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Assessing quality of clearcut habitats for amphibians: Effects on abundances versus vital rates in the southern toad (*Bufo terrestris*)

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ABSTRACT

Forest clearcutting is a form of habitat alteration that drastically alters the landscape and may contribute to declines in amphibian populations. Indeed, many studies have documented decreases in amphibian abundances and species richness in clearcuts. The development of effective conservation strategies to reduce the effects of timber harvesting has been hindered by lack of knowledge of the mechanisms underlying these changes in abundance. To better understand the potentially negative consequences of forest clearcutting, we used field enclosures in forested and clearcut habitats to examine changes in the survival and growth of juvenile southern toads (*Bufo terrestris*) over a two-month period. We also conducted a comparative monitoring study using drift fences and pitfall traps in forests and clearcuts to determine the effect of clearcutting on the abundance of juvenile southern toads. We found no significant effect of habitat on the number of juvenile southern toads captured in forests or clearcuts. In contrast, the average survival of toads in clearcut enclosures was significantly reduced compared to that of toads in forested enclosures (17 ± 5% versus 61 ± 3%). Toads surviving in clearcuts were also significantly smaller than those surviving in forested enclosures (27.9 ± 0.1 mm versus 30.3 ± 0.8 mm SVL). Our results highlight the difficulty in interpreting abundance patterns as a sole metric for habitat comparison. Because there is much interest in studying the effects of habitat alteration on amphibian populations, we recommend that future studies place more emphasis on determining changes in vital rates of populations following habitat alteration.

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1. Introduction

Habitat alteration is a major factor in the global decline of amphibians (Collins and Storfer, 2003; Stuart et al., 2004). Timber harvesting is one form of habitat alteration that may decrease the suitability of the terrestrial environment for amphibians. Clearcutting and other intensive forest management practices create a mosaic of fragmented habitats, with potentially negative consequences for amphibian populations. The increased air and soil temperatures and reduced ground litter in early-successional habitats (Russell et al., 2004) may reduce survival and migratory success of amphibians (deMaynadier and Hunter, 1999). Clearcuts may also become barriers to movement if amphibians avoid entering them in favor of forested habitats (Rothermel and Semlitsch, 2002; Chan-McLeod, 2003; Rothermel, 2004). Because up to 82% of amphibian species are forest-dependent (Stuart et al., 2004), forest management practices have the potential to affect a large proportion of amphibians and contribute to ongoing population declines.

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Recognition of population declines and concerns over the potentially negative effects of timber harvesting have generated much interest in the response of amphibians to forest alteration. deMaynadier and Hunter (1995) reviewed the literature regarding the effects of clearcutting on amphibians and found that, in general, both abundance and richness are reduced following harvest. However, these patterns are often dependent on forest type, elevation, and species assemblage, and several studies contradict this negative trend. A number of studies in the southeastern US, for example, have documented greater amphibian abundances in clearcuts compared to unharvested reference sites (Pais et al., 1988; Phelps and Lancia, 1995; Clawson et al., 1997; Ryan et al., 2002). While at least one author has warned that species abundances are not a direct measure of habitat quality (Van Horne, 1983), many studies continue to focus on changes in abundance as the sole metric for habitat comparisons.

Relying on abundances to compare the quality of habitats is problematic for several reasons. First, populations do not always respond immediately to habitat change, but often exhibit time lags (Brooks et al., 1999). Therefore, abundances of some species may not decrease initially despite the habitat being of poorer quality. Second, if a poor-quality habitat patch with high animal mortality is sustained by immigration from other patches, abundances will provide the misleading appearance that there is no effect of habitat alteration. Third, determining the effects of habitat alteration on abundances does not indicate which processes are responsible for observed changes. Determining whether altered habitats affect species by influencing migration or by causing changes in survival or reproduction is important in formulating subsequent conservation strategies. Lastly, comparative abundance surveys, especially for amphibians and reptiles, often rely on the number of animal captures as a proxy for animal abundance in different habitats. Captures from any sampling method (e.g., pitfall traps) are a product of both population abundances and detection probabilities, which are partly a function of the behavior of the animals and their activity levels. If behavior or movement rates vary among habitats with differing levels of alteration, then resulting abundance estimates are likely to be biased (Bailey et al., 2004b). For these reasons, determining changes in vital rates (birth, immigration, death, or emigration) following habitat alteration provides the only direct measure of habitat quality (Van Horne, 1983; Armstrong, 2005).

We studied the effects of forest clearcutting on the southern toad (*Bufo terrestris*) using two approaches simultaneously. First, we conducted a comparative abundance survey using drift fences with pitfall traps to compare abundances of southern toads in recent clearcuts with abundances in adjacent unharvested pine (*Pinus* spp.) forests. Second, we performed an experimental study using field enclosures to determine the effects of clearcutting on the survival and growth of juvenile southern toads. The juxtaposition of these two approaches allowed us to evaluate the quality of forest clearcuts for a common amphibian species by comparing both an indirect and direct measure of habitat quality. Consequently, our results illuminate a larger problem in interpreting the effects of forest management on amphibians and demonstrate why more research should focus on changes in vital rates of amphibian populations following habitat alteration.

2. Materials and methods

2.1. Study species

Southern toads (*B. terrestris*) are habitat generalists that are often encountered in highly fragmented landscapes, including suburban areas and golf courses (Scott et al., 2003). As anurans, they generally are able to tolerate higher temperatures and desiccation risks than many amphibian species, especially in comparison to salamanders (Stebbins and Cohen, 1995; Zug, 2001). They are also capable of storing and reabsorbing large quantities of water in their bladders (Thorson and Svihla, 1943; Hillyard, 1999). These factors may predict a tolerance to warmer temperatures found in altered landscapes. For these reasons, the response of southern toads in our studies can be viewed as a conservative metric for examining the effects of forest clearcutting on amphibians.

2.2. Abundance survey

We selected four forested sites on the US Department of Energy's Savannah River Site in Barnwell County, South Carolina, as part of the LEAP (Land-use Effects on Amphibian Populations) study, a multi-regional, collaborative investigation of the effects of land-use practices on migratory success and demographics of pond-breeding amphibians. These sites are second-growth forests comprised predominantly of loblolly pine (*Pinus taeda*) in the Upper Coastal Plain of the southeastern US. Each study site was a circular area 350 m in diameter centered on an isolated, seasonal wetland. Each wetland was located at least 200 m from paved roads, powerline right-of-ways, and other open areas. We divided each study site into four 4-ha quadrants delineated by two perpendicular transects that intersected at the center of the wetland (Fig. 1). Each quadrant was randomly assigned one of four treatments: (1) an unharvested control (>30 years old); (2) a partially harvested stand, in which the canopy was thinned to approximately 85% of that in the control; (3) a clearcut with coarse woody debris retained (CC-retained); and (4) a clearcut with coarse woody debris removed (CC-removed). The last treatment represents the most extreme level of alteration and produces a habitat typical of even-aged forest management in the southeastern US. Logging was completed at the sites in March 2004.

In April 2004, we installed nine 15-m sections of drift fence in each quadrant at all four sites. We placed six 8-L pitfall traps (30 cm in diameter and 25 cm high) paired on opposite sides of each section of drift fence, yielding a total of 54 pitfall traps within each quadrant. Pitfall traps contained 1–3 cm of standing water and floating sponges in the bottom. We constructed the drift fences of aluminum flashing buried 15 cm into the ground and standing 45 cm tall. We distributed the drift fences evenly throughout each quadrant to maximize the likelihood of capturing animals in the treatments (Fig. 1). We checked the drift fences daily from 1 June to 28 July 2004 and recorded all amphibian captures, including juvenile southern toads. Animals were released on capture and were

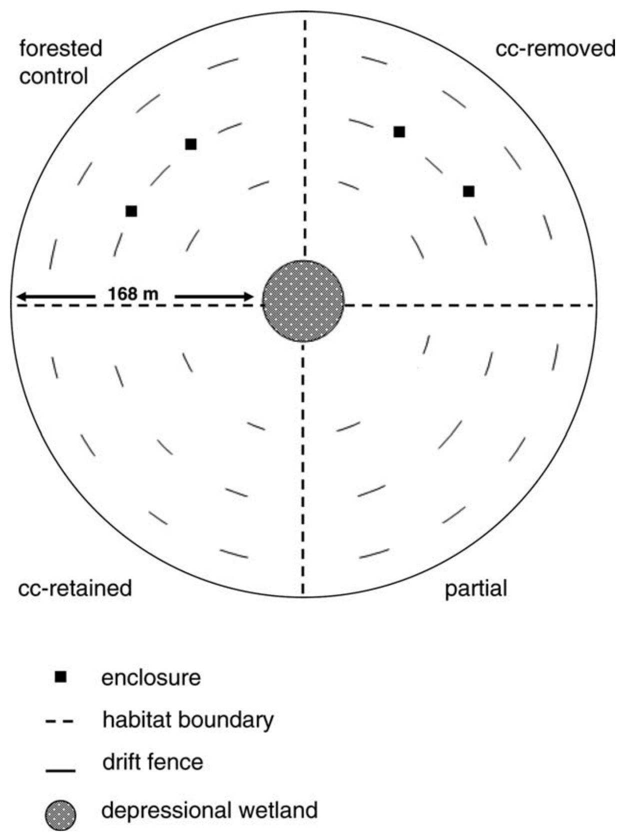

Fig. 1 – Diagram showing the arrangement of drift fences and enclosures at each site. See text for description of the four treatments. Note that drift fences, wetland, and enclosures are not pictured to scale.

not marked, yielding raw counts. This period of drift fence monitoring coincided with the period during which juvenile southern toads leave their natal ponds to establish terrestrial home ranges, where they abide until they become reproductive adults. For the purposes of this study, we only analyzed captures from the control and CC-removed treatments to permit a direct comparison of abundance data to the enclosure study conducted in these two habitats.

2.3. *Field enclosure study*

We measured growth and survival of juvenile southern toads in terrestrial enclosures in the control and CC-removed treatments. Although enclosures confine the animals, they eliminate the possibility that other processes, such as immigration and emigration, affect perceived abundance within the experimental populations and ensure that the growth and survival of the toads is a reflection of time spent in a single habitat. We constructed two enclosures in each treatment (control and CC-removed) at two of the four sites for a total of four control enclosures and four CC-removed enclosures. We constructed enclosures of aluminum flashing buried 25 cm deep, standing 65 cm tall and measuring 4 m × 4 m. Similarly constructed enclosures of larger sizes have been used in studies of terrestrial density-dependence in ambystomatid salamanders (Pechmann, 1995). We systematically placed the enclosures within the interior of the treatments at least 50 m from the edges of the quadrant (Fig. 1). We minimized disturbance to the soil and ground cover within the enclosures to maintain an environment suitably representative of the overall treatment.

We collected 240 recently metamorphosed southern toads as they emigrated from a wetland located near our study sites. We maintained all toads indoors in ventilated containers at 23 °C on paper towels wet with aged well-water for less than one week prior to release into experimental enclosures. We randomly assigned groups of 30 toads to each of the eight enclosures. Our experimental density of 1.8 toads per m^2 is lower than natural densities encountered along pond margins during the post-metamorphic period (Beck and Congdon, 1999). We individually marked each animal by toe-clipping and recorded snout-vent length (SVL) and fully hydrated mass prior to release into the enclosures on 10 July 2004. After one month, we censused the animals for three consecutive days, recording the SVL and mass of each animal in the field upon capture with an Ohaus® Scout Pro battery-powered balance. We released all animals back into the enclosures immediately following data collection. We repeated this process again two months after the initial release.

To census the toads, we hand-captured them in the enclosures between 06:00 and 08:00 each morning during the three-day sampling periods. Consecutive days of censusing during a sampling period combined with individual marking of toads enabled the capture histories to be analyzed in a robust-design mark-recapture format using program MARK (Pollock, 1982; White and Burnham, 1999). No animals were captured on the third day of censusing that had not been previously captured in one of the two earlier days, resulting in high probabilities of successfully capturing surviving toads during the census periods. Subsequent population estimates derived from program MARK for each census period differed only slightly (i.e., by one or two animals) from the minimum number known alive during the census period (MNKA; Krebs, 1966). Because program MARK does not currently allow fit-testing for robust design recapture models (Bailey et al., 2004a), we opted to use the typically more conservative MNKA at each interval for the comparison of survival rates. Although not shown here, tests of our hypotheses based on model-derived population estimates resulted in the same conclusions.

2.4. *Statistical analyses*

To test whether clearcutting affected the number of juvenile southern toads captured at drift fences, we performed an analysis of variance (ANOVA) on the total captures at all nine drift fences within a treatment, using site as a blocking factor. To test the effect of treatment on survival of penned toads over two months, we performed a multivariate repeated measures analysis of variance (MANOVA; Von Ende, 2001) using the MNKA at each of the three intervals and we accounted for the nestedness of the enclosures within two sites in our analysis. To test the effect of treatment on body size over two months, we performed a repeated measures MANOVA using the mean SVL from each enclosure at each of the three intervals, again accounting for the nestedness of the enclosures within two sites.

