgroup were 38 stems/ha in the 10 cm dbh class.

delayed oak dominance hypothesis (Oliver 1978). Self-thinning by a pioneer species during the stem exclusion stage of stand development, however, is an alternative, simple explanation for changes of relative basal area occurring at a young age in the long-term development (100+ years) of this stand (Figure 6). The uneven-age diameter distribution pattern of the even-aged stems shown in Figure 7, however, offers a preview of their possible future composition in the Frothingham study area. Our second-decade results agree with early development dynamics of similar stands reported by Beck and Hooper (1986) at 20 years. Information on older development of mesophytic clearcut stands is sparse in the southern Appalachians, but 60-year results from a burned and salvage-cut old-growth forest (Della-Bianca 1983) suggest that much of our study area could eventually be dominated by a composition of yellow-poplar and mixed oaks, similar to the preharvest stand (Table 1).

Testing the delayed oak dominance hypothesis was an objective of our study, but we were unable to satisfactorily (statistically) accomplish that. Because a long-term study over a complete rotation would be required for a conclusive confirmation of the hypothesis, we were uncertain about the type of convincing supporting evidence needed from our short-term observations because results differed by opening position. Assuming that trees ≥10 cm form the intermediate and higher canopy in the 20-year-old openings, the opening centers were dominated by a 3:1 ratio of yellow-poplar to oaks (Figure 7). The predicted transition rate for change of crown class is low for oaks (Ward and Stephens 1994). This suggests little shift in species composition in the centers of openings of the future stand unless unfavorable climatic conditions (such as drought) results in slower growth or high mortality of yellow-poplar on submesic sites (Hilt 1985) or application of intermediate management treatments, such as crop tree release (Miller et al. 1994). At the edges of openings, however, the mean ratio of large (≥10 cm) yellow-poplar to oaks was approximately 1:1 and is likely to remain at that level or slowly change in favor of oaks with future stand development, which is in agreement with the hypothesis. In our study, however, site preparation and competition release treatments using herbicide three years after harvest were likely a contributing factor favoring the codominance of oaks with yellow-poplar, particularly around the edges of openings. Although a tenet of the hypothesis (shifting species dominance) has been addressed in a few studies in mixed-hardwood stands conducted during the stem exclusion stage (Morrissey et al. 2008), we found only one study that specifically addressed the hypothesis. Steiner et al. (2018) examined regenerated oak-red maple stands midway through rotations and reported variable support for the hypothesis among three ecoregions in central Pennsylvania and across a range of site qualities. They reported oaks were replacing red maple in the central Appalachian Mountains, particularly on dry-mesic sites, but the reverse was apparent in other ecoregions and on higher quality sites. Although we found no significant change in subgroup dominance (expressed as stem density) between the two age classes of reproduction in our study, changes in the total tree community were significant, which we suggest provides tentative support for the oak dominance hypothesis.

Table 3. Mean (standard deviation) Simpson's diversity index of the preharvest, mixed-hardwood stand and reproduction cohort ages resulting from two harvest entries by position in the group selection openings of the Frothingham group selection study area.

Position in Opening	Preharvest Stand	Harvest Entry (Cohort Age)		p*
		2005 (10 years)	1988 (20 years)	
Center	N/A**	0.429 (0.174)***	0.622 (0.108)	.002
Edge	N/A	0.602 (0.221)	0.702 (0.098)	.141
Overall	0.881	0.603 (0.147)	0.704 (0.782)	.035

*p determined from F statistic resulting from one-way analysis of variance of harvest entry date by position in opening.
**N/A, not applicable because of sampling design.
***Larger values of Simpson's index are associated with greater species diversity.

The response of reproduction in relation to the herbicide site preparation and followed three years by competition release treatments could not be evaluated because controls were lacking. As reported by Smith (1981) in mixed-oak stands of the central Appalachians, the herbicide treatments undoubtedly provided some benefit to survival and growth of reproduction in the preferred oak subgroups at 20 years of age and, hence, increased tentative support for the dominance hypothesis particularly in the periphery of openings during the near future. In our hypothesis tests, the chronosequence design of our study likely reduced sensitivity of the analysis to detect significant temporal changes of oak reproduction dynamics relative to yellow-poplar during the second decade of growth. Our results agree with findings reported by Beck and Hooper (1986) on permanent plots showing continued self-thinning of yellow-poplar and high survival of oaks after two decades of development in a clearcut, mixed cove hardwoods stand on a mesic site. Continued development of reproduction in the harvested openings suggests future composition like that of our preharvest stand (Table 1). Results from a long-term study of a severely burned and clearcut old-growth stand of cove hardwoods suggests initial dominance by yellow-poplar through 60 years for the Frothingham study with increasing codominance by oaks and a midstory of tolerant species (Della-Bianca 1983). Although previous influences of landscape scale fire and grazing have changed during the past century, other climate-related disturbance events (and possibly more intense than before) over the long rotation of these forests will continue to influence species composition.

Composition of reproduction varied widely among openings. Field observations suggested species distributions could be strongly affected by factors not quantified in our study, such as site quality, exotic species and disturbance during timber harvest. In the 1988 entry, for example, mesophytic yellow-poplar dominated the 12 openings that were located on dry-mesic slopes or on mesic sites along drainages. However, reproduction of yellow-poplar was absent from an opening on a ridge crest, possibly because of the likely xeric moisture regime there and lack of nearby seed sources. Dominant reproduction at that ridge site consisted of the oak subgroups, particularly chestnut oak, and hickories together with red maple and the undesirable subgroup. In another small opening on a mesic site harvested in 1988, the canopy was dominated by the nonnative, shade intolerant, light-seeded, tree-of-heaven (*Ailanthus altissima* [Mill.] Swingle) with a few yellow-poplar trees in the intermediate and codominant crown classes. This was a surprising finding because yellow-poplar had been "outcompeted" by an exotic species, for which there was no known seed source in the vicinity of the Frothingham study area and because tree-of-heaven is dioecious. Rebbeck et al. (2017) reported similar invasion by this exotic species following timber harvest in closed oak forests of southern Ohio.

Unusual site conditions resulting from harvesting possibly contributed to variation of reproduction in some openings of our study. For example, the mean ratio of yellow-poplar to oak density of reproduction in our study was 12:1 in centers and 2:1 at edges. In one of the 2005 openings, however, the ratio was 68:1 in the center and 0.29:1 at the edge. The low yellow-poplar to oak edge ratio in this second-entry opening could have resulted from a combination of two factors: edge effects from increased light admitted from an adjacent first-entry harvest that stimulated development of advance oak reproduction and lack of forest floor disturbance, which could have reduced but not eliminated germination of new and stored yellow-poplar seeds (Clark 1970). This opening was adjacent to a permanent logging access road that allowed felling of several mature perimeter trees into the opening edge followed by removal of the butt log with minimal soil disturbance from harvest equipment. The combination of a thick layer (~1 m) of logging slash from several large tree crowns and lack of soil disturbance in this edge sample plot apparently resulted in successful intraspecific competition within a large cluster of oak reproduction (observed present before the second entry) rather than interspecific competition with a dog-hair thicket of yellow-poplar saplings, which was present in the opening center. Sander and Clark (1971) also observed that basal sprouts from oak advance reproduction crushed by logging residues can emerge and grow rapidly in height.

The response variable of yellow-poplar height was significantly, directly associated with opening size only in the presence of center or edge position and only for the relatively large openings formed during the 2005 harvest (Figure 5). The quadratic effect of opening size in our model agrees with Sander and Clark (1971) who provided a curvilinear relationship of the area shaded by the adjacent forest in relation to opening size. Where timber production is a regeneration priority, Sander and Clark (1971) suggest a minimum opening size of 0.2 ha to reduce edge effects on growth of intolerant species. In agreement with our results, Smith (1977) reported better height growth of yellow-poplar as opening size increased from 0.018 to 0.45 ha and better growth in centers of openings compared with edges.

Several design issues weakened findings from our study. A design with increased power to detect effects of opening size could have resulted from harvest of replicated fixed areas rather than variable sizes, as in our study. Lack of preharvest inventories in openings reduced our ability to assess competition with focal species (oaks) by specific shade tolerant competing species, such as red maple. In retrospect, sampling edges of openings in a random direction increased variability of inventory data and did not allow testing of effects of aspect on reproduction, which was significant in the CHR north of the Ohio River (Weigel and Parker 1997, Morrissey et al. 2008). In the central Appalachians, however, Smith (1977) reported no effect of aspect on reproduction. We avoided placing edge samples next to logging roads to reduce effects of increased side light on reproduction in openings. Except for the instance mentioned above, however, we failed to note if edge plots sampled in the second entry were adjacent to openings created by the first entry. Also, an estimate of site index from each opening could have been useful to account for variation in some response variables, such as basal area of the yellow-poplar subgroup. Finally, our study used a chronosequence design to detect temporal trends of reproduction development at two stand ages, which introduced site and climatic related variation that would not have been present with permanent plots established in the first entry. We suggest the deficiencies in our study should be considered for addressing in future investigations, particularly increased sample size using permanent, paired center/edge plots on cardinal directions.

Effects are unknown concerning the herbicide release treatment three years after harvest on growth of desirable species, primarily the oak subgroups. Given the rapid growth of yellow-poplar observed in this study and the characteristically slow height growth of small oak reproduction, beneficial effects of the herbicide release treatment could have been minimal, particularly in opening centers with high seedling densities yellow-poplar. The site preparation herbicide treatment of stumps, however, was apparently beneficial because red maple and other prolifically sprouting species (sourwood) were minor components of the inventoried reproduction; Smith (1981) reported similar results from herbicide site preparation treatments in West Virginia. The efficacy of herbicide release treatments on targeted reproduction development has not been widely evaluated in group selection harvests and could be a subject of evaluation in the next entry of the Frothingham group selection study.

Results from our study suggest group selection can be used to regenerate mature, mixed-hardwood stands with reproduction of desirable intolerant species, primarily with yellow-poplar but also with a small component of oaks. Stand management by group selection, however, requires additional administrative investment compared with conventional even-aged methods. For example, the harvesting access road system will be used for each entry and should be designed before the first entry to avoid disturbances to existing regenerated openings in future entries, which occurred in one opening of the Frothingham study. Detailed mapping of opening locations is desirable for distribution of harvests throughout the stand to meet wildlife habitat and visual objectives. Minimum sizes of openings must be considered for economics of harvest operations. For example, based on predicted crown sizes, an opening of 0.05 ha could include one tree in an oak subgroup with dbh of 152 cm or approximately 12 trees averaging 30 cm dbh (Bechtold 2003). Conventional opening size is a width about twice the height of the adjacent forest (Miller et al. 1994). Forest height in the Frothingham study area averaged about 33 m, resulting in an opening width of 66 m and size of about 0.34 ha. Larger opening sizes are considered patch clearcuts but achieve the same management objective of an uneven-aged stand structure consisting of three or more cohort ages (Miller et al. 1994). In central Appalachian mixed-hardwood stands, Smith (1981) recommended 0.20 ha as the minimum size to obtain satisfactory density of oak reproduction and adequate diversity of other desirable intolerant species. Additional information on implementation of uneven-aged management using the group selection method in Appalachian oak stands is provided by Miller et al. (1994).

Conclusions

Preliminary results from this long-term study show reproduction in 10- and 20-year-old openings consisted largely of intolerant yellow-poplar and lesser, about equal amounts of midtolerant oaks and tolerant undesirable species. Opening size, which ranged from 0.05 to 0.41 ha, had no effect on stem density or basal area of reproduction in any of the six subgroups of species. Position in openings, however, strongly affected species composition of reproduction with yellow-poplar dominating centers and codominating edges with oaks. During the second decade of reproduction development, basal area of the yellow-poplar and desirable subgroup (largely sweet birch and black locust) decreased while the proportion of other species subgroups increased, particularly oak and undesirable species subgroups. Our results suggest that composition of reproduction in openings as small as 0.05 ha can consist of desirable intolerant species, but the administration and economics of operational harvesting of only small openings could be questionable. Although an abundance of small openings could affect economics of harvesting using group selection, their distribution within stands could be important to achieve nontimber management goals, such as aesthetics and wildlife habitat. Also, we reported that stem diameters of 10- and 20-year-old even-aged reproduction followed a balanced, uneven-aged distribution, which was a technical concern by some early opponents of group selection. Early results from this long-term, operational-scale study suggest that group selection can be a viable method of uneven-aged management in dry-mesic southern Appalachian mixed-hardwood forests.

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Chapter 8

Forests and Climate Change in the Southeast USA

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The forests of the southeastern USA have seen many changes over the past 200 years. From cutting of the virgin forest in the 19th century to the expansion and later contraction of farming and the rise of plantation forestry in the 20th century, the structure and function of southern forests continues to evolve. Climate change represents another factor that is changing Southeast (SE) forests and forestry practices. Current and continued projected warming likely will increase the risk of wildfire, wind, insect, and disease damage to southeastern forests. Increased air temperatures also likely will lead to decreased forest water yield, even as the population of the SE USA continues to expand. Conflicts between maximizing forest carbon sequestration as a mitigation strategy for climate change and need for water likely will continue well into the 21st century. However, scientists are actively working with land managers to address these issues. Although the form of SE forests will continue to change due to old and new pressures, forest managers are becoming increasingly better prepared to cope with these challenges. This chapter examines some of the pressing issues and management options associated with global change in the SE USA.