We tested the hypothesis that larger animals had greater survival by pooling all toads in clearcut enclosures and forested enclosures separately and conducting logistic regressions to test for an effect of initial SVL on survival to the first month. We repeated this procedure to test for an effect of initial SVL on survival to the second month. We also used non-parametric bootstrap resampling (Lunneborg, 2000) to test whether animals that perished in the second month of the enclosure study, regardless of treatment, represented a non-random sample of all penned toads with respect to their growth rate in the first month. Most toads that were recaptured in the enclosures and weighed in the field appeared to lose body mass because initial release weights were recorded in the laboratory when toads were fully hydrated. Thus, change in body mass was a reflection of both growth and hydration state at the time of capture, whereas SVL was more likely a reflection of growth alone. Therefore, we performed two resampling analyses, one using change in SVL and one using change in body mass.

All statistical assumptions were examined prior to analyses and no transformations were needed. All statistical analyses were performed using SAS® version 9 (SAS Institute Inc., 2000) and significance was evaluated at the $\alpha = 0.05$ level.

3. Results

A total of 357 juvenile southern toads were captured in the four CC-removed quadrants and 307 toads were captured in the four forested control quadrants from 1 June to 28 July 2004. Juvenile toads were captured at drift fences in CC-removed clearcuts more frequently than in unharvested forests at three of the four sites (Fig. 2). However, neither treatment nor site had a significant effect on the number of captures (treatment: $F_{1,3} = 0.18$, $p = 0.70$; site: $F_{3,3} = 2.76$, $p = 0.21$).

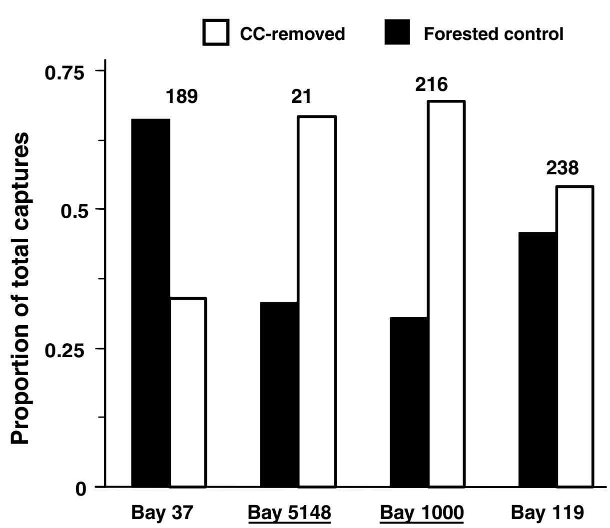

Fig. 2 – Proportion of juvenile southern toads captured in forested controls versus CC-removed habitats at each study site using drift fences and pitfall traps. Our field enclosures were located at Bay 5148 and Bay 1000. The total number of captures of juvenile southern toads at each site is given above the bars.

In contrast, we found significant effects of treatment and time on the number of toads surviving over two months in the experimental enclosures (Table 1; Fig. 3). There were no effects of site, time-by-site, or time-by-treatment interactions on survival (Table 1). The average survival of toads in clearcut enclosures after two months was 17 ± 5% whereas the average survival in forested enclosures was 61 ± 3%. Individual contrasts revealed that treatment significantly affected survival of toads in the second month (Table 2).

The mean SVL of juvenile toads in both forested and clearcut enclosures increased over two months as the animals grew (Table 3). However, there was a significant treatment effect as the mean SVL of toads in forested enclosures increased significantly more than that of toads in clearcut enclosures (Table 3; Fig. 4), a response that was consistent through time (Table 4). Toads that survived in clearcut pens averaged 27.9 ± 0.1 mm SVL whereas toads that survived in forested pens averaged 30.3 ± 0.8 mm SVL. The results of the logistic regressions suggest that initial body size was not an important predictor of survival to any month in either clearcuts or forests (Table 5).

Results of non-parametric bootstrap resampling suggest that growth rate, as measured by change in SVL in the first month, did not correlate to greater survival in the second

Table 1 – Results of the repeated-measures analysis of variance of the effects of habitat treatment, site, and time on the number of surviving southern toads in enclosures over two months

	df	MS	F	p
Between-subject				
Site	1	66.667	3.92	0.119
Treatment	2	124.333	7.31	0.044
Error	4	17		
	df	Wilks' λ	F	p
Within-subject				
Time	2,3	0.0025	80.87	0.003
Time × site	2,3	0.5008	1.49	0.354
Time × treatment	4,6	0.0837	3.68	0.076

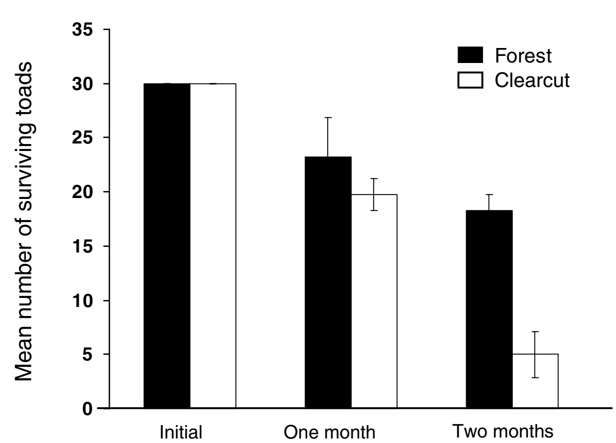

Fig. 3 – Mean number of surviving toads (±1 SE) in each treatment (n = 4 enclosures per treatment).

Table 2 – Results of individual contrasts from a repeated-measures analysis of variance testing the effects of treatment and site on the number of surviving toads in enclosures at each interval

Source	df	MS	*F*	*p*
First month interval				
Mean	1	578.0	28.54	0.006
Site	1	72.0	3.56	0.132
Treatment	2	24.5	1.21	0.388
Error	4	20.3		
Second month interval				
Mean	1	882.0	51.88	0.002
Site	1	8.0	0.47	0.530
Treatment	2	121.0	7.12	0.048
Error	4	17.0		

Table 4 – Results of individual contrasts from a repeated-measures analysis of variance testing the effects of treatment and site on mean snout-vent length of toads in enclosures

Source	df	MS	*F*	*p*
First month interval				
Mean	1	22.883	28.54	0.002
Site	1	0.008	3.56	0.898
Treatment	2	1.349	1.21	0.146
Error	4	0.417		
Second month interval				
Mean	1	4.720	5.08	0.087
Site	1	0.544	0.59	0.487
Treatment	2	2.037	2.19	0.228
Error	4	0.929		

Table 3 – Results of the repeated-measures analysis of variance of the effects of habitat treatment, site, and time on the mean snout-vent length of southern toads in enclosures over two months

	df	MS	*F*	*p*
Between-subject				
Site	1	0.118	0.25	0.642
Treatment	2	6.125	13.09	0.018
Error	4	0.467		
	df	Wilks' λ	*F*	*p*
Within-subject				
Time	2,3	0.0631	22.25	0.016
Time × site	2,3	0.7975	0.38	0.712
Time × treatment	4,6	1.667	2.97	0.439

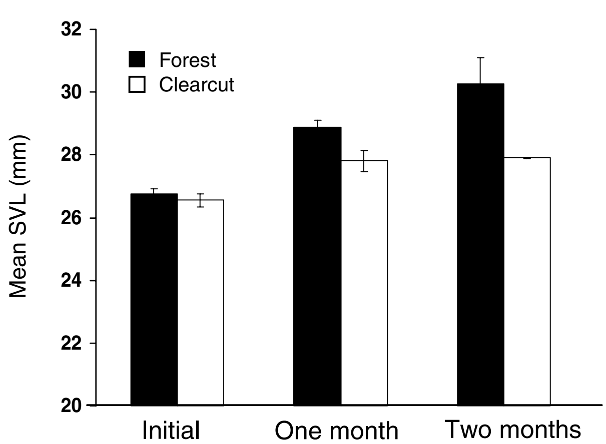

Fig. 4 – Mean snout-vent length of surviving toads (±1 SE) in each treatment (*n* = 4 enclosures per treatment).

month (Table 6). However, with respect to body mass, there was a strong indication that the toads that perished in the second month were a non-random subset of all penned toads. Toads that perished in the second month lost more mass in the first month, on average, than did toads randomly drawn from the total pool of animals that survived the first month (Table 6).

Table 5 – Results of the logistic regressions testing for effect of initial body size on survival among toads in enclosures

	χ^2	*p*
Clearcut pens		
Survival to first month	0.03	0.86
Survival to second month	0.01	0.94
Forested pens		
Survival to first month	1.15	0.29
Survival to second month	0.02	0.76

Table 6 – Mean change in snout-vent length and body mass during the first month of toads that perished in the second month (n = 84)

	Mean change	90% CI
SVL (cm)	1.4	1.21 to 1.85
Mass (g)	−0.189	−0.185 to −0.068

Confidence intervals were derived from 1000 bootstrap resampled subsets (of size $n = 84$) drawn from the entire pool of toads that survived the first month.

4. Discussion

4.1. *Effects of clearcutting on toad abundances*

Clearcutting typically has a negative effect on amphibian abundances and richness (Petranka et al., 1994; Ash, 1997; Grialou et al., 2000; Knapp et al., 2003; Karraker and Welsh, 2006). Indeed, several studies have suggested that forest cover is a critical factor that determines the distribution and density of many species (Porej et al., 2004; Herrmann et al., 2005). However, the response of amphibians to clearcutting varies considerably among species and physiographic regions (deMaynadier and Hunter, 1995; Russell et al., 2004). The results of our study agree with others that show little effect of clearcutting on anuran abundances, and in some cases, increases in anuran abundances (e.g., Pais et al., 1988; Phelps and Lancia, 1995;

Clawson et al., 1997; Ryan et al., 2002). A critical and untested assumption in some of these studies, and one that is true of the abundance survey in this study, is that capture probabilities and detection do not vary with treatment. Ideally, the application of mark-recapture techniques to analyze captures of individually marked animals can reduce possible bias resulting from such assumptions and can clarify the inferences made about the effects of habitat type on amphibian populations based on abundance measures.

4.2. *Effects of clearcutting on survival and body size*

The reduced survival and body size of southern toads in clearcuts indicates that clearcuts are poor-quality habitats for these amphibians. Juvenile toads experienced higher mortality in clearcuts, and those that did survive were smaller in size than their forest-dwelling cohorts after two months. Increased juvenile mortality can reduce population sizes by eliminating future reproductive animals (Vonesh and De la Cruz, 2002). Additionally, smaller body size in juvenile amphibians results in a delayed onset of maturity (e.g., Smith, 1987; Semlitsch et al., 1988; Berven, 1990; Scott, 1990, 1994), which reduces the number of reproductive animals available in the breeding adult population. The results of the logistic regressions on body size suggest that changes in the mean size of toads in enclosures were products of growth and not a reflection of differential survival based on body size.

Because southern toads have a greater tolerance to the conditions found in clearcuts than do many other amphibians (Thorson and Svihla, 1943), other amphibians are likely to suffer even greater physiological responses to clearcutting. However, the results from studies of other species have been mixed. Chazal and Niewiarowski (1998) found no effects of clearcutting on body mass, clutch size, lipid storage, or the number of recaptured mole salamanders (*Ambystoma talpoideum*) maintained in 100 m^2 field enclosures when compared to salamanders from forested enclosures. In contrast, Rothermel and Luhring (2005), using very small (0.33 m^2) enclosures, found that mortality of *A. talpoideum* could occur quite rapidly in recent clearcuts, particularly if salamanders did not have access to burrows. In the 16 m^2 enclosures used in our study, we found that mortality of juvenile southern toads in clearcuts increased significantly after the first month. Apart from species differences, there are at least two other explanations for the variation in results among these enclosure-based studies. First, as field enclosures increase in size, relocating highly fossorial amphibians can become difficult, and relying on pitfall traps to recapture animals for survival comparisons (as in Chazal and Niewiarowski, 1998) may begin to approximate comparisons of capture data from drift fence studies. Second, and more plausibly, larger field enclosures may incorporate more habitat complexity, allowing amphibians to find suitable refugia in otherwise hostile environments. Thus, while altered habitats are generally of poorer quality due to desiccating conditions and other factors, the ability to find and use suitable microhabitats in a larger landscape may mitigate some of the negative impacts associated with forest removal. Studies that specifically examine habitat selection and use by amphibians can greatly improve our understanding of amphibian responses to forest alteration.