Key Findings

- Warming air temperatures likely will increase regional drying through increased forest water use via evapotranspiration (ET) regardless of changes in precipitation, and this drying will likely increase wildfire risk across SE USA forests.
- Longer growing seasons will likely increase the risk of insect outbreak and very likely will expand the northern range of some species, such as the southern pine beetle.
- Under most scenarios, increasing temperatures and decreasing precipitation will result in a greater uptake of soil water by forests and lead to reductions in streamflow.
- Despite climate and land use changes, forests in the southeastern USA will likely continue to provide a sink of atmospheric carbon dioxide (CO_2).
- The potential savannafication of the SE, in which forests are converted into more open woodlands due to a combination of hotter and drier conditions, could be one of the most profound potential climate change impacts in the USA

8.1 Historical Perspective

The forests of the Southeast (SE) USA have seen extensive change during the past century. Currently, 60% of the SE landscape is forested (Wear and Greis 2002). In 1860, about 43% of the SE land area was reported as under cultivation, but a substantial part of the farm holdings that remained in forest were used for grazing livestock (Smith and Darr 2009). Timberland continued to decline until the early 1920s due to the continued expansion of settlements. Significant changes in agriculture took place after 1920 that resulted in abandonment of large areas of crop and pasture lands. Some of the abandoned land was planted with trees, but most of the land reverted naturally to forest, leading to increases in timberland acreage (Wear et al. 2007). By the late 1950s and early 1960s, decline of timberland began again in the SE, caused primarily by the clearing of forests for soybean and other crop production. Much of this timberland reduction occurred in bottomland hardwood forest areas of the Mississippi Delta.

Throughout the 1970s, timberland was cleared for agricultural use and for an expanding export market. The decade beginning in 1982 marked a slowing of forest cover loss with the National Resources Inventory reporting roughly a half million-acre loss (less than 1%) in forestland in the SE (Wear et al. 2007). That trend has continued into the 21st century as softwood pulp prices have fallen by 50% since 1998, and the forest products industry divested approximately 75% of its timberland holdings (Butler and Wear, In Press). Although market prices will likely continue to be a driving factor in forest land area, other ecosystem services such as climate change, wildlife protection, drinking water supply, and recreation may increasingly influence the distribution and composition of SE forests (Wear and Greis 2002).

8.2 Southeastern Forest Types

The southeastern USA is not comprised of a single forest type, but of many. This assessment of forests and climate change focuses on six distinct forest areas within the SE: the Atlantic and East Gulf Coastal Plain, Piedmont, Appalachian/Cumberland, Mid-South, Coastal, and the Mississippi Alluvial Valley. Current inventory data shows that more than 30 million hectares of upland hardwood forests dominate the SE, followed by more than 15 million hectares of planted pine, approximately 13 million hectares of natural pine and bottomland hardwoods, and more than 3 million hectares oak-pine forest types (Butler and Wear 2012). These forest ecosystems provide a multitude of goods and services including clean water and air, wildlife habitat, recreation and aesthetics, timber and fiber production, and CO_2 sequestration. This chapter reviews current and future stresses on services provided by SE forests, and examines how forest management could be used to cope, adapt, or mitigate negative impacts.

Atlantic and East Gulf Coastal Plain. Historically, most of the southeastern Coastal Plain was dominated by fire-dependent longleaf pine (*Pinus palustris*) savannas (Christensen 2000). However, upland closed-canopied forests occur in mesic areas protected from frequent fire or where fire suppression has occurred. Notable examples of old-growth mesophytic beech-magnolia forests are present in the Apalachicola National Forest of the Florida panhandle. Other Coastal Plain broadleaved forests include those dominated by southern oak species such as swamp chestnut oak (*Quercus michauxii*), cherry bark oak (*Quercus pogoda*), and live oak (*Quercus virginiana*), as well as hickories (*Carya spp.*) and loblolly pine (*Pinus taed*a). American holly (*Ilex opaca*), spice bush (*Lindera benzoin*), and pawpaw (*Asimina triloba*) are common in the understory and subcanopy (Christensen 2000).

The distribution of current and potential future droughts and associated fire risk varies as does the potential impacts on trees species. Several dendrochronological analyses of Coastal Plain longleaf pine trees demonstrate the impact of growing season drought severity in relationship to reduced tree growth rates, as well as positive impacts of warmer winter temperatures (Bhuta et al. 2009, Henderson and Grissins-Mayer 2009). Less climatic research has been conducted in closed-canopied upland forests, but increasing fire frequencies in these Coastal Plain forests, due to ongoing and potential future droughts, may be a major impact on the distribution of some forests of this region (Wade et al. 2000). However, a recent study by Gruhn and White (2011) examined the northward range expansion of southern magnolia by comparing establishment success with climatic and topographic variables. Although minimum winter temperatures and the number of frost-free days were important determinants of establishment success, precipitation was not.

In addition to drought and fire, Coastal Plain forests and other ecosystems are also particularly vulnerable to hurricanes. Hurricane Isabel in September 2003 damaged 15% of trees, particularly canopy trees, in a maturing hardwood forest of the Virginia Coastal Plain (Prengaman et al. 2008). Hurricane Isabel was only a Category 2 storm, so increased frequencies of Category 4 and 5 hurricanes as a consequence of climate change likely will have even more profound effects (Webster et al. 2005, Knutson et al. 2011). Hurricane Katrina is an example of the damage caused by a strong hurricane (MIFI 2005).

Southern Appalachians. These forests cover much of the high elevation areas of the north-central southern region that includes eastern Tennessee and Kentucky, western North Carolina and Virginia, and northern Georgia. The southern Appalachian forests are some of the most diverse in North America (Clark et al. 2011). Both unique species and commercially important species can be found within the region. The diversity of these forests is controlled by regional and local weather patterns that can be highly variable due to the mountainous terrain (Clark et al. 2011). As with other mountain systems, the high elevation forests of the southern Appalachian ecosystems are at particular risk from a warming climate. A 3^{o}C increase in July temperature would raise climate-elevation bands by about 480m, resulting in the extirpation of the rare red spruce-Fraser fir (*Picea rubens* and *Abies fraseri*) alpine forests growing at the highest elevations in North Carolina and harboring federally threatened animal species, including the North Carolina flying squirrel (Delcourt and Delcourt 1998). Many of the mid-elevation "cove" forests, which are currently dominated by mesic, fire-intolerant tree species, are extremely diverse in terms of canopy trees, spring ephemeral wildflowers, and amphibians. Since the early 1980s this region has warmed and precipitation variability has increased. If these trends continue, they could lead to substantial change in the structure and function of future southern Appalachian forests.

In addition to determining biodiversity, climate variability also controls forest growth. For example, the annual growth rate of five dominant oak species can be severely affected by growing season drought intensity (Speer et al. 2009). During drought years, observed oak forests showed diminished productivity and accumulated 40% less carbon compared to a year of average precipitation (Noormets et al. 2008). If projected temperature increases are accompanied by decreased growing season precipitation, the combined changes may reduce the competitiveness of oaks in the southern Appalachians and elsewhere in the SE (Ibáñez et al. 2008).

Wildfires also shape the structure and function of forests within the southern Appalachians. A recent study suggested that fires occurred fairly frequently over the past 4,000 years in a variety of southern Appalachian forest types including those now dominated by mesic hardwoods, including tulip poplar (*Liriodendron tulipifera*) (Fesenmyer and Christiansen 2010). These researchers found that fire return intervals appear to have been of centuries-scale duration in the time period 4,000 to 1,000 years before present, and were likely often severe. Fires became more frequent approximately 1,000 years ago and were thus likely less severe due to less accumulated fuels build-up. The increased frequency of fire coincided with the occupation by Woodland Tradition Native Americans. If drought and drought-induced fires become more common in the southern Appalachians, fire-tolerant oak and hickory species may become more abundant over less-tolerant tulip poplar, maple (*Acer spp.*), basswood (*Tilia americana*), birch (*Betula* spp.) and magnolia (*Magnolia* spp.) species, potentially reducing diversity in currently highly-diverse mesic forests (Fesenmyer and Christiansen 2010).

Piedmont. The Piedmont region lays southeast of the Appalachian region and stretches from east-central Alabama through central Georgia, northwestern South Carolina, and central North Carolina and Virginia. These forests are dominated by a mixture of pine and deciduous species (Figure 8.1) of high commercial importance (Van Lear et al. 2004). Dale et al. (2010) used ecosystem models and an ensemble of general circulation model (GCM) scenarios to project that in the southeastern Piedmont and Appalachians, southern mixed hardwoods and pine forests on the Piedmont were the most susceptible to changes induced by warmer, and particularly drier, climates. Under the driest of the three climate scenarios considered by Dale et al. (2010), a southern mixed forest transitioned from very high tree species diversity with 14 commonly co-dominant species to very low forest diversity, dominated by loblolly pine, southern red oak (*Quercus falcata*), and Shumard's oak (*Quercus shumardii*). Dale et al. (2010) also found that the less-diverse forests may be more susceptible to insect and pathogen pests, and that hickory (*Carya spp.*) species tended to increase in relative importance under the climate change scenarios considered. Conversely, under those projections the biomass of chestnut (*Quercus prinus*) and black oaks (*Quercus velutina*) tended to decline across Tennessee, as the hickories appeared to be better able to grow in the warmer, drier climate relative to the oak species.

Research from the Duke Free-Air CO_2 Enrichment (FACE) experiment and the Oak Ridge FACE experiment in the southern Appalachians suggests that an approximate doubling of atmospheric CO_2 increases the productivity of the canopy loblolly pine and sweet gum (*Liquidambar styraciflua)* trees by 23% to 27% (DeLucia et al. 2005, Norby et al. 2005). However, when examining the juvenile tree species most likely to comprise the future forests, elevated CO_2 conditions favored the population biomass growth of less productive, shade-tolerant tree species southern sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*) as well as woody vines such as poison ivy (*Toxicodendron radicans*) (Mohan et al. 2006, 2007, and 2008) and exotic Japanese honeysuckle (*Lonicera japonica*) (Belote et al. 2004). So, increased atmospheric CO_2 levels may benefit a variety of species, but it is unclear from these few studies how elevated CO_2 levels coupled with other potential stresses may affect the composition of future forests as a whole.

Coastal wetland forests. Coastal wetland forests exist in the transition between the Coastal Plain and maritime ecosystems and are responsive to changes in climate and freshwater outflow resulting from varying patterns and frequencies of freeze, drought, storm, sea level, and runoff events. Because saltmarshes and mangroves thrive in the

Figure 8.1 Mixed conifer and deciduous Piedmont forest in the southeastern USA.

intertidal zone between land and sea, these systems are expected to undergo the most severe changes from marine effects, such as sea level rise and salinity. They are also affected by freshwater drainage effects (e.g., flooding, elevated nutrient loading, and pollutant discharge), and by extreme climate events (e.g., freezing air temperatures, drought, and hurricanes) (Michener et al. 1997, Erwin 2009). For example, mangroves (*Rhizophora* spp.) are halophytes that thrive along tropical coastlines reaching latitudinal limits along the northern Gulf Coast in Texas, Louisiana, and Florida. Historical lapses in freeze events and extreme drought events may account for the northward establishment of red mangrove, which are cold sensitive (Montagna et al. 2009). Warming sea and surface temperatures under predicted climate change scenarios will likely increase the frequency and severity of drought episodes in western parts of the Southeast (Caldwell et al. 2012), while decreasing the periodicity of hard freezes that cause dieback of frost-intolerant tropical plant species (Montagna et al. 2009). Mangrove populations have persisted in fringe populations along subtropical coastal settings of Texas, Louisiana, and Florida but have been undergoing recent expansion in latitudes above the tropical Everglades region, where mangroves traditionally have dominated the coastal land margin (Michot et al. 2010, Doyle et al. 2010). Local populations of black mangrove *(Avicennia germinans)* in coastal Louisiana have expanded in area, density, and stature since the last damaging freeze two decades ago (Michot et al. 2010). If the period between severe freeze events lengthens under climate changes, mangrove expansion is expected to succeed landward and poleward along the northern Gulf Coast changing the proportion of saltmarsh area (Krauss et al. 2008). Mangroves have the added benefit of possessing unique root structures that may help stabilize coastal areas from erosion (McKee et al. 2007, Cherry et al. 2009). A shift from saltmarsh dominated coastlands to mangrove dominated shores, due to climatic changes, may also lead to shifts in fish species present (Ley et al. 1999), and reductions in some bird populations (e.g., brown pelican, *Pelecanus occidentalis*) (Visser et al. 2005).

Climate change poses some immediate and long-term threats to the health, function, and biodiversity of tidal wetlands along the coastal margin of the SE USA. Tidal forests of the Gulf Coast and elsewhere have been undergoing dieback and retreat from sea-level rise during the 20th century (Montagna et al. 2009). This trend is expected to continue or be exacerbated under projected increases of global sea level rise (Montagna et al. 2009). Coastal ecosystems of the western Gulf of Mexico are even more vulnerable due to the high rates of land subsidence that drive relative sea level rates that equal or exceed high Intergovernmental Panel on Climate Change (IPCC) projections for accelerated global sea level rise expected with climate warming during the 20th century (Doyle et al. 2007 and 2010). In all coastal counties and region-wide, sea level rise of any rate or origin, relative or eustatic, is expected to cause widespread loss or retreat of coastal forests as dictated by local environmental settings (Doyle et al. 2010). Mangrove forests that dominate tropical shores of southern Florida are expected to migrate inland with increasing sea level and increase the proportion of forested habitat in coastal areas.

Mississippi Alluvial Plain and adjacent regions. The Mississippi Alluvial Plain (MAP) forests, which extend up north to southern Illinois and Kentucky, west to Tennessee; and into the western Gulf Coastal Plain, are similar to those of the Atlantic and Eastern Gulf Coastal Plain forests but can include different levels of nutrients and soil types. Alfisol soils, which are more fertile than highly-weathered clay Ultisol soils or sandy Entisols, are common along the alluvial plain of the Mississippi River as well locations in Alabama (Christensen 2000). Seasonal temperature variations increase away from the coast and frost-free growing season durations decline appreciably from south to north. Although covered more extensively in Natural Ecosystems (Chapter 11), the freshwater swamp forests of the MAP in Louisiana are particularly threatened by a combination of drought and intrusion of saltwater triggered by drought conditions (Hoeppner 2008). Drought also has been linked to increased fire frequency and size in Mississippi, particularly in counties dominated by pines in the southern part of the state (Grala and Cook 2010). The importance of drought for this region is underscored by paleo-ecological work examining extended drought impacts during the mid- to late-Holocene period including the Medieval Warm Period (approximately 800 to 1200 CE) that characterized much of the Northern Hemisphere. During these times, vegetation loss was severe enough to coincide with the formation of low mounds and dunelike features that characterize much of the currently forested regions in the south-central USA today (Seifert et al. 2009).

8.3 Changes in Forest Type Across the South

The forests of the SE USA are currently highly diverse but they are not necessarily stable under a changing climate (for discussion of projected changes see Chapter 2). The potential savannafication of the SE, in which forests are converted into more open woodlands due to a combination of hotter and drier conditions, could be one of the most profound potential climate change impacts in the USA. Predictions for the SE include emergence of savanna ecosystems (Hansen et al. 2001, Bachelet et al. 2001), with expansion of Coastal Plain species into the Piedmont and Appalachians (Iverson et al. 2008). However, the SE is also expected to have future climates and vegetation compositions that are currently not found within the region (Williams and Jackson 2007). The combination of future climate, soils, and land cover may not resemble anyplace currently within vegetation dispersal distances (Williams and Jackson 2007). Current Coastal Plain climates are most similar to those expected for the Piedmont, but this region differs in soils, hydrology, and historical fire frequencies (Christensen 2000). Clay soils of the southeastern mountains and Piedmont are more similar to each other than those of the sandy Coastal Plain, and it is unclear how species may shift distributions in response to changes in SE climates.

Climate envelope models use the climate where a species occurs today to predict where suitable climates will likely occur in the future. However, climate envelop models themselves do not predict the future locations of tree species, as they do not account for rates of migration, habitat fragmentation, and other issues (Iverson et al. 2008). Genetic evidence suggests late Quaternary and early Holocene migration of trees species following the last ice age likely occurred at much slower rates than what would be required to keep pace with current and future climate change (McLachlan et al. 2005, Anderson et al. 2006, Mohan et al. 2009). Molecular work using chloroplast DNA suggests these paleo-rates were much less than 100 m per year, yet current global temperatures are shifting poleward at rates exceeding 1 km per year (McLachlan et al. 2005, Anderson et al. 2006). Migration rates of plant populations depend largely on rare long-distance seed dispersal events (LDD) which may not be frequent enough to result in the rapid migrations needed to keep track with species' current climates. Successful seedling recruitment and colonization after LDD is further limited by successful germination, growth, and survival (Ibáñez et al. 2007, Mohan et al. 2009). Recent work suggests that 59% of the 92 tree species examined were exhibiting range contractions at both the northern and southern boundaries (Zhu et al. 2011). Only 21% of eastern temperate tree species were shifting ranges northward, and 16% were shifting ranges southward (Zhu et al. 2011). This is in contrast to the expectation that juvenile trees of the eastern USA may currently be expanding northward in response to warming over the last several decades (Zhu et al. 2011).

Climate effects on canopy tree mortality rates are highlighted in work by Lines et al. (2010). Using data from across the eastern USA they found that tree mortality was six to nine times lower in areas with an intermediate temperature range (8°C to 10°C) compared to those areas with higher or lower temperatures. Mortality increased with increasing temperatures for species that currently exist in a range where average annual air temperature ranges between 10°C to 15°C. Areas with mean annual temperatures of more than 15°C, which currently includes much of the southeastern Piedmont and most of the Coastal Plain, exhibited much higher rates of tree mortality, suggesting that overall tree survivorship may decline with warmer temperatures. Therefore, northern parts of the SE may also see sharp increases in forest decline with increasing annual temperatures associated with regional warming. Conversely, historical tree mortality was minimized at intermediate amounts of annual precipitation, but mortality rates increases were much greater where annual precipitation was lowest. Therefore, future shifts in precipitation patterns within the region could also impact forest mortality.

8.4 Current and Projected Forest Stresses

Expansion and contraction in forest range and survivorship are often not directly a function of climate or climate change, but indirectly a function of climate impacts on other stressors such as insect populations and wildfire. Drought may weaken a forest, but it may be another biotic or abiotic factor that is the actual cause of death (McNulty and Boggs 2010). Forests in the southeastern USA are characterized by frequent natural disturbances such as fire, wind and ice storms, drought, insects and disease (Dale et al. 2001). Under a changing climate, many of these disturbances are projected to continue and may be amplified by climate change, and a series of disturbances may be required to significantly impact forest mortality. The major types of disturbance across the southeastern USA are outlined in the following sections.

Wildfires

The SE contains some of most productive forest land in the USA (Wear et al. 2007). As forest productivity increases so does fuel for wildfire. The combination of favorable climate and abundant fuel loads create a high fire-return rate of three to five years (Stanturf et al. 2002). The SE leads the nation in number of wildfires per year. The region averaged approximately 45,000 fires per year from 1997 through 2003 (Gramley 2005). Climate change may increase the frequency and intensity of wildfires (Blate et al. 2009).

Wildfires can lead to severe environmental consequences. Emissions from wildfires are an important source of atmospheric carbon. Furthermore, smoke particles are a source of atmospheric aerosols, which affect atmospheric radiative transfer through scattering and absorbing solar radiation and through modifying cloud microphysics (Charlson et al. 1992). These processes can further modify clouds and precipitation and atmospheric circulation (Ackerman et al. 2000, Liu 2005a and 2005b). In addition, wildfires release large amounts of particulate matter (PM) and other air pollutants that can degrade air quality (Riebau and Fox 2001). Wildland fires contribute an estimated 15% of total PM and 8% of CO_2 emissions over the southeastern USA (Barnard and Sabo 2003).

Weather and climate are determinants for wildfires along with fuel properties and topography (Pyne et al. 1996). Fire activities vary from one fire season to another. Fire weather and climate influence wildfire behavior and account for fire variability at various time scales. Under warm and dry conditions, fire seasons become longer and fires ignite more easily and spread more quickly. There is evidence that wildfires, especially catastrophic wildfires, have increased in recent decades in both the USA and other parts of the world (Piñol et al. 1998, Westerling et al. 2006). Among the converging factors were extreme weather events such as extended drought and climate change (Goldhammer and Price 1998, Stocks et al. 2002). Many climate models have projected significant climate change by the end of this century due to the greenhouse effect (IPCC 2007), including an overall increase in temperature worldwide and a drying trend in many subtropical and mid-latitude regions. Thus, wildfires likely will increase in these regions. Fire potential will increase significantly in several global geographic regions, including some areas in the USA (Liu et al. 2009).

Climate change may have various impacts on fires in the SE. Temperature is projected to increase across the South and would contribute to increased fire frequency and intensity, total burned area and longer fire seasons. In addition, temperature change can indirectly impact fires by changing fuel conditions. Increased temperature will reduce fuel moisture due to increased evaporation and, therefore, increase the threat of wildfires. The impact of climate change on fuel loading is more complex. Increased air temperature can increase fuel loading if the growing time is lengthened and there is sufficient soil moisture for tree growth. However, if increased air temperature also reduces soil moisture, tree productivity and fuel loading could *decrease* despite the extended length of the growing season.

The contributions of precipitation and humidity are also complex. Projections for precipitation are less certain than those for air temperature. Projected precipitation change often shows no clear trends even over large areas, including the southeastern USA (McNulty et al. 2012). Model agreement over projected precipitation decrease is higher in many subtropical and mid-latitude ecosystems outside the SE. This reduced precipitation would reduce fuel moisture and therefore increase fire potential in these regions. However, precipitation reduction would reduce available water for plant growth, leading to less fuel and therefore lower fire potential. Nevertheless, most GCMs also project more frequent precipitation anomalies such as drought that in turn could increase fire risk.

Hurricanes

Hurricanes, which are tropical cyclones with sustained winds equal to or greater than 119 km per hour, can cause massive economic damage to forests (see chapter 2 for more detail on future hurricane projections). In 2005, Hurricane Katrina heavily damaged forests along the Louisiana and Mississippi Gulf coasts (Chambers et al. 2007, and Stanturf et al. 2007). McNulty (2002) estimated that a single Category 3, 4, or 5 hurricane can destroy the equivalent of 10% of the annual carbon sequestered in the USA. Owing to its size, intensity and trajectory, Hurricane Katrina may have had 6 to 14 times that impact (Chambers et al. 2007). In 2005, winds from Hurricane Katrina damaged 22 million m^3 of timber estimated at a value of $1.4 billion to $2.4 billion dollars. Impacts are not limited to loss of wood volume and quality; ecosystem services provided by these forests also can also be impaired.

There are four main factors that determine the extent and severity of wind damage on forests: climate, soils, topography, and stand conditions (Wilson 2004). Hurricanes obviously represent an extreme climatic event. Trees growing in soil conditions that restrict root growth and depth are consistently more prone to uprooting. Variation in wind-throw along topographical gradients is more complicated and is often confused with damage due to species type and soil variation. There are many stand attributes that help determine tree susceptibility to wind-throw. These include height to diameter ratios, height, spacing, recent thinning, and impacts of previous disturbance on creating exposed edges that contain trees more vulnerable to wind-throw. Tree species composition may also impact the degree of damage from hurricanes. Therefore, stand composition and stocking levels represent stand attributes that can be manipulated by forest managers to reduce hurricane impacts.

Some evidence suggests that longleaf pine (*Pinus palustris*) might also be more tolerant to high winds than either slash pine (*P. elliottii*) or loblolly pine (*P. taeda*). In a study of the Hobcaw Forest, in coastal South Carolina after Hurricane Hugo, Gresham et al. (1991) reported that longleaf pine suffered less damage than loblolly pine. It was noted that species native to the Coastal Plain may be adapted better to the disturbance regimes found there. For example, longleaf pine, baldcypress (*Taxodium distichum*), and live oak (*Quercus virginiana*) suffered less damage than forest species with broad distribution ranges Gresham et al. (1991).

Johnsen et al. (2009) found that following hurricane Katrina, longleaf pine suffered less mortality (7%) than loblolly pine (26%). In addition to being potentially more resistant to wind-throw, longleaf pine is also more drought and fire resistant than the commonly planted loblolly pine (Landers et al. 1995). Wind damage increases with tree size, but the frequency and severity varies with species, site, wind parameters, and stand characteristics, specifically canopy evenness and age distribution, making it difficult to distinguish those tree species that appear to be more or less susceptible to wind damage (Gresham et al. 1991). The southeastern USA Coastal Plain is highly prone to hurricane events (Stanturf et al. 2007), and intense hurricanes occur two out of every three years across the region (McNulty 2002). Similar to historical natural fire regimes, the selection pressure of frequent high velocity winds has been a driving factor in forest composition.

Insects

Many types of insects damage southeastern forests, but the southern pine beetle (*Dendroctonus frontalis Zimm*.) is the most commercially destructive. Southern pine beetles caused more than $900 million in damage to SE pine forests between 1960 and 1990. Higher winter air temperatures are expected to increase over-wintering beetle larva survival rate, and higher annual air temperatures are expected to allow the beetles to produce more generations per year (Ayres and Lombardero 2000). Both of these factors could increase beetle populations. Other climate changes may work to reduce beetle populations. On the one hand, field research has demonstrated that moderate drought stress can increase pine resin production and, therefore, reduce the colonization success rate of the beetle (McNulty et al. 1998). However, severe drought stress reduces resin production and greatly increases the susceptibility of trees to beetle infestation (McNulty and Boggs 2010).

In addition to length and timing of the breeding season, other factors will likely impact the amount of insect caused damage under future climate conditions including the minimum winter air temperature and the prompt removal and destruction of infected timber (Rodriguez 1966). However, another factor closely linked to climate change may also impact insect success. Although it is one of the principle drivers of rising global air temperatures, CO_2 also increases forest productivity. Gan (2004) used an ecosystem model in conjunction with climate scenarios to predict that climate change would increase forest production by more than 7% during this century. The increase in productivity was a function of increased air temperature, longer growing season, and elevated atmospheric CO_2. However, southern pine beetle damage is also projected to increase by 4 to 7 times current levels, which would cause damage estimated at $500 to $800 million year per year (Gan 2004).

Potentially, some of the challenging impacts of climate change will be those conditions for which we have not considered or prepared for, such as previously unobserved combinations of environmental conditions that interact in new and unique ways. This concern is not unique to science. One such event occurred in the high elevations red spruce (*Picea rubens*, Sarg.) forests of western North Carolina. From 1999 until 2002, the area around Mt. Mitchell was in a period of extended heat wave and drought (McNulty and Boggs 2010). This southerly section of the Appalachian Mountains received some of the highest rates of acidic deposition in the eastern USA and contain remnant species present from the last glaciation, such as red spruce or hemlock that may be most at risk of extirpation. In 2001 some of the red spruce stands in the area began to die in large numbers while other stands of red spruce survived within the area (Figure 8.2). An examination of the sites found that stands with predominantly live trees and sites with predominantly dead trees had very different site characteristics. The sites with predominantly dead trees had much faster growth rates and higher soil nitrogen concentrations prior to the drought, compared to the historically slower growth rates and lower soil nitrogen concentrations from sites that largely survived the drought. In addition to the drought, there were signs that all the sites were attacked by southern pine beetles, a species that does not normally inhabit high elevation areas (Williams and Liebhold 2002). All the trees were attacked, but the trees that survived successfully repelled the phloem eating beetles. These were the trees from the poorer quality (i.e., lower soil nitrogen content) sites. Conversely, the trees that were unable to repel the beetles came from the higher quality (i.e., high soil nitrogen content) sites. The authors suggested that those factors that allowed stands to have the most vigorous growth under average climatic conditions also made these stands the most susceptible to mortality once those conditions changed. In combination, insects, drought, and nitrogen deposition ultimately combined to cause the observed forest mortality. If any one of these factors were not present, the trees may not have died. While in retrospect, the mechanisms for decline seem clear, forest managers have historically not been taught to consider vigorous forest stands as unhealthy. However, under a changing climate the definition of forest health, resilience, and resistance may need to be reevaluated (Thompson et al. 2009).