Clearcuts used in forest management at the Savannah River Site typically range from 2 to 30 ha (Krementz and Christie, 2000). Due to the small size of clearcuts used in our study (<4 ha) and the ability of adult southern toads to move long distances overnight (up to 300 m; Graeter, 2005), clearcuts in our study may have been easily traversed by juvenile southern toads. When juvenile toads spend short amounts of time in clearcuts, their probability of surviving is likely comparable to that of toads in forests. In contrast, sizeable clearcuts that require lengthy passages (>30 days based on the current study) to escape could result in increased animal mortality due to the greater amount of time spent in poor-quality habitat. For amphibians that are less vagile or have high site fidelity (e.g., *Ensatina exchscholtzii* and *Plethodon elongatus*; Karraker and Welsh, 2006), clearcuts may represent significant barriers that trap populations and contribute to local declines.

Our field enclosure study provides critical insight into the processes that can reduce amphibian abundance following habitat alteration not revealed by drift fence or monitoring studies. Canopy removal during forest clearcutting causes an increase in daytime temperatures that can accelerate desiccation or exceed lethal limits, leading to rapid mortality (e.g., Rothermel and Luhring, 2005). Although many juvenile southern toads in our study lost body weight in the first month relative to their fully hydrated initial mass, we found that individuals that lost the most mass in the first month were significantly less likely to survive to the second month. Therefore, dehydration probably influenced toad mortality in clearcuts. Other possible reasons for reduced survival in clearcuts include an increase in predation, inadequate prey populations, or a reduction in time spent foraging as animals acted to minimize water loss in recent clearcuts. However, enclosures probably excluded many non-avian predators (e.g., colubrid snakes), possibly reducing predation on toads. Additional manipulative studies are needed to identify the specific causes of decreased amphibian survival following clearcutting.

5. Conservation implications

Although estimates of vital rates provide the only direct measures of the effects of habitat alteration on amphibians and other wildlife (Armstrong, 2005), many studies continue to focus on changes in abundance and richness as indicators of habitat quality. The results of our study suggest that differences in abundance should not be used as the sole metric of habitat quality and that a more thorough experimental approach incorporating estimation of vital rates may be required to understand the implications of habitat alteration.

Vital rates are directly affected by habitat change, often without the inherent time lags that occur with population sizes (Brooks et al., 1999). Therefore, they may be particularly useful in the early identification of problems arising from habitat alteration. Also, examining vital rates can identify which demographic processes are responsible for changes in local populations (i.e., survival, reproduction, or migration), providing planners with explicit targets for conservation management. In our study, field enclosures proved to be an effective tool for studying juvenile survival and growth in isolation from other demographic processes.

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Habitat alteration increases invasive fire ant abundance to the detriment of amphibians and reptiles

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ORIGINAL PAPER

Habitat alteration increases invasive fire ant abundance to the detriment of amphibians and reptiles

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Abstract Altered habitats have been suggested to facilitate red imported fire ant (*Solenopsis invicta*) colonization and dispersal, possibly compounding effects of habitat alteration on native wildlife. In this study, we compared colonization intensity of wood cover boards by *S. invicta* among four forest management treatments in South Carolina, USA: an unharvested control (>30 years old); a partially thinned stand; a clearcut with coarse woody debris retained; and a clearcut with coarse woody debris removed. Additionally, we compared dehydration rates and survival of recently metamorphosed salamanders (marbled salamanders, *Ambystoma opacum*, and mole salamanders, *A. talpoideum*) among treatments. We found that the number of wood cover boards colonized by *S. invicta* differed significantly among treatments, being lowest in the unharvested forest treatments and increasing with the degree of habitat alteration. Salamanders that were maintained in experimental field enclosures to study water loss were unexpectedly subjected to high levels of *S. invicta* predation that differed among forest treatments. All known predation by *S. invicta* was restricted to salamanders in clearcuts. The amount of vegetative ground cover was inversely related to the likelihood of *S. invicta* predation of salamanders. Our results show that *S. invicta* abundance increases with habitat disturbance and that this increased abundance has negative consequences for amphibians that remain in altered habitats. Our findings also suggest that the presence of invasive *S. invicta* may compromise the utility of cover boards and other techniques commonly used in herpetological studies in the Southeast.

Keywords *Ambystoma* · Clearcutting · Cover boards · Forest management · Mole salamander · *Solenopsis invicta*

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Introduction

Red imported fire ants (*Solenopsis invicta)* have rapidly expanded across the southeastern United States following their accidental introduction into Alabama in the 1930s (Wojick et al. 2001). They are ravenous scavengers and predators that have quickly become a dominant invasive in many parts of the Southeast. Among the more important impacts that

S. invicta have in many ecosystems are their interactions with wildlife. They displace native ants through competition and can reduce total arthropod densities (Porter and Savignano 1990). Additionally, *S. invicta* are known to prey on young birds, small mammals and reptiles (Allen et al. 1994), although interactions with most wildlife remain understudied.

Solenopsis invicta have been suggested as contributing to the declines of the Eastern Kingsnake (*Lampropeltis getula*, Wojcik et al. 2001; Allen et al. 2004), Southern Hognose Snake (*Heterodon simus*, Tuberville et al. 2000), and Texas Horned Lizard (*Phrynosoma cornutum*, Goin 1992). Direct data incriminating *S. invicta* in reptile declines is lacking, but the aforementioned species have all experienced population declines concomitant with increases in the local distribution of *S. invicta*. Also, Slater and Allen (2002) demonstrated that herpetofaunal richness and abundance in several South Carolina communities responded positively to *S. invicta* eradication, suggesting that *S. invicta* can suppress amphibian and reptile populations across a landscape. Several other studies have reported direct predation of reptiles or reptile nests by *S. invicta*, also documenting reduced hatching success (Montgomery 1996; Allen et al. 1997; Reagan et al. 2000).

Solenopsis invicta is considered a "weedy" species because colonies multiply rapidly and quickly infiltrate disturbed and early-successional habitats (Tschinkel 1987, 1988). In fact, large-scale habitat disturbance has been hypothesized to promote their invasion (Zettler et al. 2004). One form of large-scale habitat disturbance that is ubiquitous in the Southeast is forest clearcutting. Approximately 810,000 ha of forest are clearcut annually in the southeastern United States (Siry 2002), providing a probable avenue for the ongoing spread of *S. invicta*. Additionally, because clearcutting has been shown to negatively affect amphibian and reptile populations (e.g., Russell et al. 2004; Todd and Rothermel 2006), the dual threats of invasive *S. invicta* and habitat alteration may compound negative impacts on reptiles and amphibians, possibly causing greater local population declines than either threat singly.

As part of an experimental study of amphibian and reptile responses to forest management in the Upper Coastal Plain of South Carolina, we used cover boards and small enclosures to examine changes in relative abundance and dehydration rates of amphibians and reptiles following forest harvesting. Cover boards offer alternative cover that approximates natural refugia used by amphibians and reptiles and are often used to survey animal populations (Grant et al. 1992; Heyer et al. 1994). Here, we compare colonization rates of artificial cover by *S. invicta* among four forest harvest treatments. We also test whether predation of amphibians by *S. invicta* varied among treatments. Specifically, we examined whether *S. invicta* colonized more cover boards in clearcuts than in forested habitats and whether predation of amphibians by *S. invicta* increased with increasing habitat alteration. Our observations have important implications regarding the effects of *S. invicta* colonization of disturbed habitats on reptiles and amphibians.

Methods

Study sites

In 2003, we selected four forested sites on the US Department of Energy's Savannah River Site (SRS) in Barnwell County, South Carolina. The SRS is comprised predominantly of second-growth managed loblolly pines (*Pinus taeda*) and mixed hardwoods in the Upper Coastal Plain of the southeastern United States (see also Todd and Rothermel 2006). We centered each of the four circular experimental sites on small, isolated, seasonal wetlands (Carolina bays, hereafter referred to as sites) that hold water during winter and early spring. The circular sites extended outward from the wetland boundaries for 168 m. Each wetland was located at least 200 m from paved roads, powerline rights-of-way, and other open areas. We divided each site into four 4-ha quadrants delineated by two perpendicular transects that intersected at the center of the wetland (Fig. 1). Each quadrant was randomly assigned one of four treatments, (1) an unharvested control (>30 years old); (2) a partially thinned stand, in which the canopy was thinned to approximately 85% of that in the control (thinned forest); (3) a clearcut with coarse woody debris retained (CC-retained); and (4) a clearcut with coarse woody debris removed (CC-removed), with the added constraint that the two forested plots were always opposite from each other (Fig. 1). The most altered habitat type, a clearcut with coarse woody

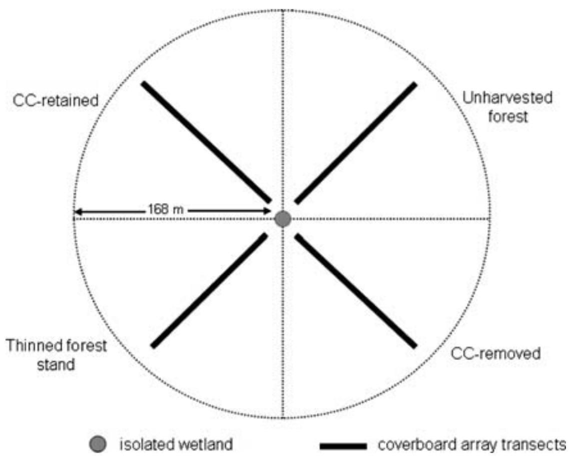

Fig. 1 Diagram of one of the four experimental sites showing the arrangement of the four habitat treatments and the orientation of cover board arrays which contained cover boards spaced 5, 20, 35, 50, 75, 100, 125, and 150 m from the edge of the centrally-located wetland

debris removed, was characteristic of forest harvesting practices in even-aged managed forests in the southeastern United States, but also represented other forms of forest conversion such as agriculture, residential development, and power line rights-of-way. The thinned canopy stands were representative both of canopy thinning practices and some sustainable harvesting methods (i.e., size-selective harvesting). Logging at the study sites commenced in February 2004 and was completed at all four sites by 8 April 2004. We did not perform any additional site preparation such as replanting, harrowing, burning, or herbicide application.

Cover board study

We initiated a cover board study in 2005 to determine the relative abundance and microhabitat preferences of reptiles and amphibians in the four treatments. In April 2005, we placed eight wooden cover boards along a transect running down the approximate center of each quadrant at all four experimental sites (Fig. 1). The cover boards were made of 1.9-cm thick untreated plywood and measured 58 cm × 119 cm. We placed the cover boards 5, 20, 35, 50, 75, 100, 125, and 150 m from the wetland edge at the center of each site (Fig. 1). Beginning 1 June 2005 and ending 31 July 2005, we checked cover boards every 3–4 days between 0800 and 1300 hours, recording the presence and location of *S. invicta* colonies and any herpetofaunal species found under cover boards. We did not treat ant mounds with pesticides or disturb them any more than was necessary to look beneath cover boards. We used an analysis of variance (ANOVA) with experimental sites as replicated blocks to test for differences among the four treatments in the total number of reptiles and amphibians found under cover boards. We also used an ANOVA with experimental sites as replicated blocks to test whether the number of cover boards colonized by *S. invicta* differed among the four treatments.

Enclosure study

In 2005, we also initiated a short-term study of dehydration rates of two species of salamander (marbled salamanders, *Ambystoma opacum*, and mole salamanders, *A. talpoideum*) at two of the experimental sites (Bay 1000 and Bay 5148). For this study, we installed 12 small enclosures in a 3 × 4 grid (1.1 m apart) in the center of each quadrant, following Rothermel and Luhring (2005). Strips of fiberglass screening (46 cm high × 66 cm wide) were caulked to the upper 6 cm of a 24-cm section of 15.2-cm diameter galvanized pipe or PVC pipe. Overlapping edges of the screen were then hand-sewn with a needle and 20-lb monofilament fishing line. Each open cylinder was then buried in the ground so that only the screen tops remained above ground. Burrows were constructed in half of the enclosures by driving a section of 2.2-cm diameter polyvinyl chloride (PVC) pipe 10 cm into the ground at an approximately 30° angle. After adding the salamander, the enclosure was closed from the top by rolling the screen down and securing with binder clips.

Prior to the start of the experiment, we assessed the microhabitat within 0.5 m of each enclosure (Bartelt et al. 2004; Watson et al. 2003). We measured litter depth and visually estimated the percentage of ground cover that was bare soil, leaf litter (including twigs <4 cm diameter), herbaceous vegetation (including vines), and shrub vegetation (woody plants <7.6 cm DBH). We also recorded the presence of coarse woody debris (≥ 4 cm diameter, including stumps) and presence of foliage >1 m high.

The *A. opacum* used in this experiment were recently metamorphosed juveniles collected from Rainbow Bay on the Savannah River Site in South Carolina on 17 May 2005. The *A. talpoideum* used in this experiment were collected as larvae from Ellenton Bay on the Savannah River Site and raised to metamorphosis in cattle tanks. Forty-eight postmetamorphic salamanders of each species were collected and kept in plastic trays lined with moist paper towels and fed crickets ad libitum until three days prior to the experiment.

Salamanders were transferred to individual containers containing 1 cm of water at 1700 h on 29 June 2005, one day prior to the start of the experiment. On the day of the experiment, we measured the SVL and initial mass of each salamander to the nearest 0.01 g using a Scout II electronic balance. Salamanders were then randomly assigned to an enclosure, transported to the field sites, and added to the enclosures between 1845 and 2230 h on 30 June. An i-button temperature logger was added simultaneously with the salamander to each individual enclosure and used to record the hourly temperature. We also measured the soil moisture within each enclosure using a TH_2O portable soil moisture meter.

After the first 12 h, we returned to weigh salamanders, measure soil moisture, and record whether salamanders with burrows were in or out of the burrow. The original goal of the experiment was to measure water loss of salamanders over a 72-h period and compare dehydration rates between species and among treatments. However, at 24 h, a thunderstorm with heavy rain prevented us from weighing salamanders and also gave them an opportunity to rehydrate, so we simply recorded precipitation and whether or not the salamanders were alive. At 48 h, we recorded precipitation, soil moisture, and salamander mass. We decided to terminate the study and remove surviving salamanders at 48 h because we observed unexpectedly high levels of predation of salamanders by red imported fire ants.

Mass lost during the 48 h was attributed to water loss. Thus, we used the percent reduction in mass relative to initial (fully hydrated) mass as a response variable indicative of dehydration. We tested the effects of forest management treatment, burrow availability, and species on dehydration rate using repeated measures analysis of variance (ANOVA; PROC GLM). We used stepwise logistic regression (PROC LOGISTIC; SAS 9.1) to determine which microhabitat characteristics most affected *S. invicta* predation of salamanders. Because no salamanders were predated by *S. invicta* in the two forested treatment types, we modeled the probability of salamanders in clearcuts being predated by *S. invicta* as a function of the following independent variables: burrow availability, species, litter depth, percent cover of bare ground, percent cover of herbaceous vegetation, percent cover of shrubs, presence of coarse woody debris, and presence of foliage. We used $\alpha = 0.15$ as the significance criterion for entry of a variable into the model and $\alpha = 0.20$ as the criterion for removal (Hosmer and Lemeshow 2000).

Results

Cover board study

We captured 38 animals of 9 amphibian and reptile species under cover boards from 7 June to 31 July 2005, including marbled salamanders, *Ambystoma opacum*, green anoles, *Anolis carolinensis*, southern toads, *Bufo terrestris*, black racers, *Coluber constrictor*, southeastern five-lined skinks, *Eumeces inexpectatus*, five-lined skinks, *E. fasciatus*, eastern narrow-mouthed toads, *Gastrophryne carolinensis*, ground skinks, *Scincella lateralis*, and southeastern crowned snakes, *Tantilla coronata*. There was no significant difference in the number of amphibians and reptiles captured under cover boards among treatments, although the ANOVA model fit the data poorly ($F_{3,9} = 0.22$, $P = 0.88$, $R^2 = 0.20$). In contrast, cover board colonization by *S. invicta* varied significantly among treatments ($F_{3,9} = 28.83$, $P < 0.001$; Fig. 2). Least significant difference tests revealed that the most altered habitat, clearcuts with coarse woody debris removed (CC-removed), had the most cover boards colonized by *S. invicta* ($P < 0.05$). Clearcuts with coarse woody debris retained (CC-retained) also had significantly more cover boards colonized by *S. invicta* than either of the two forested treatments, which did not differ from each other (Fig. 2). Of the 38 animals captured during two months, only one animal was ever captured under a cover board after it had been colonized by *S. invicta*, a small *E. inexpectatus*.

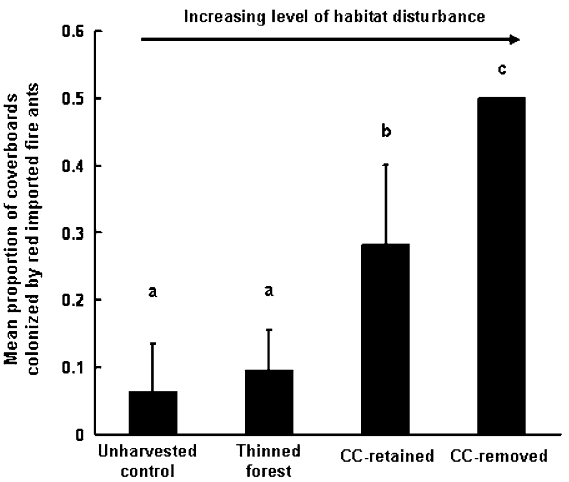

Fig. 2 Mean (±SE) proportion of cover boards colonized by red imported fire ants in each treatment from 1 June 2005 to 31 July 2005. Treatments are arranged in order of increasing disturbance (n = 4 replicates; see methods for details)

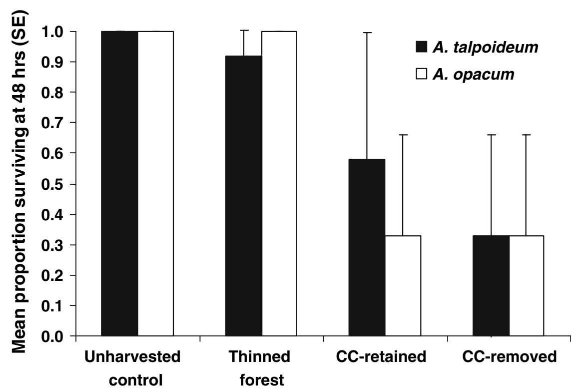

Fig. 3 Mean (±SE) percent survival of juvenile *Ambystoma talpoideum* (AMTA) and *A. opacum* (AMOP) during a 48-h enclosure study in June 2005 (n = 2 replicates)

Enclosure study

Although survival in both clearcut treatments was low, the most disturbed treatment (CC-removed) had the fewest surviving salamanders of all treatments (33.3% survived). In contrast, survival was greater in forested treatments, with 100% survival in the unharvested controls (Fig. 3). Survival was also greater at Bay 1000 than at Bay 5148 (87.5% and 50.0%, respectively). With one exception, all salamander deaths were due to predation by *S. invicta*. The cause of death was unknown for one salamander in a thinning treatment. Overall, survival of *Ambystoma opacum* and *A. talpoideum* were comparable (67% vs. 71% respectively).

The stepwise logistic regression procedure resulted in a final model that included only one microhabitat variable, percent shrub cover within 0.5 m of the enclosure (SHRUBCOV Wald $\chi^2 = 5.0189$, d.f. = 1, $P = 0.0251$). The final model was significantly better than the intercept-only model (Likelihood Ratio Test, $\chi^2 = 8.9946$, d.f. = 1, $P = 0.0027$) and adequately fit the data according to a Hosmer-Lemeshow Goodness-of-Fit Test ($\chi^2 = 8.5663$, d.f. = 6, $P = 0.1995$). The estimated odds ratio for each 10% reduction in shrub cover was 2.084 (95% Wald confidence limits: 1.096, 3.962).

Water loss (proportional mass loss relative to initial mass) over 48 h varied among habitat treatments ($F_{3,52} = 2.67$, $P = 0.0570$, $n = 63$; Fig. 4). Furthermore, there were significant effects of site ($F_{1,52} = 8.94$, $P = 0.0042$, $n = 63$) and species ($F_{1,52} = 39.76$, $P < 0.0001$, $n = 63$; Fig. 4) on water loss over 48 h and these effects were consistent across treatments (Treatment × Species $F_{3,52} = 0.42$, $P = 0.7384$, $n = 63$). Burrow availability did not affect water loss ($F_{1,52} = 0.42$, $P = 0.5209$, $n = 63$). However, rain caused some burrows to collapse, and at 24 h, we found many salamanders on the surface of the leaf litter, presumably absorbing water. In addition, several *A. talpoideum* that were not provided with artificial burrows managed to burrow under the top layer of soil.

Discussion

Clearcutting dramatically changes forest habitat, and extensive site preparation for replanting or other conversion typically eliminates ground cover and understory vegetation. As a result, leftover patches of litter or coarse woody debris have been suggested as providing the only remaining refugia available for many small amphibians and reptiles that would otherwise succumb to harsh environmental conditions created by clearcuts (deMaynadier and Hunter 1995). Using our wood cover boards, we found that woody debris in clearcuts is heavily colonized by *S. invicta*, turning apparent refugia into predatory traps where small reptiles and amphibians may be consumed or otherwise molested by stinging, invasive fire ants. *Solenopsis invicta* are known to prey on amphibians

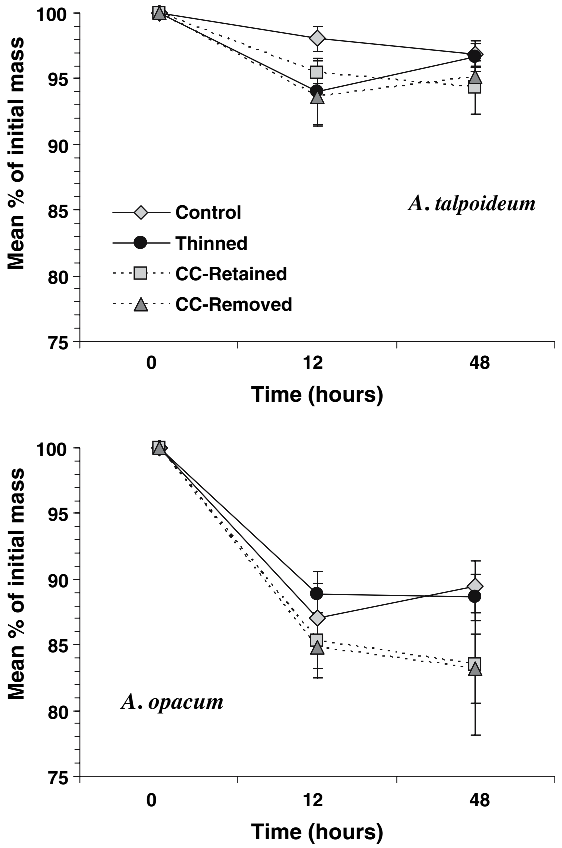

Fig. 4 Mean (±SE) percent of initial body mass lost over 48 h of juvenile *A. talpoideum* and *A. opacum* in four forest management treatments. Data are pooled across bays and burrow treatment

and reptiles and have been documented killing or consuming eastern box turtles, *Terrapene carolina*, Houston toads, *Bufo houstonensis*, and hatchling alligators, *Alligator mississippiensis* (Freed and Neitman 1988; Montgomery 1996; Allen et al. 1997). Additionally, *S. invicta* are known predators of reptile eggs (Moulis 1996; Buhlmann and Coffman 2001), and may negatively affect a greater diversity of oviparous reptiles than previously assumed, simply by predating nests laid in, or near, open habitats. Therefore, it is likely that *S. invicta* exacerbate negative effects caused by large-scale habitat alteration such as forest clearcutting.

We did not find any evidence that forest treatment affected the number of amphibians and reptiles captured under cover boards. However, with very few animals captured, our power to detect differences in abundance among treatments was low. Ryan et al. (2002) also showed that capture rates of amphibians and reptiles under cover boards can be low compared to other survey methods. Thus, greater effort or longer sampling may have been necessary to detect differences in animal abundance among our treatments. Unfortunately, vast cover board infestations in altered treatments by *S. invicta* may hamper the use of cover boards to compare herpetofaunal diversity and abundance among habitats, a technique that is critical to many comparative studies (e.g., Heyer et al. 1994). We captured only a single animal under a cover board that had been colonized by *S. invicta* during the two months of the study, possibly because the use of cover objects by amphibians or reptiles is negatively correlated with *S. invicta* presence. This is a topic that has not been studied but which warrants further investigation. Importantly, if *S. invicta* influence the distribution of amphibians and reptiles under cover boards, many studies that rely on cover boards to determine habitat effects on amphibians and reptiles will likely have substantial biases that can affect interpretations and subsequent management recommendations.