Elevated Atmospheric CO_2

Although CO_2 is not considered a disturbance factor for forests, atmospheric CO_2 could impact forest structure and function. Atmospheric carbon dioxide (CO_2) levels have increased nearly 35% since preindustrial times, from about 280 ppm to more than 380 ppm (IPCC 2007). Depending on the growth and emissions scenario used, atmospheric CO_2 may rise as high as 850 ppm by 2100 (IPCC 2007).

Figure 8.2 Red spruce (*Picea rubens Sarg.*) mortality in western North Carolina due to a combination of drought, southern pine beetles, and acid rain.

While carbon dioxide is the primary driver of anthropogenic climate change, it is also the basis of plant photosynthesis. Given that plant photosynthesis is not saturated at current CO_2 levels, anthropogenic increases in CO_2 will almost certainly lead to higher rates of photosynthesis if sufficient soil nutrients are available to support the elevated CO_2 induced growth (Oren et al. 2001). However, greater photosynthesis may not translate to significantly greater forest productivity and plant carbon storage, and gains in productivity may not be sustainable over the long term (Norby et al. 2010).

8.5 Ecosystem Services

Southeastern forests have been a major source of ecosystem goods and services for thousands of years (Anderson and Sassaman 1996). Current changes in demographics and climate may change the value of and need for some ecosystem services, but an overall reliance on southeastern ecosystems for societal and economic purposes will remain. In addition to goods and services such as timber and protection of water supplies, southeastern forests are considered important sinks for atmospheric CO_2, and part of a strategy to slow global warming. These services are outlined in the next section.

Forest Productivity and Carbon Sequestration

Large areas in the SE are actively managed for wood production at varying levels of intensity. For example, site preparation; weed control; fertilization; stocking, such as planting density and thinning; and genetic improvement can all impact forest productivity. Attention is being focused on the role forests play in sequestering some of the anthropogenic carbon inputs to the atmosphere in biomass and soils, while conserving existing carbon stocks through informed resource management (Blate et al. 2009).

The role of southeastern forests in providing a steady supply of timber and fiber is of particular importance in meeting current and future timber and fiber needs across the USA because forest harvests have substantially decreased across the other regions. As a whole, the South's forest sector produces approximately 60% of the total wood production in the USA (Prestemon and Abt 2002).

Climate change has the potential to impact forest productivity and carbon sequestration. Increases in forest carbon sequestration (a result of forests storing carbon in soils and woody tissues) can slow down the rate of atmospheric CO_2 increase and therefore help to slow down global warming. Southeastern forests also have been estimated to account for 36% of the carbon sequestered in the conterminous United States (Turner et al. 1995). Han et al. (2007) estimated that each year forests in the SE sequester 13% of regional greenhouse emissions in soils and long-lived forest products, such as lumber. Southeastern forests also contain about 30% of the nation's carbon stock (Mickler et al. 2004) and play a prominent role in the regional and global carbon cycle (Turner et al. 1995).

Forest Water Resources

When compared with other land uses, managed and unmanaged forests provide the cleanest and most stable water supplies for drinking water, recreation, power generation, aquatic habitat, and groundwater recharge. Large acres of forestland in the Appalachians and Piedmont are the headwaters of many river systems in the SE (Sun et al. 2011). These watersheds provide a disproportionately higher amount of the regional water supply than the Coastal Plain because these forests occupy areas with relatively high precipitation and low evapotranspiration (Brown et al. 2008).

The impacts of climate change on forest structure and functions are likely to result in negative consequences on water quantity and quality of forested watersheds through altering key hydrologic fluxes including precipitation and evapotranspiration, and the biogeochemical processes (Sun et al. 2011). An increase in air temperature means an increase in energy availability and atmospheric water demand. Thus for the humid southeastern USA, water shortages are expected to increase. For example, Walter et al. (2004) concluded ecosystem ET has been increasing at a rate of 10.4 mm per decade across six major basins that cover a majority of the watersheds in the USA. As more water is evapotranspired form the soil, less water will flow through the soil, and into streams and rivers. There will also be less water recharging shallow aquifers as tree water use (i.e., ET) increases with increasing air temperature (see Chapter 10. for more details on how forests use and yield water).

Shifts in tree species due to changes in climate, fire regime, and invasive species are likely to increase ecosystem transpiration rates and alter the carbon and nutrient balances. An increase in frequency of high intensity storm events will increase rainfall erosivity thus the potential for increased soil erosion and sedimentation (Marion et al. In press). An example of this increased soil erosion potential was forecast for the

Uwharrie National Forest where severe soil erosion was predicted to increase significantly under future climate changes (Figure 8.3).

Ecosystem model simulations and multiple watershed vegetation manipulation experiments suggest that activities that do not result in a forest type conversion or a coppice stand structure will not substantially alter streamflow responses to extreme precipitation events (Ford et al. 2011). However, based on forest conversion experiment studies, the conversion of deciduous forests (either naturally or by forest management) to pine monocultures in the Appalachians substantially altered the streamflow response to extreme annual precipitation. The pine increased soil permeability and rain fall absorption, but the pines also use more soil water than do the hardwoods. Thus, forest management may reduce flood risk but also exacerbate drought. Tradeoff between managing forests for opposite extremes should be carefully considered by water resource managers for contingency land use planning (Ford et al. 2011).

Increased frequency of heavy rainfall events will likely impact forest communities and increase flood occurrence. If there is an increase spring and summer droughts, it likely will make forest vegetation vulnerable to stresses due to high ET demands in the Coastal Plain region. Forests can also modulate regional climate by controlling energy and water transfers between the atmosphere and forested land-surface (Liu 2011). Forest restoration, afforestation, or both are expected to play important roles in mitigating the impacts of climate change on water resources in these regions.

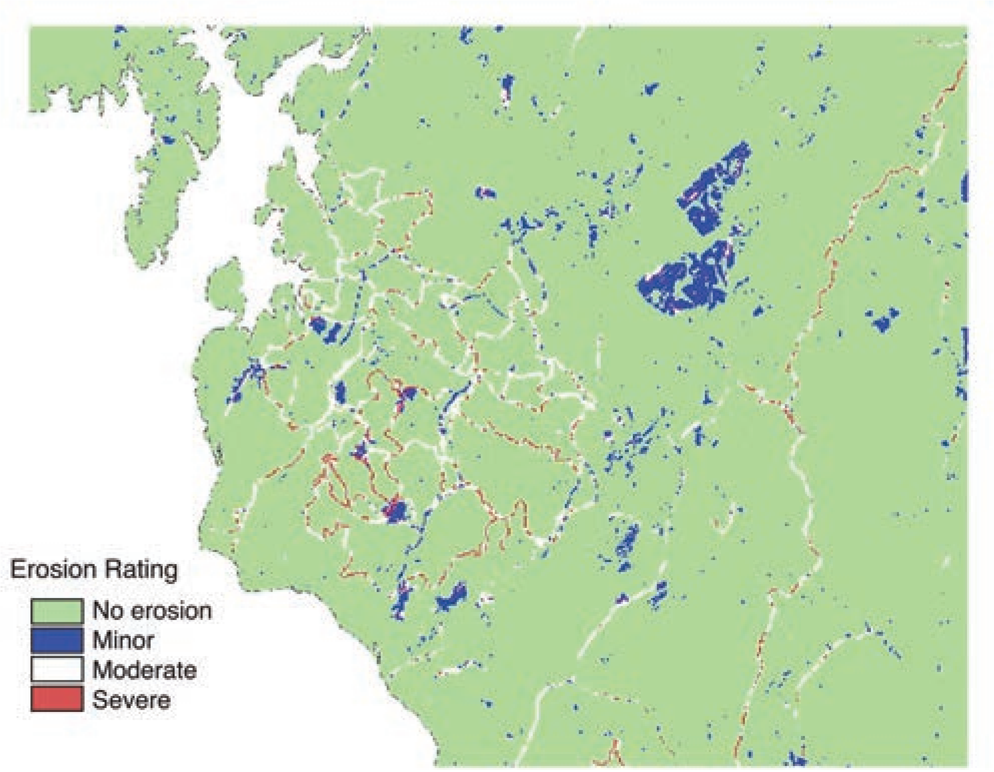

Figure 8.3 Revised Universal Soil Loss Equation predictions of soil erosion areas within the Uwharrie National Forest by 2030.

Regional modeling with a monthly scale water supply and demand model called the Water Supply Stress Index (WaSSI) suggests ecosystem water stress across the eastern USA will likely increase in the next 50 years, especially during the summer and fall seasons, due to increase water demand and reduced water yield (Caldwell et al. 2011).

8.6 Adaptation and Mitigation Options

In general, the biological productivity of SE forests likely will be enhanced by atmospheric carbon enrichment, as long as precipitation does not decline or air temperature does not increase soil moisture stress to a level that would offset potential CO_2 benefits on productivity. Use of forest resources is also anticipated to adapt to changes in productivity (de Steiger and McNulty 1998). For instance, a northward shift in forest productivity (Figure 8.4) is projected to lead to relative increases in the proportion of regional timber harvests that come from the northern reaches of the region. This may compensate for harvest reduction in the southeastern parts, which are projected to be more negatively affected by the biophysical effects of climate change. In addition, landowners are projected to switch land between forestry and agricultural in places and at times where the change in relative productivity warrants it.

There are a variety of other adaptation strategies to address climate trends and extremes. Potential adaptation strategies include genetic and silvicultural system improvements that increase water use efficiency or water availability. Increasing

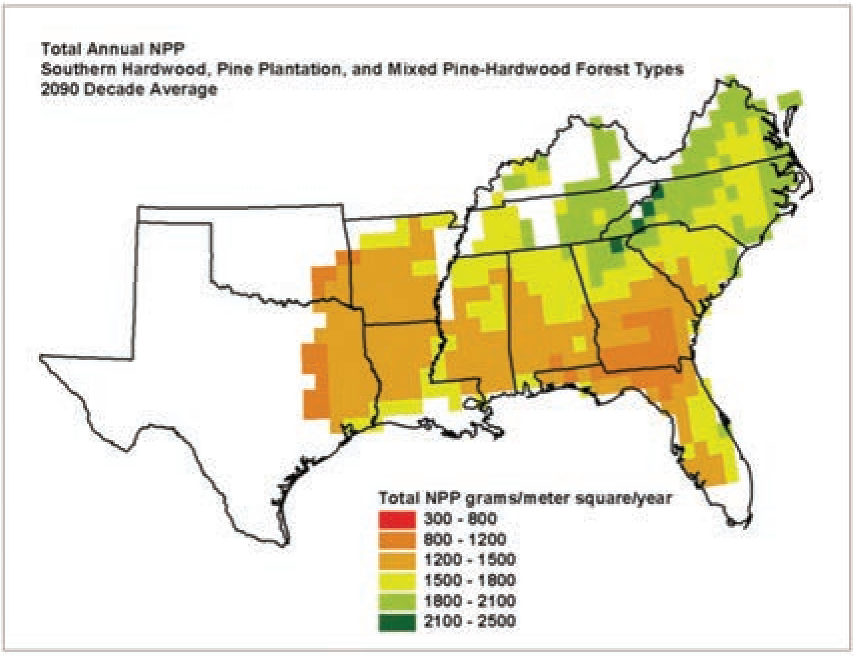

Figure 8.4 Forest model predictions of increased carbon sequestration (measured at net primary productivity, NPP) in the northern sections of the southern USA due to increasing air temperature by the end of this century.

knowledge of the role of fire, hurricanes, droughts, and other natural disturbances will be important in developing forest management regimes and increasing stand productivity in ways that are sustainable over the long term. Under a hotter, drier climate, an aggressive fire management strategy may prove important in this region (Dale et al. 2001). Timber productivity associated with increased temperature, growing season length, and CO_2 enrichment may be further enhanced by improved genetics, bioengineering, use of marginal agricultural land for tree production, and more intensive forest management (Schmidtling et al. 2004, Oren et al. 2004). Reduction of air pollutants, such as ozone and nitrogen oxides, may also be an important strategy for increasing forest productivity due to the potential for synergistic stress impacts (McNulty and Boggs 2010, Figure 8.5).

Increased use of fertilizers may increase forest productivity and carbon sequestration in an effort to partially mitigate greenhouse gas emissions. More than 400,000 ha of pine plantations are now fertilized each year with nitrogen which increases forest productivity (Albaugh et al. 2007). Fertilization can also decrease carbon losses by reducing soil respiration, and thus increasing forest carbon sequestration (Butnor et al. 2003). Other management tools that directly impact carbon sequestration include species selection, modification of initial planting density, and rotation length and thinning.