Although postmetamorphic salamanders experienced marginally higher dehydration rates in clearcuts than in forested treatments, predation by *S. invicta* was the more significant source of mortality in our short-term enclosure study. We suspect that in the absence of precipitation, there would have been a highly significant effect of habitat treatment on water loss (e.g., Rothermel and Luhring 2005). Dehydration occurred more rapidly during the first 12 h of the experiment than in the 24 h following the rainfall (Fig. 4). However, *S. invicta* had already begun preying on salamanders within 12 h and ultimately accounted for 29 of 30 salamander deaths. Known *S. invicta* predation occurred exclusively in the clearcuts and resulted in an overall mortality rate of 30% within 48 h. This was unexpected; only one of 48 *A. talpoideum* died as a result of *S. invicta* predation in a similar study conducted in 2004 at Bay 1000 (Rothermel and Luhring 2005). The occupation phase of *S. invicta*, in which population expansion occurs, can take several years (Porter et al. 1988); thus, the increased fire ant predation in 2005 likely reflected an increase in abundance of *S. invicta* within the clearcuts in the 17 months since logging occurred.

Although we certainly did not intend to subject caged salamanders to fire ant predation, we believe the outcome of our enclosure experiment offers some important insights into the vulnerability of amphibians to this invasive predator. The results of our logistic regression suggest that the risk of fire ant predation is related to small-scale differences in vegetative cover. We found that the risk of predation by fire ants approximately doubled with every 10% reduction in shrub cover. Thus, small amphibians that move through, or inhabit, relatively open microhabitats where there is little shade from woody vegetation may be more vulnerable to foraging ants. We do not think the outcome of the predator–prey interaction would have been different had the salamanders not been caged. Ambystomatid salamanders are ground-dwelling, relatively slow-moving animals and are not capable of leaping or hopping to evade predators. If *S. invicta* can prey on hatchling alligators (Allen et al. 1997), then it is very likely they can prey successfully on mole salamanders. Juvenile salamanders are probably even less able to escape than are larger adults. Finally, because ambystomatids are only active nocturnally, they would be vulnerable to foraging *S. invicta* that locate them in their burrows during the day. According to our logistic regression analysis, burrow availability in the salamander enclosures did not reduce the probability of fire ant predation, suggesting that salamanders with burrows were equally susceptible to fire ant predation as salamanders without burrows.

Overall, our results revealed greater abundance of *S. invicta* in disturbed habitats compared to undisturbed habitats, similar to findings in other studies. For example, Stiles and Jones (1998) found that *S. invicta* mounds were more common in disturbed habitats such as active power line rights-of-way than along dirt roads in undisturbed closed-canopy forests. They also reported that *S. invicta* mounds were found more frequently along roadsides and forest edges than expected at random. Also, Zettler et al. (2004) found that clearcutting in deciduous forests in South Carolina increased *S. invicta* populations, results that agree with our findings in pine forests on the SRS. For amphibians and reptiles that may already be adjusting to the environmental changes that accompany forest harvesting, *S. invicta* pose an additional challenge with which they must cope.

Solenopsis invicta appear to be permanently established in much of their currently invaded range and they continue to expand across the United States. There is an urgent need to better study and document their effects on wildlife populations, particularly for susceptible and declining species such as many amphibians and reptiles. The possible synergistic effects of *S. invicta* and large-scale habitat alteration may lead to further population declines of amphibians and reptiles, particularly in the Southeast. Lastly, researchers and resource managers should be aware of, and consider studying, the possible effects that *S. invicta* can have on current monitoring and study techniques.

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Diet and food availability of the Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*): implications for dispersal in a fragmented forest

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A history of timber harvest in West Virginia has reduced red spruce (*Picea rubens*) forests to < 10% of their historic range and resulted in considerable habitat fragmentation for wildlife species associated with these forests. The Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*) has been described as a red spruce obligate subspecies that must traverse this fragmented landscape to disperse among remnant red spruce patches. Food availability in the forest matrix surrounding red spruce may be a limiting factor to successful dispersal of *G. s. fuscus*. We examined the diet of flying squirrels using stable isotope analysis and used vegetation surveys to determine the availability of diet items in the habitats encountered by *G. s. fuscus* in the matrix surrounding red spruce fragments. Stable isotope analysis suggested hypogeous fungi, epigeous fungi, and invertebrates contributed the most to the diet of *G. s. fuscus*, followed by lichen. Tree buds contributed the least in spring, and beechnuts contributed the least in fall. The vegetation surveys revealed that no habitat type had a greater availability of the diet items that contributed most to the assimilated diet of *G. s. fuscus*, suggesting that stand age and structure may be more important for diet-item availability than habitat type.

Key words: diet, dispersal, fragmentation, *Glaucomys sabrinus fuscus*, *Picea rubens*, red spruce, stable isotope analysis, Virginia northern flying squirrel, West Virginia

In the central Appalachians in West Virginia, broad-scale disturbance and other anthropogenic factors reduced red spruce (*Picea rubens*) to < 10% of its historic range within the last century (Stephenson 1993). Further habitat degradation and heavy metal deposition have impeded regeneration of red spruce, transitioning forests from red spruce to young, regenerating oak (*Quercus* spp.), maple (*Acer* spp.), and American beech (*Fagus grandifolia*)-dominated hardwood forests, which have not yet reached the complex stage that is characteristic of old-growth forests (Schuler et al. 2002; USFWS 2013). The patchy distribution and potential ongoing decline of high-elevation red spruce forest may threaten the long-term viability of spruce-adapted species with limited dispersal capabilities, such as the Virginia northern flying squirrel (*Glaucomys sabrinus fuscus*). Recently removed from the endangered species list (USFWS 2013), *G. s. fuscus* is thought to be reliant upon this threatened forest type and must disperse through young, managed stands to maintain a viable population (Hanski 1991; Arbogast et al. 2005).

Evidence of reduced genetic variability suggests limited dispersal of *G. s. fuscus* among the remaining pockets of red spruce (Arbogast et al. 2005). Successful dispersal through fragmented habitats is one of the most important components of maintaining a functional metapopulation (Levins 1969; Johnson and Gaines 1990; Gilpin 1991; Hanski 1991; Venebal and Brown 1993; Clobert et al. 2012). Without dispersal, populations suffer from decreased genetic variability (Gilpin 1991) and higher extinction risks (Fahrig and Merriam 1994; Thomas 2000). Habitat fragmentation may limit dispersal of flying squirrels by increasing dispersal time (Smith et al. 2011), which may reduce fitness through increased risk to predation (Smith 2012) or through indirect effects (e.g., increased energy costs during dispersal—Flaherty et al. 2010a), that ultimately reduce fitness (Stamps et al. 2005; Bonte et al. 2012). Flying squirrels are capable of moving several km per night while searching for a suitable home range (Weigl 1974; Selonen and Hanski 2004; Hanski and Selonen 2009; Pyare et al. 2010). However, if the energetic cost of dispersal exceeds the energy stored by dispersing individuals, forage availability may limit flying squirrel movements (Flaherty et al. 2010a, 2010b; Bonte et al. 2012).

Availability of diet items for northern flying squirrels may be influenced by timber harvest and age of forest stands. Hypogeous and epigeous fungi, staples of flying squirrel diets in the Pacific Northwest (Maser et al. 1985; Pyare et al. 2002; Flaherty et al. 2010b), exhibited reduced availability in younger, managed stands (Amaranthus et al. 1994; Luoma et al. 2004; Flaherty et al. 2010b). Furthermore, lichen abundance in New England was strongly influenced by forest structure, with a higher diversity and availability in stands with increasing stand age and complexity (Selva 1994). Therefore, identification of diet items that are most important to *G. s. fuscus* and their availability in habitat that must be utilized for dispersal will improve our understanding of potential limits to dispersal in northern flying squirrels (Smith 2012).

We estimated the availability of diet items in managed forest types surrounding red spruce and used stable isotope analysis to characterize the diet of *G. s. fuscus*. We hypothesized that diet items that provided the highest percent contribution to the assimilated diet of *G. s. fuscus* would be less available in managed conifer, hardwood, and mixed conifer-hardwood habitat types compared to mature red spruce habitat in West Virginia (Loeb et al. 2000; Flaherty et al. 2010b). Specifically, our overall research objective was to examine whether food resources may be a limiting factor for dispersal of *G. s. fuscus* in fragmented landscapes. Understanding potential limits to dispersal and ways to increase dispersal success provides important information for conservation practitioners when considering how to best maintain a functional metapopulation of *G. s. fuscus* and ensure population viability.

Materials and Methods

Study area.—Our field site was in the northern half of the Monongahela National Forest (MNF) and Blackwater State Park located in the Allegheny Mountains of West Virginia, United States. Periods of extensive logging, fire, and development in the early 19th century removed most of the established, old-growth forest and disrupted the natural succession of the region, resulting in what is now a predominantly regenerating young-growth forest with an altered species composition (Pielke 1981; Schuler et al. 2002). Dominant canopy trees included silver maple (*Acer saccharium*), yellow birch (*Betula alleghaniensis*), American beech, red spruce, and eastern hemlock (*Tsuga canadensis*). Mid-story trees included young American beech and rhododendron (*Rhododendron maximum*), and dominant forest-floor species included ferns, blueberries (*Vaccinum erythrocarpum*), and blackberries (*Rubus* spp.). Precipitation ranges from 120 to 150 cm annually, most of which comes from snowfall (Stephenson 1993), and average temperatures range from 0°C to 21°C.

Food-item availability surveys.—We used vegetation, pitfall, and truffle surveys to evaluate the availability of diet items (Flaherty et al. 2010b) identified in > 20% of fecal samples in a previous study of *G. s. fuscus* diet by Mitchell (2001). To account for differences in diet-item availability within habitat types, we surveyed 4 previously delineated habitat types: red spruce, conifer, hardwood, and mixed conifer-hardwood. Red spruce forest was defined as having > 50% red spruce cover mixed with other conifer or hardwood species, whereas conifer forest had > 50% cover of various conifer species (e.g., pine, *Pinus* spp.) other than spruce. Mixed hardwood-conifer forest was defined as having an even mixture of conifer and hardwood species, whereas hardwood forest had < 10% conifer in the overstory. We surveyed a total of 60 stratified random plots, 15 in each of the 4 habitat types, and conducted the full suite of vegetation, pitfall trap, and truffle surveys (described below) at each site. We used preliminary data from the surveys completed during the 2014 field season to determine the total number of survey transects needed to detect differences in vegetation and food availability among the 4 habitat types with a statistical power of 0.80. Using G*Power software (Faul et al. 2007) and *F*-test fixed-effects analysis of variance (ANOVA) procedure, we determined that 15 survey sites in each habitat type would provide adequate power to evaluate differences among habitats. We restricted plots to < 350 m from the road for accessibility.

We used point-center quadrat plots with a diameter of 20 m. We placed 2 50-m field tapes (OTR50M; Keson, Aurora, Illinois) in the 4 cardinal directions (Flaherty et al. 2010b). To estimate availability of red spruce and American beech at each plot, we measured the distance to the nearest tree in each quadrat from the center of the plot, identified each tree to species, estimated height using a digital hypsometer (Nikon Forestry PRO Laser Rangefinder/Hypsometer; Nikon Vision Co., Ltd., Tokyo, Japan), and measured diameter at breast height (DBH) using a DBH tape (No. 59571; Forestry Suppliers). We calculated the importance value, the dominance of a tree species at a site (Cottam and Curtis 1956; Loeb et al. 2000; Schuler et al. 2002), for red spruce and American beech on each plot as:

$$(\% \text{ relative basal area} + \% \text{ relative density} + \% \text{ relative frequency}). \tag{1}$$

This importance value for spruce has been used in other studies as a surrogate for availability of hypogeous fungi (Loeb et al. 2000).

At each of the 4 ends of the tape, we counted red spruce cones in a 1 × 1 m plot as an index of availability of conifer seeds, and then used a hand rake to search for truffles, the fruiting bodies of hypogeous fungi, in the organic soil layer, measuring fresh truffle biomass with a Pesola scale (10 g; Pesola AG, Schindellegi, Switzerland). We measured % cover along the 20-m tapes to estimate the availability of epigeous fungi and downed wood. To estimate invertebrate abundance, we buried 473-ml cups flush with the ground as pitfall traps. We placed 5 traps every 5 m along a randomly selected field tape and covered the cups with plastic plates to provide cover and protection from rain and with space between the ground and plate to allow invertebrates to walk underneath. Pitfall traps remained in the ground for 4 days and contents were then stored frozen to preserve the specimens for later identification. We identified invertebrates to class using Borror and White (1998).

For measurements of downed wood and invertebrates, we calculated the mean for each plot and used an ANOVA with a log transformation to adjust for non-normal data. For availability of epigeous fungi and red spruce and American beech importance values, we used a Kruskal–Wallis test to evaluate differences among habitat types (Zar 2010). To estimate the detection probability of invertebrates among habitat types, we used a proportion *z*-test (Zar 2010:549). We collected 10 independent samples for each diet item identified by Mitchell (2001) from red spruce habitats for stable isotope analysis.

For all statistical analyses performed during our study, we accepted a probability of $\alpha = 0.05$ to indicate statistical significance.