The effects of silvicultural treatments, such as planting density, thinning and rotation length, on carbon sequestration were analyzed by simulating carbon flux under

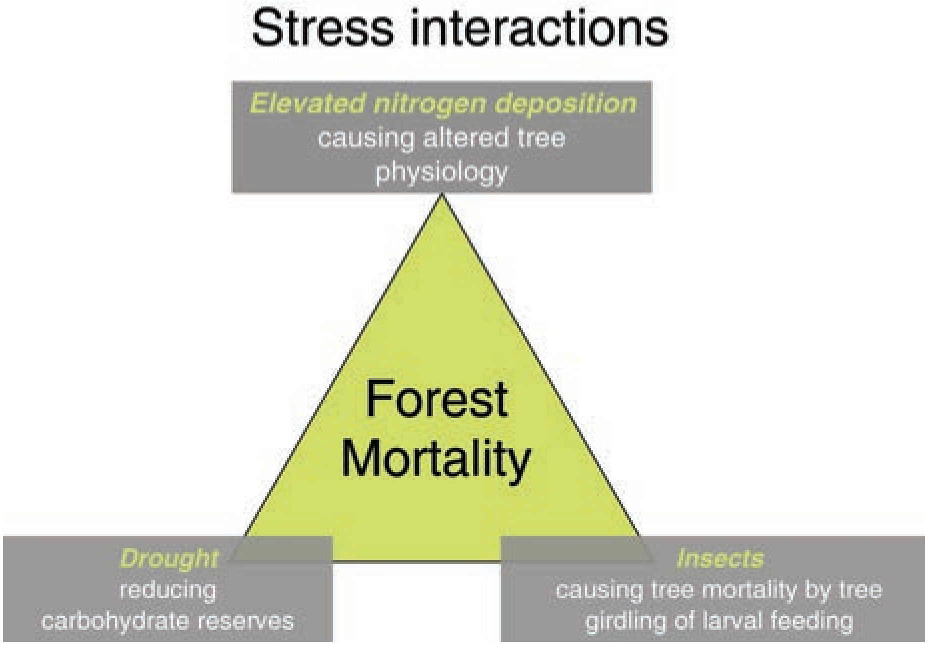

Figure 8.5 Interactions of climate (e.g., drought), biological (e.g., insects) and abiotic (e.g., fire or acid rain) can combine to cause forest mortality. The interactive stresses may be related (e.g., drought and fire) or unrelated (e.g., drought and acid rain). Any single stress may not have caused the mortality, but as climate change continues the potential for more frequent, more severe, and synergistic stress increases.

different climate and management scenarios for loblolly pine and slash pine plantations established in the southeastern USA Lower Coastal Plain (Gonzalez-Benecke et al. 2010 and 2011). Increasing the rotation length increased carbon stock in both species. Canadell and Raupach (2008) cited longer harvesting cycles as a major management strategy for increasing forest carbon stocks.

Improved understanding of climate change impacts and adaptation options are only useful if this information can be conveyed to the land manager. New web-based models and tools are being developed to allow for easier, more site specific climate change assessments. For example, the web-based Distrib/Shift forest species distribution model gives users the ability to examine which tree and bird species will likely become more and less dominant in that area over the coming years and decades (Iverson et al. 2011). Similarly, the web-based WaSSI (Water Supply Stress Index) hydrologic model gives land managers the ability to examine the impacts of climate, population and land use change on water supply and demand on their watersheds. Finally, Web-based tools like TACCIMO (Template for Assessing Climate Change Impacts and Management Options) allow the user to search scientifically reviewed literature on climate change impacts for their area, and then to further use TACCIMO to search for management options to address or adapt to these changes. Significantly improved graphic user interfaces (Figure 8.6), data storage, and internet access speeds have greatly improved the application of these tools.

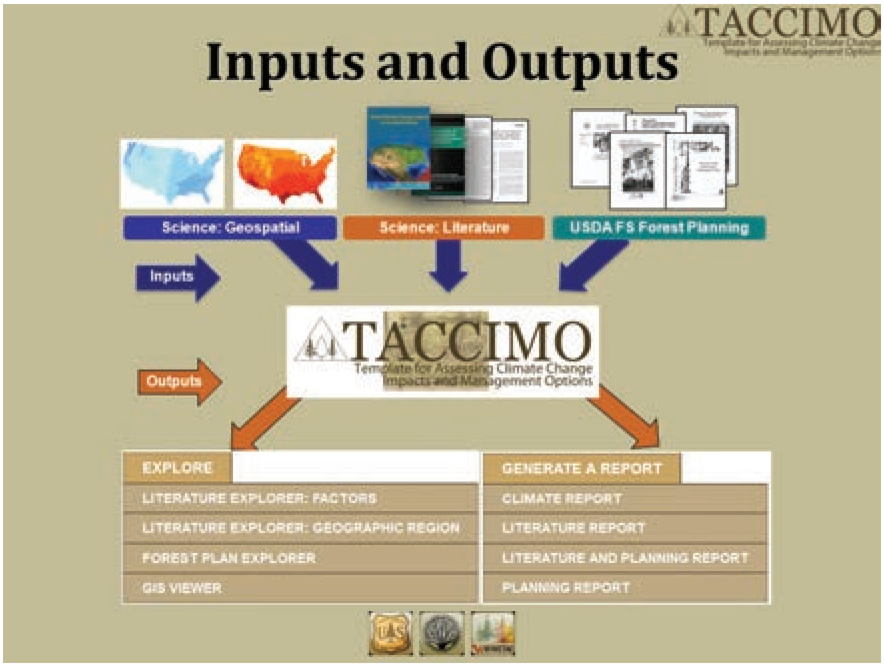

Figure 8.6 Web-based tools such as TACCIMO (Template for Assessing Climate Change Impacts and Management Options) are increasingly being used to easily translate scientific knowledge into the hands of the land manager.

8.7 Conclusions

Southeastern forests are as diverse as the cultures that exist within them. The wide range of tree, plant, and animal species make the region both resistant and susceptible to change. Some species will not be able to adapt to rapidly changing climatic conditions; other species will fill vacated niches that develop. Protecting the overall integrity of the ecosystem will be less of a challenge than protecting all of the parts. Several independent studies suggest that remnant species present from the last glaciation, such as red spruce or hemlock may be most at risk of extirpation. If changes result in warmer, drier conditions in some parts of the SE, conditions could favor more drought-tolerant species such as oaks and long-leaf pine.

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Chapter 8

Forests and Climate Change in the Southeast USA

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The forests of the southeastern USA have seen many changes over the past 200 years. From cutting of the virgin forest in the 19th century to the expansion and later contraction of farming and the rise of plantation forestry in the 20th century, the structure and function of southern forests continues to evolve. Climate change represents another factor that is changing Southeast (SE) forests and forestry practices. Current and continued projected warming likely will increase the risk of wildfire, wind, insect, and disease damage to southeastern forests. Increased air temperatures also likely will lead to decreased forest water yield, even as the population of the SE USA continues to expand. Conflicts between maximizing forest carbon sequestration as a mitigation strategy for climate change and need for water likely will continue well into the 21^{st} century. However, scientists are actively working with land managers to address these issues. Although the form of SE forests will continue to change due to old and new pressures, forest managers are becoming increasingly better prepared to cope with these challenges. This chapter examines some of the pressing issues and management options associated with global change in the SE USA.

Key Findings

- Warming air temperatures likely will increase regional drying through increased forest water use via evapotranspiration (ET) regardless of changes in precipitation, and this drying will likely increase wildfire risk across SE USA forests.
- Longer growing seasons will likely increase the risk of insect outbreak and very likely will expand the northern range of some species, such as the southern pine beetle.
- Under most scenarios, increasing temperatures and decreasing precipitation will result in a greater uptake of soil water by forests and lead to reductions in streamflow.
- Despite climate and land use changes, forests in the southeastern USA will likely continue to provide a sink of atmospheric carbon dioxide (CO_2).
- The potential savannafication of the SE, in which forests are converted into more open woodlands due to a combination of hotter and drier conditions, could be one of the most profound potential climate change impacts in the USA

8.1 Historical Perspective

The forests of the Southeast (SE) USA have seen extensive change during the past century. Currently, 60% of the SE landscape is forested (Wear and Greis 2002). In 1860, about 43% of the SE land area was reported as under cultivation, but a substantial part of the farm holdings that remained in forest were used for grazing livestock (Smith and Darr 2009). Timberland continued to decline until the early 1920s due to the continued expansion of settlements. Significant changes in agriculture took place after 1920 that resulted in abandonment of large areas of crop and pasture lands. Some of the abandoned land was planted with trees, but most of the land reverted naturally to forest, leading to increases in timberland acreage (Wear et al. 2007). By the late 1950s and early 1960s, decline of timberland began again in the SE, caused primarily by the clearing of forests for soybean and other crop production. Much of this timberland reduction occurred in bottomland hardwood forest areas of the Mississippi Delta.

Throughout the 1970s, timberland was cleared for agricultural use and for an expanding export market. The decade beginning in 1982 marked a slowing of forest cover loss with the National Resources Inventory reporting roughly a half million-acre loss (less than 1%) in forestland in the SE (Wear et al. 2007). That trend has continued into the 21st century as softwood pulp prices have fallen by 50% since 1998, and the forest products industry divested approximately 75% of its timberland holdings (Butler and Wear, In Press). Although market prices will likely continue to be a driving factor in forest land area, other ecosystem services such as climate change, wildlife protection, drinking water supply, and recreation may increasingly influence the distribution and composition of SE forests (Wear and Greis 2002).

8.2 Southeastern Forest Types

The southeastern USA is not comprised of a single forest type, but of many. This assessment of forests and climate change focuses on six distinct forest areas within the SE: the Atlantic and East Gulf Coastal Plain, Piedmont, Appalachian/Cumberland, Mid-South, Coastal, and the Mississippi Alluvial Valley. Current inventory data shows that more than 30 million hectares of upland hardwood forests dominate the SE, followed by more than 15 million hectares of planted pine, approximately 13 million hectares of natural pine and bottomland hardwoods, and more than 3 million hectares oak-pine forest types (Butler and Wear 2012). These forest ecosystems provide a multitude of goods and services including clean water and air, wildlife habitat, recreation and aesthetics, timber and fiber production, and CO_2 sequestration. This chapter reviews current and future stresses on services provided by SE forests, and examines how forest management could be used to cope, adapt, or mitigate negative impacts.

Atlantic and East Gulf Coastal Plain. Historically, most of the southeastern Coastal Plain was dominated by fire-dependent longleaf pine (*Pinus palustris*) savannas (Christensen 2000). However, upland closed-canopied forests occur in mesic areas protected from frequent fire or where fire suppression has occurred. Notable examples of old-growth mesophytic beech-magnolia forests are present in the Apalachicola National Forest of the Florida panhandle. Other Coastal Plain broadleaved forests include those dominated by southern oak species such as swamp chestnut oak (*Quercus michauxii*), cherry bark oak (*Quercus pogoda*), and live oak (*Quercus virginiana*), as well as hickories (*Carya spp.*) and loblolly pine (*Pinus taed*a). American holly (*Ilex opaca*), spice bush (*Lindera benzoin*), and pawpaw (*Asimina triloba*) are common in the understory and subcanopy (Christensen 2000).

The distribution of current and potential future droughts and associated fire risk varies as does the potential impacts on trees species. Several dendrochronological analyses of Coastal Plain longleaf pine trees demonstrate the impact of growing season drought severity in relationship to reduced tree growth rates, as well as positive impacts of warmer winter temperatures (Bhuta et al. 2009, Henderson and Grissins-Mayer 2009). Less climatic research has been conducted in closed-canopied upland forests, but increasing fire frequencies in these Coastal Plain forests, due to ongoing and potential future droughts, may be a major impact on the distribution of some forests of this region (Wade et al. 2000). However, a recent study by Gruhn and White (2011) examined the northward range expansion of southern magnolia by comparing establishment success with climatic and topographic variables. Although minimum winter temperatures and the number of frost-free days were important determinants of establishment success, precipitation was not.

In addition to drought and fire, Coastal Plain forests and other ecosystems are also particularly vulnerable to hurricanes. Hurricane Isabel in September 2003 damaged 15% of trees, particularly canopy trees, in a maturing hardwood forest of the Virginia Coastal Plain (Prengaman et al. 2008). Hurricane Isabel was only a Category 2 storm, so increased frequencies of Category 4 and 5 hurricanes as a consequence of climate change likely will have even more profound effects (Webster et al. 2005, Knutson et al. 2011). Hurricane Katrina is an example of the damage caused by a strong hurricane (MIFI 2005).

Southern Appalachians. These forests cover much of the high elevation areas of the north-central southern region that includes eastern Tennessee and Kentucky, western North Carolina and Virginia, and northern Georgia. The southern Appalachian forests are some of the most diverse in North America (Clark et al. 2011). Both unique species and commercially important species can be found within the region. The diversity of these forests is controlled by regional and local weather patterns that can be highly variable due to the mountainous terrain (Clark et al. 2011). As with other mountain systems, the high elevation forests of the southern Appalachian ecosystems are at particular risk from a warming climate. A 3^{o}C increase in July temperature would raise climate-elevation bands by about 480m, resulting in the extirpation of the rare red spruce-Fraser fir (*Picea rubens* and *Abies fraseri*) alpine forests growing at the highest elevations in North Carolina and harboring federally threatened animal species, including the North Carolina flying squirrel (Delcourt and Delcourt 1998). Many of the mid-elevation "cove" forests, which are currently dominated by mesic, fire-intolerant tree species, are extremely diverse in terms of canopy trees, spring ephemeral wildflowers, and amphibians. Since the early 1980s this region has warmed and precipitation variability has increased. If these trends continue, they could lead to substantial change in the structure and function of future southern Appalachian forests.

In addition to determining biodiversity, climate variability also controls forest growth. For example, the annual growth rate of five dominant oak species can be severely affected by growing season drought intensity (Speer et al. 2009). During drought years, observed oak forests showed diminished productivity and accumulated 40% less carbon compared to a year of average precipitation (Noormets et al. 2008). If projected temperature increases are accompanied by decreased growing season precipitation, the combined changes may reduce the competitiveness of oaks in the southern Appalachians and elsewhere in the SE (Ibáñez et al. 2008).

Wildfires also shape the structure and function of forests within the southern Appalachians. A recent study suggested that fires occurred fairly frequently over the past 4,000 years in a variety of southern Appalachian forest types including those now dominated by mesic hardwoods, including tulip poplar (*Liriodendron tulipifera*) (Fesenmyer and Christiansen 2010). These researchers found that fire return intervals appear to have been of centuries-scale duration in the time period 4,000 to 1,000 years before present, and were likely often severe. Fires became more frequent approximately 1,000 years ago and were thus likely less severe due to less accumulated fuels build-up. The increased frequency of fire coincided with the occupation by Woodland Tradition Native Americans. If drought and drought-induced fires become more common in the southern Appalachians, fire-tolerant oak and hickory species may become more abundant over less-tolerant tulip poplar, maple (*Acer spp.*), basswood (*Tilia americana*), birch (*Betula* spp.) and magnolia (*Magnolia* spp.) species, potentially reducing diversity in currently highly-diverse mesic forests (Fesenmyer and Christiansen 2010).

Piedmont. The Piedmont region lays southeast of the Appalachian region and stretches from east-central Alabama through central Georgia, northwestern South Carolina, and central North Carolina and Virginia. These forests are dominated by a mixture of pine and deciduous species (Figure 8.1) of high commercial importance (Van Lear et al. 2004). Dale et al. (2010) used ecosystem models and an ensemble of general circulation model (GCM) scenarios to project that in the southeastern Piedmont and Appalachians, southern mixed hardwoods and pine forests on the Piedmont were the most susceptible to changes induced by warmer, and particularly drier, climates. Under the driest of the three climate scenarios considered by Dale et al. (2010), a southern mixed forest transitioned from very high tree species diversity with 14 commonly co-dominant species to very low forest diversity, dominated by loblolly pine, southern red oak (*Quercus falcata*), and Shumard's oak (*Quercus shumardii*). Dale et al. (2010) also found that the less-diverse forests may be more susceptible to insect and pathogen pests, and that hickory (*Carya spp.*) species tended to increase in relative importance under the climate change scenarios considered. Conversely, under those projections the biomass of chestnut (*Quercus prinus*) and black oaks (*Quercus velutina*) tended to decline across Tennessee, as the hickories appeared to be better able to grow in the warmer, drier climate relative to the oak species.

Research from the Duke Free-Air CO_2 Enrichment (FACE) experiment and the Oak Ridge FACE experiment in the southern Appalachians suggests that an approximate doubling of atmospheric CO_2 increases the productivity of the canopy loblolly pine and sweet gum (*Liquidambar styraciflua)* trees by 23% to 27% (DeLucia et al. 2005, Norby et al. 2005). However, when examining the juvenile tree species most likely to comprise the future forests, elevated CO_2 conditions favored the population biomass growth of less productive, shade-tolerant tree species southern sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*) as well as woody vines such as poison ivy (*Toxicodendron radicans*) (Mohan et al. 2006, 2007, and 2008) and exotic Japanese honeysuckle (*Lonicera japonica*) (Belote et al. 2004). So, increased atmospheric CO_2 levels may benefit a variety of species, but it is unclear from these few studies how elevated CO_2 levels coupled with other potential stresses may affect the composition of future forests as a whole.

Coastal wetland forests. Coastal wetland forests exist in the transition between the Coastal Plain and maritime ecosystems and are responsive to changes in climate and freshwater outflow resulting from varying patterns and frequencies of freeze, drought, storm, sea level, and runoff events. Because saltmarshes and mangroves thrive in the

Figure 8.1 Mixed conifer and deciduous Piedmont forest in the southeastern USA.

intertidal zone between land and sea, these systems are expected to undergo the most severe changes from marine effects, such as sea level rise and salinity. They are also affected by freshwater drainage effects (e.g., flooding, elevated nutrient loading, and pollutant discharge), and by extreme climate events (e.g., freezing air temperatures, drought, and hurricanes) (Michener et al. 1997, Erwin 2009). For example, mangroves (*Rhizophora* spp.) are halophytes that thrive along tropical coastlines reaching latitudinal limits along the northern Gulf Coast in Texas, Louisiana, and Florida. Historical lapses in freeze events and extreme drought events may account for the northward establishment of red mangrove, which are cold sensitive (Montagna et al. 2009). Warming sea and surface temperatures under predicted climate change scenarios will likely increase the frequency and severity of drought episodes in western parts of the Southeast (Caldwell et al. 2012), while decreasing the periodicity of hard freezes that cause dieback of frost-intolerant tropical plant species (Montagna et al. 2009). Mangrove populations have persisted in fringe populations along subtropical coastal settings of Texas, Louisiana, and Florida but have been undergoing recent expansion in latitudes above the tropical Everglades region, where mangroves traditionally have dominated the coastal land margin (Michot et al. 2010, Doyle et al. 2010). Local populations of black mangrove *(Avicennia germinans)* in coastal Louisiana have expanded in area, density, and stature since the last damaging freeze two decades ago (Michot et al. 2010). If the period between severe freeze events lengthens under climate changes, mangrove expansion is expected to succeed landward and poleward along the northern Gulf Coast changing the proportion of saltmarsh area (Krauss et al. 2008). Mangroves have the added benefit of possessing unique root structures that may help stabilize coastal areas from erosion (McKee et al. 2007, Cherry et al. 2009). A shift from saltmarsh dominated coastlands to mangrove dominated shores, due to climatic changes, may also lead to shifts in fish species present (Ley et al. 1999), and reductions in some bird populations (e.g., brown pelican, *Pelecanus occidentalis*) (Visser et al. 2005).

Climate change poses some immediate and long-term threats to the health, function, and biodiversity of tidal wetlands along the coastal margin of the SE USA. Tidal forests of the Gulf Coast and elsewhere have been undergoing dieback and retreat from sea-level rise during the 20th century (Montagna et al. 2009). This trend is expected to continue or be exacerbated under projected increases of global sea level rise (Montagna et al. 2009). Coastal ecosystems of the western Gulf of Mexico are even more vulnerable due to the high rates of land subsidence that drive relative sea level rates that equal or exceed high Intergovernmental Panel on Climate Change (IPCC) projections for accelerated global sea level rise expected with climate warming during the 20th century (Doyle et al. 2007 and 2010). In all coastal counties and region-wide, sea level rise of any rate or origin, relative or eustatic, is expected to cause widespread loss or retreat of coastal forests as dictated by local environmental settings (Doyle et al. 2010). Mangrove forests that dominate tropical shores of southern Florida are expected to migrate inland with increasing sea level and increase the proportion of forested habitat in coastal areas.

Mississippi Alluvial Plain and adjacent regions. The Mississippi Alluvial Plain (MAP) forests, which extend up north to southern Illinois and Kentucky, west to Tennessee; and into the western Gulf Coastal Plain, are similar to those of the Atlantic and Eastern Gulf Coastal Plain forests but can include different levels of nutrients and soil types. Alfisol soils, which are more fertile than highly-weathered clay Ultisol soils or sandy Entisols, are common along the alluvial plain of the Mississippi River as well locations in Alabama (Christensen 2000). Seasonal temperature variations increase away from the coast and frost-free growing season durations decline appreciably from south to north. Although covered more extensively in Natural Ecosystems (Chapter 11), the freshwater swamp forests of the MAP in Louisiana are particularly threatened by a combination of drought and intrusion of saltwater triggered by drought conditions (Hoeppner 2008). Drought also has been linked to increased fire frequency and size in Mississippi, particularly in counties dominated by pines in the southern part of the state (Grala and Cook 2010). The importance of drought for this region is underscored by paleo-ecological work examining extended drought impacts during the mid- to late-Holocene period including the Medieval Warm Period (approximately 800 to 1200 CE) that characterized much of the Northern Hemisphere. During these times, vegetation loss was severe enough to coincide with the formation of low mounds and dunelike features that characterize much of the currently forested regions in the south-central USA today (Seifert et al. 2009).

8.3 Changes in Forest Type Across the South

The forests of the SE USA are currently highly diverse but they are not necessarily stable under a changing climate (for discussion of projected changes see Chapter 2). The potential savannafication of the SE, in which forests are converted into more open woodlands due to a combination of hotter and drier conditions, could be one of the most profound potential climate change impacts in the USA. Predictions for the SE include emergence of savanna ecosystems (Hansen et al. 2001, Bachelet et al. 2001), with expansion of Coastal Plain species into the Piedmont and Appalachians (Iverson et al. 2008). However, the SE is also expected to have future climates and vegetation compositions that are currently not found within the region (Williams and Jackson 2007). The combination of future climate, soils, and land cover may not resemble anyplace currently within vegetation dispersal distances (Williams and Jackson 2007). Current Coastal Plain climates are most similar to those expected for the Piedmont, but this region differs in soils, hydrology, and historical fire frequencies (Christensen 2000). Clay soils of the southeastern mountains and Piedmont are more similar to each other than those of the sandy Coastal Plain, and it is unclear how species may shift distributions in response to changes in SE climates.

Climate envelope models use the climate where a species occurs today to predict where suitable climates will likely occur in the future. However, climate envelop models themselves do not predict the future locations of tree species, as they do not account for rates of migration, habitat fragmentation, and other issues (Iverson et al. 2008). Genetic evidence suggests late Quaternary and early Holocene migration of trees species following the last ice age likely occurred at much slower rates than what would be required to keep pace with current and future climate change (McLachlan et al. 2005, Anderson et al. 2006, Mohan et al. 2009). Molecular work using chloroplast DNA suggests these paleo-rates were much less than 100 m per year, yet current global temperatures are shifting poleward at rates exceeding 1 km per year (McLachlan et al. 2005, Anderson et al. 2006). Migration rates of plant populations depend largely on rare long-distance seed dispersal events (LDD) which may not be frequent enough to result in the rapid migrations needed to keep track with species' current climates. Successful seedling recruitment and colonization after LDD is further limited by successful germination, growth, and survival (Ibáñez et al. 2007, Mohan et al. 2009). Recent work suggests that 59% of the 92 tree species examined were exhibiting range contractions at both the northern and southern boundaries (Zhu et al. 2011). Only 21% of eastern temperate tree species were shifting ranges northward, and 16% were shifting ranges southward (Zhu et al. 2011). This is in contrast to the expectation that juvenile trees of the eastern USA may currently be expanding northward in response to warming over the last several decades (Zhu et al. 2011).

Climate effects on canopy tree mortality rates are highlighted in work by Lines et al. (2010). Using data from across the eastern USA they found that tree mortality was six to nine times lower in areas with an intermediate temperature range (8°C to 10°C) compared to those areas with higher or lower temperatures. Mortality increased with increasing temperatures for species that currently exist in a range where average annual air temperature ranges between 10°C to 15°C. Areas with mean annual temperatures of more than 15°C, which currently includes much of the southeastern Piedmont and most of the Coastal Plain, exhibited much higher rates of tree mortality, suggesting that overall tree survivorship may decline with warmer temperatures. Therefore, northern parts of the SE may also see sharp increases in forest decline with increasing annual temperatures associated with regional warming. Conversely, historical tree mortality was minimized at intermediate amounts of annual precipitation, but mortality rates increases were much greater where annual precipitation was lowest. Therefore, future shifts in precipitation patterns within the region could also impact forest mortality.

8.4 Current and Projected Forest Stresses

Expansion and contraction in forest range and survivorship are often not directly a function of climate or climate change, but indirectly a function of climate impacts on other stressors such as insect populations and wildfire. Drought may weaken a forest, but it may be another biotic or abiotic factor that is the actual cause of death (McNulty and Boggs 2010). Forests in the southeastern USA are characterized by frequent natural disturbances such as fire, wind and ice storms, drought, insects and disease (Dale et al. 2001). Under a changing climate, many of these disturbances are projected to continue and may be amplified by climate change, and a series of disturbances may be required to significantly impact forest mortality. The major types of disturbance across the southeastern USA are outlined in the following sections.

Wildfires

The SE contains some of most productive forest land in the USA (Wear et al. 2007). As forest productivity increases so does fuel for wildfire. The combination of favorable climate and abundant fuel loads create a high fire-return rate of three to five years (Stanturf et al. 2002). The SE leads the nation in number of wildfires per year. The region averaged approximately 45,000 fires per year from 1997 through 2003 (Gramley 2005). Climate change may increase the frequency and intensity of wildfires (Blate et al. 2009).

Wildfires can lead to severe environmental consequences. Emissions from wildfires are an important source of atmospheric carbon. Furthermore, smoke particles are a source of atmospheric aerosols, which affect atmospheric radiative transfer through scattering and absorbing solar radiation and through modifying cloud microphysics (Charlson et al. 1992). These processes can further modify clouds and precipitation and atmospheric circulation (Ackerman et al. 2000, Liu 2005a and 2005b). In addition, wildfires release large amounts of particulate matter (PM) and other air pollutants that can degrade air quality (Riebau and Fox 2001). Wildland fires contribute an estimated 15% of total PM and 8% of CO_2 emissions over the southeastern USA (Barnard and Sabo 2003).