Hair collection.—We deployed modified Tomahawk live traps (No. 201; Tomahawk Live Traps, Hazelhurst, Wisconsin—Trapp and Flaherty in press) that served as minimally invasive, single-capture hair snares in the MNF to collect hair from *G. s. fuscus*. We used plastic zip ties and wire to disable the locking mechanism that would normally have prevented captured individuals from escaping the trap. As the trapped individual pushed the door open to escape, they contacted 4 wire brushes (6.35 cm diameter; The Mill-Rose Company, Mentor, Ohio) attached to the perimeter of the door that snared hair samples upon contact. The trap door then closed behind the individual, thereby creating a single-capture system. In 2015, we suspended wooden dowel rods wrapped in double-sided packing tape from the sides of the trap and behind the treadle to increase the volume of collected hair as the individual moved through the trap (Suckling 1978; Sanecki and Green 2005; Schwingel and Norment 2010).

We attached traps horizontally 1.5 m from the ground on the bole of a tree following procedures described in Carey et al. (1991). We baited traps with a mixture of peanuts, peanut butter, oats, and molasses, switching to black oil sunflower seeds during periods of high black bear (*Ursus americanus*) activity. Bait was suspended from the top of the trap using a paperclip and wax paper to reduce bait consumption by mice (*Peromyscus* spp.). We covered the traps with a tarp to protect the bait and brushes from rain.

We deployed 10 snares at 4 locations: 2 along Canaan Loop Road (39.074 N, −79.471 W) and 2 in Blackwater State Park (39.112 N, −79.491 W) of West Virginia from May to October 2014. Because *G. s. fuscus* is a species of conservation concern, we checked traps daily to prevent any permanent captures until fully confident the traps functioned as expected and squirrels would escape, at which point we checked the traps every 3 days. If traps could not be checked as scheduled, we deactivated the traps until regular checking resumed.

We collected brushes and tape from closed traps for processing and set the traps with new brushes and tape. We removed any hair from the wire brushes and tape with tweezers, and placed collected hair in coin envelopes or microcapillary tubes with silica desiccant. We froze the coin envelopes and stored the microcapillary tubes at room temperature until mailing the samples to the Wildlife Physiology Lab in the Department of Forestry and Natural Resources at Purdue University for identification and processing. Before reusing brushes, we used an open flame on the bristles to remove any residual hair. We replaced tape on dowel rods for deployment. We identified the hair samples to species using morphological features under a compound microscope based on methods in Trapp and Flaherty (in press).

Additionally, we received hair samples from nest boxes from the USFS Greenbrier Ranger District (located approximately 50–60 km southwest of Davis, West Virginia, in the MNF) for use in stable isotope analysis. These hair samples were collected directly from individual flying squirrels during nest box checks conducted by Forest Service and West Virginia Division of Natural Resource biologists. All methods were approved by Purdue University's Institutional Animal Care and Use Committee (Protocol #1310000959) and followed guidelines established by the American Society of Mammalogists (Sikes et al. 2011).

Stable isotope diet analysis.—We processed samples for stable isotope analysis in the Wildlife Physiology Lab at Purdue University. We cleaned and removed lipids from hair samples using a 2:1 chloroform:methanol solution and dried diet items collected during diet-availability surveys and hair for 48 h at 60°C (Cryan et al. 2004; Pauli et al. 2009). We then used a mixer mill (Retsch MM 200; Glen Mills Inc., Clinton, New Jersey) to grind each diet-item sample into a fine powder, and cut the hair into small fragments using scissors. We weighed subsamples of each sample in miniature tin weigh boats (4 × 6 mm; Costech Analytical Tech Inc., Valencia, California) using a Sartorius microbalance (model CPA2P; Arvada, Colorado) and submitted the samples to the University of Wyoming Stable Isotope Facility (UWSIF) for final analysis of stable isotope signatures. When sample quantity allowed, we weighed each sample in duplicate for quality control, and accepted sample results if the variance between the 2 subsamples did not exceed the variance of the standards (Ben-David and Flaherty 2012). Isotope data were obtained using a Costech 4010 Elemental Analyzer (Costech Analytical Technologies, Valencia, California) coupled to a Thermo DeltaplusXP IRMS mass spectrometer (Thermo Fisher Scientific, Inc., Waltham, Massachusetts). PeeDee Belemnite and atmospheric air were used as standards for $\delta^{13}C$ and $\delta^{15}N$, respectively, and the UWSIF used Glutamic 1 and Glutamic 2 as quality control reference materials. The average standard uncertainty for both $\delta^{13}C$ and $\delta^{15}N$ was 0.1 during analysis.

Using a multivariate analysis of variance (MANOVA) and post hoc Tukey's multiple comparison test as well as a k-nearest neighbor randomization test (Rosing et al. 1998), we delineated diet items into groups (Zar 2010), combining diet items that did not differ significantly ($P > 0.05$) in $\delta^{15}N$ and $\delta^{13}C$ values. We used a MANOVA to determine if the isotopic signatures differed among seasons and localities.

We used the dual-isotope linear mixing model SISUS (Stable Isotope Sourcing Using Sampling—Erhardt et al. 2014) to determine the proportional contribution of potential food items to the diet of *G. s. fuscus* by comparing the ratio of heavy to light isotopes of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) of the hair (Supplementary Data SD1) to that of the potential diet items. Initial analyses using Bayesian mixing models (Moore and Semmens 2008) defaulted to uninformative priors likely due to distribution of the data; therefore, we used SISUS to avoid problems with model convergence. To account for diet-consumer discrimination, we used a transformation of 5.3‰ for $\delta^{13}C$ and 3.5‰ for $\delta^{15}N$ based on a similar diet study of another mycophagist mammal, the red-backed vole (*Myodes gapperi*—Sare et al. 2005). Because of changes in discrimination when switching to a higher-protein diet (Kurle et al. 2014), such as when adding beechnuts to the diet, we increased the $\delta^{13}C$ to 5.7‰ for the spring diets based on a similar increase for hair samples when switching laboratory rats from a wheat diet to a wheat-fish diet (Kurle et al. 2014).

Results

Food availability.—The availability of some diet items differed among habitats. The importance value for red spruce (surrogate for hypogeous fungi availability—Loeb et al. 2000) was 1.8 times higher in red spruce habitat than conifer ($z = -1.76$, $P = 0.04$) and mixed conifer-hardwood ($z = -1.76$, $P = 0.04$) habitat. There was no difference in red spruce importance values between conifer and mixed conifer habitat ($z < 0.001$, $P < 0.50$). The hardwood habitat had no red spruce present, and therefore had an importance value of 0.00 (Table 1).

The importance value for American beech was highest in the mixed conifer-hardwood habitat, being 3.2 times more important in mixed conifer-hardwood than in conifer habitat ($z = -1.93$, $P = 0.02$), and 2.68 times more important than in spruce habitat ($z = -1.93$, $P = 0.02$). Mixed conifer-hardwood and hardwood habitats had similar American beech importance values ($z = -0.58$, $P = 0.28$). The American beech importance values for conifer, hardwood, and spruce habitats did not differ significantly (Table 1).

Epigeous fungi were 1.27 times more available in conifer habitat than in mixed conifer-hardwood habitat ($z = 1.68$, $P = 0.04$), and 1.17 times more available than in spruce habitat ($z = 1.98$, $P = 0.02$). The difference between conifer and hardwood habitats had a marginal P-value, suggesting a possible difference in epigeous fungi availability ($z = 1.44$, $P = 0.07$). The availability of epigeous fungi did not differ between hardwood habitat and mixed conifer-hardwood habitat ($z = 0.26$, $P = 0.39$) or spruce habitat ($z = 0.51$, $P = 0.30$); nor did epigeous fungi availability differ between mixed conifer-hardwood and spruce habitat ($z = 0.24$, $P = 0.41$; Table 1).

Differences in abundance of downed wood, which also may be related to availability of hypogeous fungi (Clarkson and Mills 1994), had a marginal P-value, suggesting a possible difference across habitat types ($F_{3,56} = 2.53$, $P = 0.07$; Table 1). We did not encounter any fallen arboreal lichen during the % occurrence surveys or any hypogeous fungi truffles in our truffle plots.

From the pitfall traps, we collected 275 specimens distributed among 11 taxa of invertebrates. Chilopoda made up 22% of the collected specimens, followed by Diplopoda at 21% and Coleoptera at 18%. Psocoptera were collected at the most plots (58%), followed by Diplopoda (48%) and Chilopoda (45%). Of the 3 most abundant invertebrates (Chilopoda, Diplopoda, Coleoptera), only Chilopoda differed in detection rate across habitat types, being captured in 3 times as many plots in conifer habitat than in spruce habitat ($z = 2.24$, $P = 0.03$; Table 2). Overall, invertebrate abundance did not differ among habitat types ($F_{3,56} = 0.62$, $P = 0.61$; Table 1).

Hair collection.—From May 2014 to October 2014, we collected 159 and 157 hair samples at Yellow Birch Trail and Canaan Loop Road, respectively, for a total of 316 samples. Of the 316 samples, 42 were identified as *G. s. fuscus*, of which 24 had > 0.250 µg for use in stable isotope analysis. Fifteen hair samples were from Yellow Birch Trail, and 9 were from Canaan Loop Road. Additionally, we received 9 samples from nest boxes located in the Greenbrier Ranger District. In 2015, we collected 3 hair samples from hair snares near Davis, West Virginia, and 3 were collected using hair snares in the Greenbriar Ranger District. Of the 40 hair samples processed for stable isotope analysis, 21 samples were collected in spring and 19 samples collected during fall (Table 3).

Stable isotope analysis.—The mean isotopic signature for all *G. s. fuscus* hair was $\delta^{13}C = -20.28$ (± 1.19 *SD*) and $\delta^{15}N = 5.54$ (± 1.97 *SD*). The isotopic signature for *G. s. fuscus* differed among locations ($F_{10,64} = 4.445$, $P < 0.001$; Table 3) and years ($F_{2,37} = 4.904$, $P = 0.013$), but did not differ among seasons ($F_{2,37} = 0.526$, $P = 0.595$). Because beechnuts were not available in spring, and tree buds were consumed in spring and rarely in fall, we analyzed the 2 seasons separately. Based on molting patterns of northern flying squirrels (with one primary molt generally beginning in May—Villa et al. 1999) and results from the fecal analysis in a previous study (Mitchell

Table 1.—Mean (± *SD*) importance values and measurements used to evaluate habitat for *Glaucomys sabrinus fuscus* within the 4 habitat types in the Monongahela National Forest, West Virginia, United States, from August 2014 to August 2015.

	Conifer	Hardwood	Mixed conifer	Spruce
Red spruce importance value	51.60 ± 19.55	0.00 ± 0.00	51.75 ± 19.56	95.24 ± 22.12
American beech importance value	10.71 ± 28.28	32.98 ± 63.94	34.83 ± 49.97	13.10 ± 37.14
Epigeous fungi transect intersection (cm)	2.40 ± 0.83	1.64 ± 1.28	1.89 ± 1.33	2.06 ± 0.95
Downed wood transect intersection (cm)	72.07 ± 82.40	50.27 ± 38.14	142.21 ± 140.70	135.55 ± 105.53
Lichen transect intersection (cm)	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
Invertebrate abundance (number of invertebrates/site)	2.01 ± 2.16	2.07 ± 2.07	1.38 ± 1.08	1.76 ± 1.49

Table 2.—Number of survey plots with successful captures for each taxa of invertebrate and the total number of specimens captured in West Virginia in 2014 and 2015. Each habitat type was sampled using 15 plots. The asterisk denotes a significant difference at the $\alpha = 0.05$ level.

	Conifer	Hardwood	Mixed-conifer	Spruce	Total	Total specimens captured
Acarina	2	3	1	2	8	8
Aranaea	4	5	4	3	16	21
Chilopoda	9*	7	8	3*	27	62
Coleoptera	7	8	7	11	33	51
Diplopoda	7	9	6	7	29	58
Diptera	1	0	0	0	1	1
Ensifera	1	5	0	4	10	18
Formicidae	0	1	0	1	2	2
Hymenoptera	1	0	0	0	1	1
Isopoda	0	1	2	0	3	3
Psocoptera	8	9	7	11	35	50

Table 3.—Mean (± *SD*) isotopic values for $\delta^{13}C$ and $\delta^{15}N$ of *Glaucomys sabrinus fuscus* hair samples ($n = 40$) for spring and fall in 2014 and 2015. For each year, the mean (± *SD*) isotopic values for $\delta^{13}C$ and $\delta^{15}N$ are provided for 5 locations in the Monongahela National Forest, West Virginia, United States.

	Year	*n*	$\delta^{13}C$	$\delta^{15}N$
Spring	2014	21	−20.08 ± 1.15	5.88 ± 1.93
Fall	2014	12	−20.00 ± 0.65	5.08 ± 1.36
Canaan Loop Road	2014	9	−20.34 ± 1.38	5.43 ± 1.66
Yellow Birch Trail	2014	15	−19.73 ± 0.80	5.54 ± 1.67
Nest Boxes	2014	9	−20.29 ± 0.75	5.84 ± 2.16
Spring	2015	1	−21.87	4.63
Fall	2015	6	−21.28 ± 2.78	5.42 ± 10.26
Davis, West Virginia	2015	3	−20.23 ± 3.05	8.21 ± 0.77
Upper Greenbriar	2015	3	−22.33 ± 0.73	2.61 ± 1.13

et al. 2001), we included beechnuts in the spring analysis as a diet item because the hair from individuals collected in spring should reflect the diet of the individual during fall (Dalerum and Angerbjörn 2005). Similarly, the fall analysis included tree buds because the hair collected in fall should reflect the diet of the individual during spring. However, there was no observed difference in the isotopic signature of hair between spring and fall (Table 3).

Diet items were combined into 4 groups based on a MANOVA with a post hoc Tukey's multiple comparison test ($F_{22,286} = 42.16$, $P < 0.01$; Table 4) and k-nearest neighbor ($P < 0.01$). The isotopic signature for 3 of the diet items in the groups differed across years (birch: $F_{2,16} = 13.00$, $P < 0.01$; maple: $F_{2,12} = 10.54$, $P < 0.01$; and spruce buds: $F_{2,11} = 10.32$, $P < 0.001$). These differences likely occurred because of variations in precipitation between years (121 cm in 2014, 132 cm in 2015—NOAA 2016), as well as small sample sizes (Ben-David and Flaherty 2012). Despite the temporal differences (collecting potential food items in different years), when we entered year as a covariate in the MANOVA, the variables were categorized into the same groups suggesting there was no significant difference between years.

Across seasons and locations, truffles, epigeous fungi, and invertebrates were the most important diet items based on the results of the stable isotope mixing model, contributing between 0.66 (fall) and 0.71 (spring) to the assimilated diet,

Table 4.—Sample size (*n*) and mean isotopic signature (± *SD*) for $\delta^{13}C$ and $\delta^{15}N$ for each collected diet item for *Glaucomys sabrinus fuscus* in August 2014 and August 2015 in the Monongahela National Forest, West Virginia, United States. Group letters represent a significant ($\alpha = 0.05$) difference in $\delta^{13}C$ and $\delta^{15}N$ values calculated from a multivariate analysis of variance (MANOVA) with a post hoc Tukey's multiple comparison test (Zar 2010) and a k-nearest neighbor analysis (Rosing et al. 1998).

Diet item	*n*	$\delta^{13}C$	$\delta^{15}N$	Group
Bait	9	−27.89 ± 0.28	0.04 ± 0.13	A
Beechnuts	8	−31.99 ± 01.30	−1.38 ± 0.81	A
Red spruce seeds	6	−25.54 ± 1.03	−3.00 ± 0.88	B
Lichen	10	−25.21 ± 1.13	−4.72 ± 1.25	B
American beech buds	16	−31.99 ± 1.30	−1.21 ± 1.15	C
Yellow birch buds	19	−31.56 ± 1.72	−2.62 ± 1.82	C
Blueberries	10	−29.48 ± 0.92	−3.26 ± 1.92	C
Red maple buds	15	−28.88 ± 2.16	−2.95 ± 2.38	B
Red oak buds	13	−30.58 ± 1.55	−1.80 ± 1.35	C
Red spruce buds	14	−30.33 ± 1.27	−2.10 ± 1.58	C
Epigeous fungi	19	−25.59 ± 1.30	4.96 ± 3.41	D
Invertebrates	16	−25.64 ± 0.72	3.76 ± 1.42	D
Hypogeous fungi	10	−26.66 ± 1.39	5.5 ± 1.67	D

followed by lichens, which ranged from 0.22 (spring) to 0.33 (fall), and beechnuts and vegetation at 0.07 and 0.01, respectively (Table 5; Fig. 1).

Discussion

The stable isotope analysis revealed that hypogeous fungi, epigeous fungi, invertebrates, lichen, and beechnuts were dominant components of the diet of *G. s. fuscus.* Our results demonstrate the value of stable isotope analysis as a follow-up to fecal analysis. Mitchell et al. (2001) found a high proportion of fecal pellets from *G. s. fuscus* in the MNF with tree buds present, which may have suggested a high contribution of tree buds to the assimilated diet of *G. s. fuscus.* Our results show, however, that whereas buds may be consumed by squirrels, these items are not highly assimilated into the tissues of *G. s. fuscus*. Furthermore, Mitchell et al. (2001) reported a decrease in lichen consumption in the fall, whereas our results suggest lichens play an increased role in the fall diet of *G. s. fuscus*. The disparities between the fecal analysis and

Table 5.—Relative contribution to the diet of *Glaucomys sabrinus fuscus* during spring and fall 2014–2015 in the Monongahela National Forest of West Virginia, United States. Proportions of diet items relative to the overall squirrel diet were estimated using a dual-isotope mixing model.

Diet item	Relative contribution	
	Spring	Fall
Hypogeous fungi, epigeous fungi, and invertebrates	0.66	0.71
Lichens	0.33	0.22
Vegetation	0.01	—
Beechnuts	—	0.07

stable isotope analysis may be explained by the differences in digestibility of various diet items. A wide variety of tree buds appear to be consumed by *G. s. fuscus*, but their apparent low digestibility increases their prominence in the fecal pellets. Lichen is highly digestible (Robbins 1987), and therefore when consumed may not be present in fecal matter or recognized by observers. *G. s. fuscus* may assimilate lichen more in the fall, which would account for the decrease in fecal pellet observations but increased tissue assimilation. Another consideration is the temporal relationship between diet and tissue or analysis method. Fecal samples reflect food items consumed during the last meal, whereas hair samples represent the overall assimilated diet since last molt (Dalerum and Angerbjörn 2005), which for northern flying squirrels was likely in spring (Villa et al. 1999).

Although truffles, epigeous fungi, and invertebrates comprised a majority of the diet of *G. s. fuscus*, we were unable to determine from stable isotope analysis alone which of the 3 contributed the most due to their similar isotopic signatures. A larger sample size as well as collection of specific species within each of the 3 categories may have provided sufficient evidence to differentiate separate groups. A similar study on Prince of Wales Island, Alaska, suggested that whereas invertebrate fragments were found in the fecal pellets of the Prince of Wales northern flying squirrel (*G. s. griseifrons*), a majority of them were small wings, which may be a result of incidental consumption while foraging for hypogeous and epigeous fungi (Flaherty et al. 2010b).

The consumption of tree buds in the spring and beechnuts in the fall suggests that these items supplement the diet of *G. s. fuscus*. Truffles and lichen are both low in nitrogen relative to foods with greater amounts of protein, and the nitrogen found in truffles may be indigestible by small mammals, including flying squirrels (Cork and Kenagy 1989; Dubay et al. 2008). Although tree buds and beechnuts do not provide the greatest proportional contribution to the assimilated $\delta^{15}N$ and $\delta^{13}C$ of *G. s. fuscus*, their consumption may provide alternative minerals or vitamins essential to their diet. Therefore, stable isotope analysis alone would not be able to reveal the potential importance of tree buds and beechnuts to the diet of *G. s. fuscus*. Our findings corroborate a similar study on the diet of *G. s. griseifrons* (Flaherty et al. 2010b) in Southeast Alaska. Their findings suggested a large

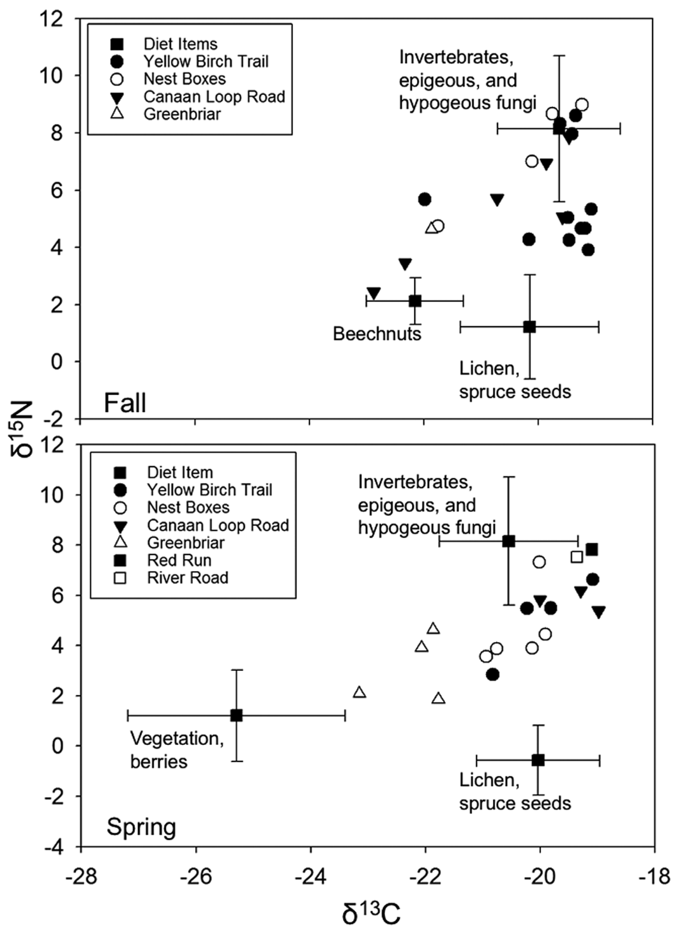

Fig. 1.—Range of isotopic means (± *SD*) for groups of potential diet items (squares) for *Glaucomys sabrinus fuscus* and isotopic signatures of individual squirrels from hair collected in the fall and spring of 2014 and 2015 in the Monongahela National Forest and Blackwater State Park, West Virginia, United States.

contribution of lichen to the overall diet of *G. s. griseifrons*, despite a similar lack of substantial amounts of lichen found in the fecal pellets of a previous study (Pyare et al. 2002). This suggests that at least 2 subspecies of northern flying squirrel may rely more on lichen as a staple diet item than previously recognized.

Our food availability surveys revealed differences in availability of diet items among habitat types. Based on the lower red spruce importance values in hardwood, conifer, and mixed-conifer habitats, hypogeous fungi appear to be largely unavailable to dispersing *G. s. fuscus* in the habitat surrounding red spruce stands. Red spruce is positively associated with hypogeous fungi (Loeb et al. 2000), but the variable and patchy distribution of hypogeous fungi made direct measurement of availability difficult. Abundance of downed wood, which is positively associated with hypogeous fungi truffles in the western part of the continent (Clarkson and Mills 1994), was not different across habitat types. However, Loeb et al. (2000) found no association between downed woody debris and the presence of hypogeous fungi truffles in the southern Appalachians, suggesting that downed wood may not be an indicator of truffle availability for *G. s. fuscus* and stand age may be a more reliable indicator.

Epigeous fungi were most available in conifer habitat, and are more available in conifer, mixed conifer-hardwood, and hardwood habitats than in red spruce habitat, suggesting epigeous fungi may be available to dispersing *G. s. fuscus*. Although arboreal lichen was not encountered during our fieldwork, surveys of lichen abundance in New England found various species of lichen in both conifer and hardwood habitats, suggesting an availability of lichen across habitat types (Selva 1994). However, Selva (1994) found a strong connection between lichen abundance and forest age, with higher lichen availability in more mature forests. Furthermore, the importance value for American beech, which may correspond to the availability of beechnuts, was higher in hardwood and mixed conifer-hardwood habitats than in red spruce habitat. The beechnut crop corresponds with the dispersal season of *G. sabrinus* (Villa et al. 1999), potentially providing forage for dispersing individuals. However, *G. s. fuscus* may encounter high levels of competition for beechnuts from hard-mast specialists, such as *G. volans* and red squirrels (*Tamiasciurus hudsonicus*), and beechnuts are only available during a limited time period. Future research should focus on additional potential limiting factors, such as the behavioral exclusion of *G. s. fuscus* by *G. volans* in the habitat surrounding red spruce (Weigl 1978) and impacts of climate change on forest configuration and patch extent of red spruce (White and Cogbill 1992). Furthermore, a greater understanding of dispersal behavior of *G. s. fuscus* may provide further insights regarding the energetic requirements of dispersing juveniles and whether dispersing juveniles forage for specific foods, or rely on energy stores while moving through the landscape (Zollner and Lima 2005).

Based on our results, management of the dispersal matrix for *G. s. fuscus* should consider prioritizing mature red spruce patches that may act as connections between larger areas of red spruce. These patches may provide hypogeous fungi truffles and lichen for dispersing *G. s. fuscus*, as well as other diet items identified through stable isotope analysis. However, regardless of forest type, mature stands typically had structural features and composition that afforded food resources, whereas younger stands did not. This suggests that managers should consider stand age and structure to a greater degree than forest type for management of habitat outside of red spruce stands for *G. s. fuscus*.