Weather and climate are determinants for wildfires along with fuel properties and topography (Pyne et al. 1996). Fire activities vary from one fire season to another. Fire weather and climate influence wildfire behavior and account for fire variability at various time scales. Under warm and dry conditions, fire seasons become longer and fires ignite more easily and spread more quickly. There is evidence that wildfires, especially catastrophic wildfires, have increased in recent decades in both the USA and other parts of the world (Piñol et al. 1998, Westerling et al. 2006). Among the converging factors were extreme weather events such as extended drought and climate change (Goldhammer and Price 1998, Stocks et al. 2002). Many climate models have projected significant climate change by the end of this century due to the greenhouse effect (IPCC 2007), including an overall increase in temperature worldwide and a drying trend in many subtropical and mid-latitude regions. Thus, wildfires likely will increase in these regions. Fire potential will increase significantly in several global geographic regions, including some areas in the USA (Liu et al. 2009).

Climate change may have various impacts on fires in the SE. Temperature is projected to increase across the South and would contribute to increased fire frequency and intensity, total burned area and longer fire seasons. In addition, temperature change can indirectly impact fires by changing fuel conditions. Increased temperature will reduce fuel moisture due to increased evaporation and, therefore, increase the threat of wildfires. The impact of climate change on fuel loading is more complex. Increased air temperature can increase fuel loading if the growing time is lengthened and there is sufficient soil moisture for tree growth. However, if increased air temperature also reduces soil moisture, tree productivity and fuel loading could *decrease* despite the extended length of the growing season.

The contributions of precipitation and humidity are also complex. Projections for precipitation are less certain than those for air temperature. Projected precipitation change often shows no clear trends even over large areas, including the southeastern USA (McNulty et al. 2012). Model agreement over projected precipitation decrease is higher in many subtropical and mid-latitude ecosystems outside the SE. This reduced precipitation would reduce fuel moisture and therefore increase fire potential in these regions. However, precipitation reduction would reduce available water for plant growth, leading to less fuel and therefore lower fire potential. Nevertheless, most GCMs also project more frequent precipitation anomalies such as drought that in turn could increase fire risk.

Hurricanes

Hurricanes, which are tropical cyclones with sustained winds equal to or greater than 119 km per hour, can cause massive economic damage to forests (see chapter 2 for more detail on future hurricane projections). In 2005, Hurricane Katrina heavily damaged forests along the Louisiana and Mississippi Gulf coasts (Chambers et al. 2007, and Stanturf et al. 2007). McNulty (2002) estimated that a single Category 3, 4, or 5 hurricane can destroy the equivalent of 10% of the annual carbon sequestered in the USA. Owing to its size, intensity and trajectory, Hurricane Katrina may have had 6 to 14 times that impact (Chambers et al. 2007). In 2005, winds from Hurricane Katrina damaged 22 million m^3 of timber estimated at a value of \$1.4 billion to \$2.4 billion dollars. Impacts are not limited to loss of wood volume and quality; ecosystem services provided by these forests also can also be impaired.

There are four main factors that determine the extent and severity of wind damage on forests: climate, soils, topography, and stand conditions (Wilson 2004). Hurricanes obviously represent an extreme climatic event. Trees growing in soil conditions that restrict root growth and depth are consistently more prone to uprooting. Variation in wind-throw along topographical gradients is more complicated and is often confused with damage due to species type and soil variation. There are many stand attributes that help determine tree susceptibility to wind-throw. These include height to diameter ratios, height, spacing, recent thinning, and impacts of previous disturbance on creating exposed edges that contain trees more vulnerable to wind-throw. Tree species composition may also impact the degree of damage from hurricanes. Therefore, stand composition and stocking levels represent stand attributes that can be manipulated by forest managers to reduce hurricane impacts.

Some evidence suggests that longleaf pine (*Pinus palustris*) might also be more tolerant to high winds than either slash pine (*P. elliottii*) or loblolly pine (*P. taeda*). In a study of the Hobcaw Forest, in coastal South Carolina after Hurricane Hugo, Gresham et al. (1991) reported that longleaf pine suffered less damage than loblolly pine. It was noted that species native to the Coastal Plain may be adapted better to the disturbance regimes found there. For example, longleaf pine, baldcypress (*Taxodium distichum*), and live oak (*Quercus virginiana*) suffered less damage than forest species with broad distribution ranges Gresham et al. (1991).

Johnsen et al. (2009) found that following hurricane Katrina, longleaf pine suffered less mortality (7%) than loblolly pine (26%). In addition to being potentially more resistant to wind-throw, longleaf pine is also more drought and fire resistant than the commonly planted loblolly pine (Landers et al. 1995). Wind damage increases with tree size, but the frequency and severity varies with species, site, wind parameters, and stand characteristics, specifically canopy evenness and age distribution, making it difficult to distinguish those tree species that appear to be more or less susceptible to wind damage (Gresham et al. 1991). The southeastern USA Coastal Plain is highly prone to hurricane events (Stanturf et al. 2007), and intense hurricanes occur two out of every three years across the region (McNulty 2002). Similar to historical natural fire regimes, the selection pressure of frequent high velocity winds has been a driving factor in forest composition.

Insects

Many types of insects damage southeastern forests, but the southern pine beetle (*Dendroctonus frontalis Zimm*.) is the most commercially destructive. Southern pine beetles caused more than $900 million in damage to SE pine forests between 1960 and 1990. Higher winter air temperatures are expected to increase over-wintering beetle larva survival rate, and higher annual air temperatures are expected to allow the beetles to produce more generations per year (Ayres and Lombardero 2000). Both of these factors could increase beetle populations. Other climate changes may work to reduce beetle populations. On the one hand, field research has demonstrated that moderate drought stress can increase pine resin production and, therefore, reduce the colonization success rate of the beetle (McNulty et al. 1998). However, severe drought stress reduces resin production and greatly increases the susceptibility of trees to beetle infestation (McNulty and Boggs 2010).

In addition to length and timing of the breeding season, other factors will likely impact the amount of insect caused damage under future climate conditions including the minimum winter air temperature and the prompt removal and destruction of infected timber (Rodriguez 1966). However, another factor closely linked to climate change may also impact insect success. Although it is one of the principle drivers of rising global air temperatures, CO_2 also increases forest productivity. Gan (2004) used an ecosystem model in conjunction with climate scenarios to predict that climate change would increase forest production by more than 7% during this century. The increase in productivity was a function of increased air temperature, longer growing season, and elevated atmospheric CO_2. However, southern pine beetle damage is also projected to increase by 4 to 7 times current levels, which would cause damage estimated at $500 to $800 million year per year (Gan 2004).

Potentially, some of the challenging impacts of climate change will be those conditions for which we have not considered or prepared for, such as previously unobserved combinations of environmental conditions that interact in new and unique ways. This concern is not unique to science. One such event occurred in the high elevations red spruce (*Picea rubens*, Sarg.) forests of western North Carolina. From 1999 until 2002, the area around Mt. Mitchell was in a period of extended heat wave and drought (McNulty and Boggs 2010). This southerly section of the Appalachian Mountains received some of the highest rates of acidic deposition in the eastern USA and contain remnant species present from the last glaciation, such as red spruce or hemlock that may be most at risk of extirpation. In 2001 some of the red spruce stands in the area began to die in large numbers while other stands of red spruce survived within the area (Figure 8.2). An examination of the sites found that stands with predominantly live trees and sites with predominantly dead trees had very different site characteristics. The sites with predominantly dead trees had much faster growth rates and higher soil nitrogen concentrations prior to the drought, compared to the historically slower growth rates and lower soil nitrogen concentrations from sites that largely survived the drought. In addition to the drought, there were signs that all the sites were attacked by southern pine beetles, a species that does not normally inhabit high elevation areas (Williams and Liebhold 2002). All the trees were attacked, but the trees that survived successfully repelled the phloem eating beetles. These were the trees from the poorer quality (i.e., lower soil nitrogen content) sites. Conversely, the trees that were unable to repel the beetles came from the higher quality (i.e., high soil nitrogen content) sites. The authors suggested that those factors that allowed stands to have the most vigorous growth under average climatic conditions also made these stands the most susceptible to mortality once those conditions changed. In combination, insects, drought, and nitrogen deposition ultimately combined to cause the observed forest mortality. If any one of these factors were not present, the trees may not have died. While in retrospect, the mechanisms for decline seem clear, forest managers have historically not been taught to consider vigorous forest stands as unhealthy. However, under a changing climate the definition of forest health, resilience, and resistance may need to be reevaluated (Thompson et al. 2009).

Elevated Atmospheric CO_2

Although CO_2 is not considered a disturbance factor for forests, atmospheric CO_2 could impact forest structure and function. Atmospheric carbon dioxide (CO_2) levels have increased nearly 35% since preindustrial times, from about 280 ppm to more than 380 ppm (IPCC 2007). Depending on the growth and emissions scenario used, atmospheric CO_2 may rise as high as 850 ppm by 2100 (IPCC 2007).

Figure 8.2 Red spruce (*Picea rubens Sarg.*) mortality in western North Carolina due to a combination of drought, southern pine beetles, and acid rain.

While carbon dioxide is the primary driver of anthropogenic climate change, it is also the basis of plant photosynthesis. Given that plant photosynthesis is not saturated at current CO_2 levels, anthropogenic increases in CO_2 will almost certainly lead to higher rates of photosynthesis if sufficient soil nutrients are available to support the elevated CO_2 induced growth (Oren et al. 2001). However, greater photosynthesis may not translate to significantly greater forest productivity and plant carbon storage, and gains in productivity may not be sustainable over the long term (Norby et al. 2010).

8.5 Ecosystem Services

Southeastern forests have been a major source of ecosystem goods and services for thousands of years (Anderson and Sassaman 1996). Current changes in demographics and climate may change the value of and need for some ecosystem services, but an overall reliance on southeastern ecosystems for societal and economic purposes will remain. In addition to goods and services such as timber and protection of water supplies, southeastern forests are considered important sinks for atmospheric CO_2, and part of a strategy to slow global warming. These services are outlined in the next section.

Forest Productivity and Carbon Sequestration

Large areas in the SE are actively managed for wood production at varying levels of intensity. For example, site preparation; weed control; fertilization; stocking, such as planting density and thinning; and genetic improvement can all impact forest productivity. Attention is being focused on the role forests play in sequestering some of the anthropogenic carbon inputs to the atmosphere in biomass and soils, while conserving existing carbon stocks through informed resource management (Blate et al. 2009).

The role of southeastern forests in providing a steady supply of timber and fiber is of particular importance in meeting current and future timber and fiber needs across the USA because forest harvests have substantially decreased across the other regions. As a whole, the South's forest sector produces approximately 60% of the total wood production in the USA (Prestemon and Abt 2002).

Climate change has the potential to impact forest productivity and carbon sequestration. Increases in forest carbon sequestration (a result of forests storing carbon in soils and woody tissues) can slow down the rate of atmospheric CO_2 increase and therefore help to slow down global warming. Southeastern forests also have been estimated to account for 36% of the carbon sequestered in the conterminous United States (Turner et al. 1995). Han et al. (2007) estimated that each year forests in the SE sequester 13% of regional greenhouse emissions in soils and long-lived forest products, such as lumber. Southeastern forests also contain about 30% of the nation's carbon stock (Mickler et al. 2004) and play a prominent role in the regional and global carbon cycle (Turner et al. 1995).

Forest Water Resources

When compared with other land uses, managed and unmanaged forests provide the cleanest and most stable water supplies for drinking water, recreation, power generation, aquatic habitat, and groundwater recharge. Large acres of forestland in the Appalachians and Piedmont are the headwaters of many river systems in the SE (Sun et al. 2011). These watersheds provide a disproportionately higher amount of the regional water supply than the Coastal Plain because these forests occupy areas with relatively high precipitation and low evapotranspiration (Brown et al. 2008).

The impacts of climate change on forest structure and functions are likely to result in negative consequences on water quantity and quality of forested watersheds through altering key hydrologic fluxes including precipitation and evapotranspiration, and the biogeochemical processes (Sun et al. 2011). An increase in air temperature means an increase in energy availability and atmospheric water demand. Thus for the humid southeastern USA, water shortages are expected to increase. For example, Walter et al. (2004) concluded ecosystem ET has been increasing at a rate of 10.4 mm per decade across six major basins that cover a majority of the watersheds in the USA. As more water is evapotranspired form the soil, less water will flow through the soil, and into streams and rivers. There will also be less water recharging shallow aquifers as tree water use (i.e., ET) increases with increasing air temperature (see Chapter 10. for more details on how forests use and yield water).

Shifts in tree species due to changes in climate, fire regime, and invasive species are likely to increase ecosystem transpiration rates and alter the carbon and nutrient balances. An increase in frequency of high intensity storm events will increase rainfall erosivity thus the potential for increased soil erosion and sedimentation (Marion et al. In press). An example of this increased soil erosion potential was forecast for the

Uwharrie National Forest where severe soil erosion was predicted to increase significantly under future climate changes (Figure 8.3).

Ecosystem model simulations and multiple watershed vegetation manipulation experiments suggest that activities that do not result in a forest type conversion or a coppice stand structure will not substantially alter streamflow responses to extreme precipitation events (Ford et al. 2011). However, based on forest conversion experiment studies, the conversion of deciduous forests (either naturally or by forest management) to pine monocultures in the Appalachians substantially altered the streamflow response to extreme annual precipitation. The pine increased soil permeability and rain fall absorption, but the pines also use more soil water than do the hardwoods. Thus, forest management may reduce flood risk but also exacerbate drought. Tradeoff between managing forests for opposite extremes should be carefully considered by water resource managers for contingency land use planning (Ford et al. 2011).

Increased frequency of heavy rainfall events will likely impact forest communities and increase flood occurrence. If there is an increase spring and summer droughts, it likely will make forest vegetation vulnerable to stresses due to high ET demands in the Coastal Plain region. Forests can also modulate regional climate by controlling energy and water transfers between the atmosphere and forested land-surface (Liu 2011). Forest restoration, afforestation, or both are expected to play important roles in mitigating the impacts of climate change on water resources in these regions.

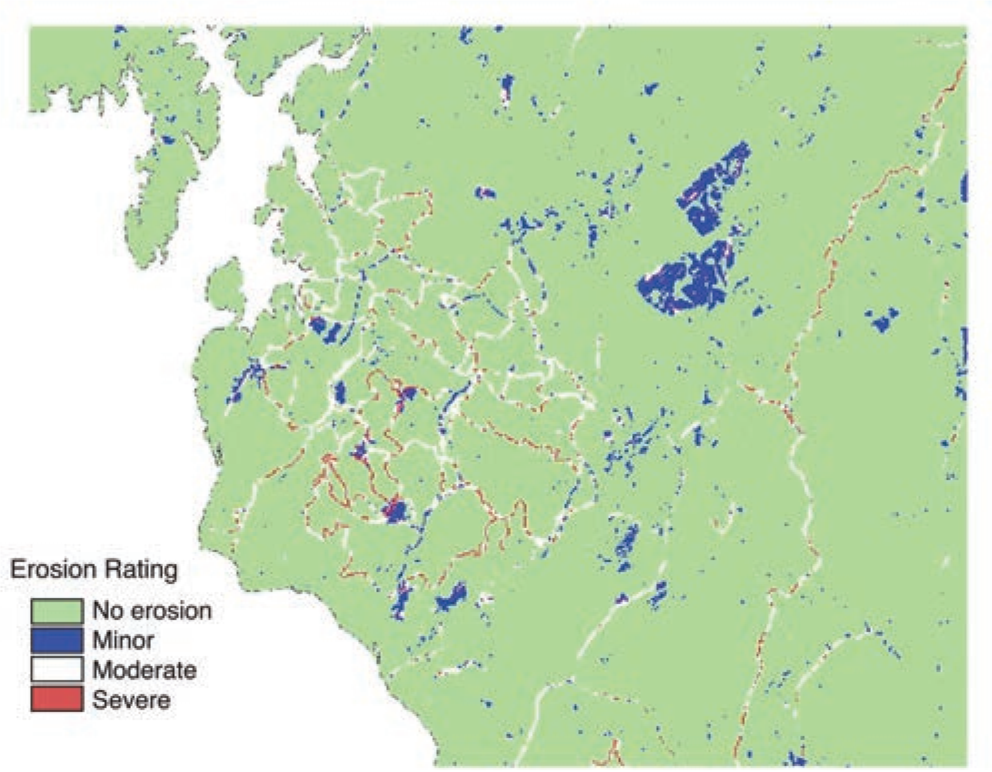

Figure 8.3 Revised Universal Soil Loss Equation predictions of soil erosion areas within the Uwharrie National Forest by 2030.

Regional modeling with a monthly scale water supply and demand model called the Water Supply Stress Index (WaSSI) suggests ecosystem water stress across the eastern USA will likely increase in the next 50 years, especially during the summer and fall seasons, due to increase water demand and reduced water yield (Caldwell et al. 2011).

8.6 Adaptation and Mitigation Options

In general, the biological productivity of SE forests likely will be enhanced by atmospheric carbon enrichment, as long as precipitation does not decline or air temperature does not increase soil moisture stress to a level that would offset potential CO_2 benefits on productivity. Use of forest resources is also anticipated to adapt to changes in productivity (de Steiger and McNulty 1998). For instance, a northward shift in forest productivity (Figure 8.4) is projected to lead to relative increases in the proportion of regional timber harvests that come from the northern reaches of the region. This may compensate for harvest reduction in the southeastern parts, which are projected to be more negatively affected by the biophysical effects of climate change. In addition, landowners are projected to switch land between forestry and agricultural in places and at times where the change in relative productivity warrants it.

There are a variety of other adaptation strategies to address climate trends and extremes. Potential adaptation strategies include genetic and silvicultural system improvements that increase water use efficiency or water availability. Increasing

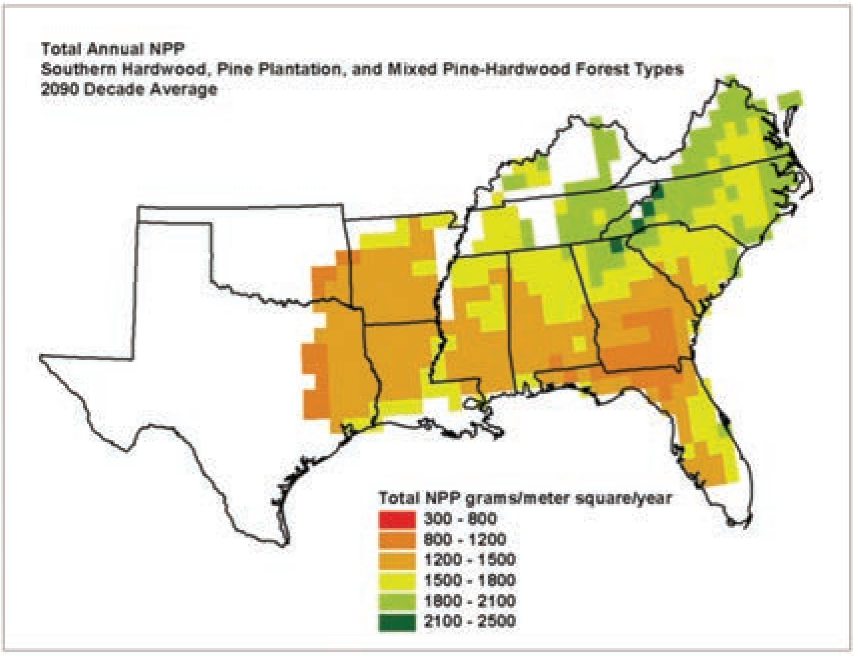

Figure 8.4 Forest model predictions of increased carbon sequestration (measured at net primary productivity, NPP) in the northern sections of the southern USA due to increasing air temperature by the end of this century.

knowledge of the role of fire, hurricanes, droughts, and other natural disturbances will be important in developing forest management regimes and increasing stand productivity in ways that are sustainable over the long term. Under a hotter, drier climate, an aggressive fire management strategy may prove important in this region (Dale et al. 2001). Timber productivity associated with increased temperature, growing season length, and CO_2 enrichment may be further enhanced by improved genetics, bioengineering, use of marginal agricultural land for tree production, and more intensive forest management (Schmidtling et al. 2004, Oren et al. 2004). Reduction of air pollutants, such as ozone and nitrogen oxides, may also be an important strategy for increasing forest productivity due to the potential for synergistic stress impacts (McNulty and Boggs 2010, Figure 8.5).

Increased use of fertilizers may increase forest productivity and carbon sequestration in an effort to partially mitigate greenhouse gas emissions. More than 400,000 ha of pine plantations are now fertilized each year with nitrogen which increases forest productivity (Albaugh et al. 2007). Fertilization can also decrease carbon losses by reducing soil respiration, and thus increasing forest carbon sequestration (Butnor et al. 2003). Other management tools that directly impact carbon sequestration include species selection, modification of initial planting density, and rotation length and thinning.

The effects of silvicultural treatments, such as planting density, thinning and rotation length, on carbon sequestration were analyzed by simulating carbon flux under

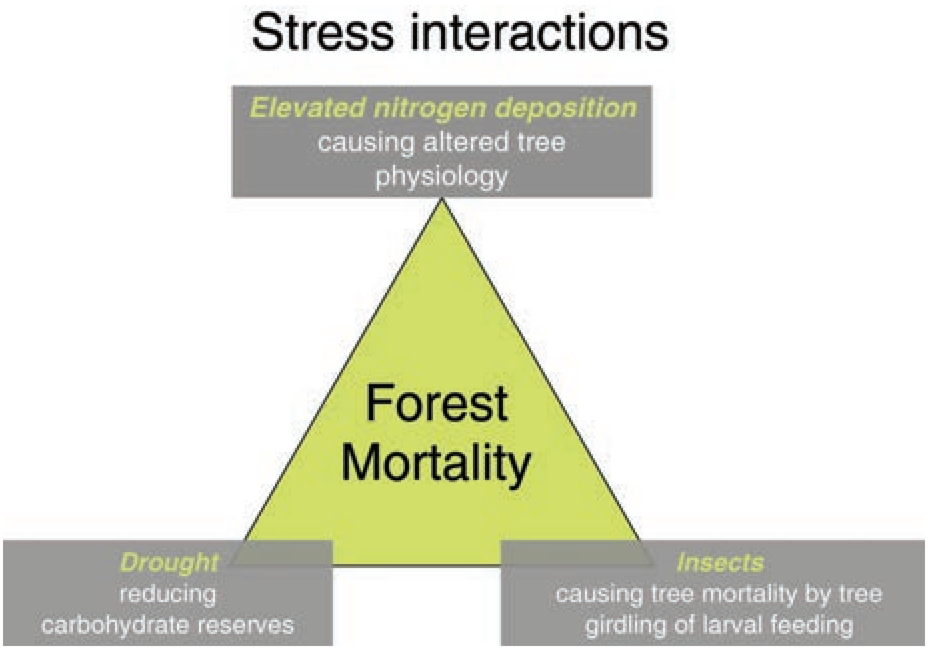

Figure 8.5 Interactions of climate (e.g., drought), biological (e.g., insects) and abiotic (e.g., fire or acid rain) can combine to cause forest mortality. The interactive stresses may be related (e.g., drought and fire) or unrelated (e.g., drought and acid rain). Any single stress may not have caused the mortality, but as climate change continues the potential for more frequent, more severe, and synergistic stress increases.

different climate and management scenarios for loblolly pine and slash pine plantations established in the southeastern USA Lower Coastal Plain (Gonzalez-Benecke et al. 2010 and 2011). Increasing the rotation length increased carbon stock in both species. Canadell and Raupach (2008) cited longer harvesting cycles as a major management strategy for increasing forest carbon stocks.

Improved understanding of climate change impacts and adaptation options are only useful if this information can be conveyed to the land manager. New web-based models and tools are being developed to allow for easier, more site specific climate change assessments. For example, the web-based Distrib/Shift forest species distribution model gives users the ability to examine which tree and bird species will likely become more and less dominant in that area over the coming years and decades (Iverson et al. 2011). Similarly, the web-based WaSSI (Water Supply Stress Index) hydrologic model gives land managers the ability to examine the impacts of climate, population and land use change on water supply and demand on their watersheds. Finally, Web-based tools like TACCIMO (Template for Assessing Climate Change Impacts and Management Options) allow the user to search scientifically reviewed literature on climate change impacts for their area, and then to further use TACCIMO to search for management options to address or adapt to these changes. Significantly improved graphic user interfaces (Figure 8.6), data storage, and internet access speeds have greatly improved the application of these tools.

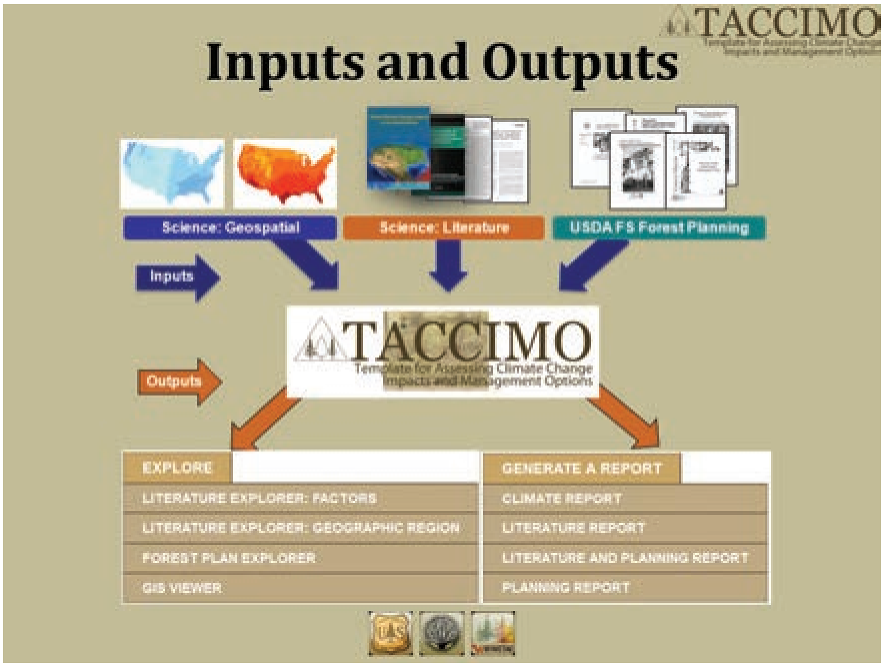

Figure 8.6 Web-based tools such as TACCIMO (Template for Assessing Climate Change Impacts and Management Options) are increasingly being used to easily translate scientific knowledge into the hands of the land manager.

8.7 Conclusions

Southeastern forests are as diverse as the cultures that exist within them. The wide range of tree, plant, and animal species make the region both resistant and susceptible to change. Some species will not be able to adapt to rapidly changing climatic conditions; other species will fill vacated niches that develop. Protecting the overall integrity of the ecosystem will be less of a challenge than protecting all of the parts. Several independent studies suggest that remnant species present from the last glaciation, such as red spruce or hemlock may be most at risk of extirpation. If changes result in warmer, drier conditions in some parts of the SE, conditions could favor more drought-tolerant species such as oaks and long-leaf pine.

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