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Supplementary Data

Supplementary data are available at *Journal of Mammalogy* online.

Supplementary Data SD1.—The $\delta^{13}C$ and $\delta^{15}N$ signatures and C:N ratios for individual *Glaucomys sabrinus fuscus* sampled in the Monongahela National Forest, West Virginia, United States, in spring and fall 2014–2015. The Canaan Loop Road, Yellow Birch Trail, Red Run, and River Road samples were all collected using hair snares near Davis, West Virginia. The nest box samples were collected directly from individual squirrels during nest box checks in the northern region of the Monongahela National Forest. The Greenbriar samples were collected in the Greenbriar Ranger District approximately 50–60 km south of Davis, from nest box surveys.

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Transportation Infrastructure and Access on National Forests and Grasslands
A Literature Review
May 2014

Introduction
The Forest Service transportation system is very large with 374,883 miles (603,316 km) of system roads and 143,346 miles (230,693 km) of system trails. The system extends broadly across every national forest and grasslands and through a variety of habitats, ecosystems and terrains. An impressive body of scientific literature exists addressing the various effects of roads on the physical, biological and cultural environment – so much so, in the last few decades a new field of "road ecology" has emerged. In recent years, the scientific literature has expanded to address the effects of roads on climate change adaptation and conversely the effects of climate change on roads, as well as the effects of restoring lands occupied by roads on the physical, biological and cultural environments.

The following literature review summarizes the most recent thinking related to the environmental impacts of forest roads and motorized routes and ways to address them. The literature review is divided into three sections that address the environmental effects of transportation infrastructure on forests, climate change and infrastructure, and creating sustainable forest transportation systems.

I. Impacts of Transportation Infrastructure and Access to the Ecological Integrity of Terrestrial and Aquatic Ecosystems and Watersheds

It is well understood that transportation infrastructure and access management impact aquatic and terrestrial environments at multiple scales, and, in general, the more roads and motorized routes the greater the impact. In fact, in the past 20 years or so, scientists having realized the magnitude and breadth of ecological issues related to roads; entire books have been written on the topic, e.g., Forman et al. (2003), and a new scientific field called "road ecology" has emerged. Road ecology research centers have been created including the Western

Transportation Institute at Montana State University and the Road Ecology Center at the University of California - Davis.[1]

Below, we provide a summary of the current understanding on the impacts of roads and access allowed by road networks to terrestrial and aquatic ecosystems, drawing heavily on Gucinski et al. (2000). Other notable recent peer-reviewed literature reviews on roads include Trombulak and Frissell (2000), Switalski et al. (2004), Coffin (2007), Fahrig and Rytwinski (2009), and Robinson et al. (2010). Recent reviews on the impact of motorized recreation include Joslin and Youmans (1999), Gaines et al. (2003), Davenport and Switalski (2006), Ouren et al. (2007), and Switalski and Jones (2012). These peer-reviewed summaries provide additional information to help managers develop more sustainable transportation systems

Impact on geomorphology and hydrology

The construction or presence of forest roads can dramatically change the hydrology and geomorphology of a forest system leading to reductions in the quantity and quality of aquatic habitat. While there are several mechanisms that cause these impacts (Wemple et al. 2001 , Figure 1), most fundamentally, compacted roadbeds reduce rainfall infiltration, intercepting and concentrating water, and providing a ready source of sediment for transport (Wemple et al. 1996, Wemple et al. 2001). In fact, roads contribute more sediment to streams than any other land management activity (Gucinski et al. 2000). Surface erosion rates from roads are typically at least an order of magnitude greater than rates from harvested areas, and three orders of magnitude greater than erosion rates from undisturbed forest soils (Endicott 2008).

[1] See http://www.westerntransportationinstitute.org/research/roadecology and http://roadecology.ucdavis.edu/

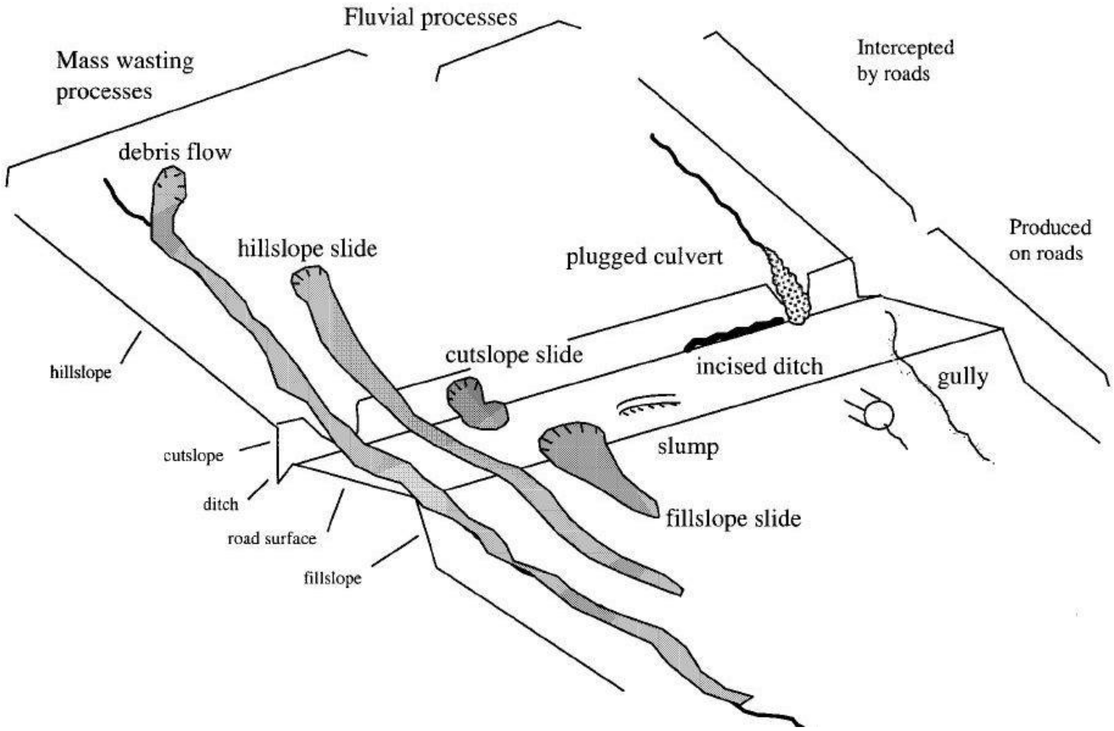

Figure 1: Typology of erosional and depositional features produced by mass-wasting and fluvial processes associate with forest roads (reprinted from Wemple et al. 2001)

Erosion of sediment from roads occurs both chronically and catastrophically. Every time it rains, sediment from the road surface and from cut- and fill-slopes is picked up by rainwater that flows into and on roads (fluvial erosion). The sediment that is entrained in surface flows are often concentrated into road ditches and culverts and directed into streams. The degree of fluvial erosion varies by geology and geography, and increases with increased motorized use (Robichaud et al. 2010). Closed roads produce less sediment, and Foltz et al. (2009) found a significant increase in erosion when closed roads were opened and driven upon.

Roads also precipitate catastrophic failures of road beds and fills (mass wasting) during large storm events leading to massive slugs of sediment moving into waterways (Endicott 2008; Gucinski et al. 2000). This typically occurs when culverts are undersized and cannot handle the volume of water, or they simply become plugged with debris. The saturated roadbed can fail entirely and result in a landslide, or the blocked stream crossing can erode the entire fill down to the original stream channel.

The erosion of road- and trail-related sediment and its subsequent movement into stream systems affects the geomorphology of the drainage system in a number of ways. The magnitude of their effects varies by climate, geology, road age, construction / maintenance practices and storm history. It directly alters channel morphology by embedding larger gravels as well as filling pools. It can also have the opposite effect of increasing peak discharges and scouring channels, which can lead to disconnection of the channel and floodplain, and lowered base flows (Furniss et al. 1991; Joslin and Youmans 1999). The width/depth ratio of the stream changes which then can trigger changes in water temperature, sinuosity and other geomorphic factors important for aquatic species survival (Joslin and Youmans 1999; Trombulak and Frissell 2000).

Roads also can modify flowpaths in the larger drainage network. Roads intercept subsurface flow as well as concentrate surface flow, which results in new flowpaths that otherwise would not exist, and the extension of the drainage network into previously unchannelized portions of the hillslope (Gucinski et al. 2000; Joslin and Youmans 1999). Severe aggradation of sediment at stream structures or confluences can force streams to actually go subsurface or make them too shallow for fish passage (Endicott 2008; Furniss et al. 1991).

Impacts on aquatic habitat and fish

Roads can have dramatic and lasting impacts on fish and aquatic habitat. Increased sedimentation in stream beds has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, and increased predation of fishes, and reductions in macro-invertebrate populations that are a food source to many fish species (Rhodes et al. 1994, Joslin and Youmans 1999, Gucinski et al. 2000, Endicott 2008). On a landscape scale, these effects can add up to: changes in the frequency, timing and magnitude of disturbance to aquatic habitat and changes to aquatic habitat structures (e.g., pools, riffles, spawning gravels and in-channel debris), and conditions (food sources, refugi, and water temperature) (Gucinski et al. 2000).

Roads can also act as barriers to migration (Gucinski et al. 2000). Where roads cross streams, road engineers usually place culverts or bridges. Culverts in particular can and often interfere with sediment transport and channel processes such that the road/stream crossing becomes a barrier for fish and aquatic species movement up and down stream. For instance, a culvert may scour on the downstream side of the crossing, actually forming a waterfall up which fish cannot move. Undersized culverts and bridges can infringe upon the channel or floodplain and trap sediment causing the stream to become too shallow and/or warm such that fish will not migrate past the structure. This is problematic for many aquatic species but especially for anadromous species that must migrate upstream to spawn. Well-known native aquatic species affected by roads include salmon such as coho (*Oncorhynchus kisutch*), chinook (*O. tshawytscha*), and chum (*O. keta*); steelhead (*O. mykiss*); and a variety of trout species including bull trout (*Salvelinus confluentus*) and cutthroat trout (*O. clarki*), as well as other native fishes and amphibians (Endicott 2008).

Impacts on terrestrial habitat and wildlife

Roads and trails impact wildlife through a number of mechanisms including: direct mortality (poaching, hunting/trapping) changes in movement and habitat use patterns (disturbance/avoidance), as well as indirect impacts including alteration of the adjacent habitat and interference with predatory/prey relationships (Wisdom et al. 2000, Trombulak and Frissell 2000). Some of these impacts result from the road itself, and some result from the uses on and around the roads (access). Ultimately, roads have been found to reduce the abundance and distribution of several forest species (Fayrig and Ritwinski 2009, Benítez-López et al. 2010).

Table 1: Road- and recreation trail-associated factors for wide-ranging carnivores (Reprinted from Gaines et al. (2003)[2]

[2] For a list of citations see Gaines et al. (2003)

Focal species	Road-associated factors	Motorized trail-associated factors	Nonmotorized trail-associated factors
Grizzly bear	Poaching Collisions Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance
Lynx	Down log reduction Trapping Collisions Disturbance at a specific site	Disturbance at a specific site Trapping	Disturbance at a specific site
Gray wolf	Trapping Poaching Collisions Negative human interactions Disturbance at a specific site Displacement or avoidance	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site
Wolverine	Down log reduction Trapping Disturbance at a specific site Collisions	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site

Direct mortality and disturbance from road and trail use impacts many different types of species. For example, wide-ranging carnivores can be significantly impacted by a number of factors including trapping, poaching, collisions, negative human interactions, disturbance and displacement (Gaines et al. 2003, Table 1). Hunted game species such as elk (*Cervus canadensis*), become more vulnerable from access allowed by roads and motorized trails resulting in a reduction in effective habitat among other impacts (Rowland et al. 2005, Switalski and Jones 2012). Slow-moving migratory animals such as amphibians, and reptiles who use roads to regulate temperature are also vulnerable (Gucinski et al. 2000, Brehme et al. 2013).

Habitat alteration is a significant consequence of roads as well. At the landscape scale, roads fragment habitat blocks into smaller patches that may not be able to support successfully interior forest species. Smaller habitat patches also results in diminished genetic variability, increased inbreeding, and at times local extinctions (Gucinski et al. 2000; Trombulak and Frissell 2000). Roads also change the composition and structure of ecosystems along buffer zones, called edge-affected zones. The width of edge-affected zones varies by what metric is being discussed; however, researchers have documented road-avoidance zones a kilometer or more away from a road (Table 2). In heavily roaded landscapes, edge-affected acres can be a significant fraction of total acres. For example, in a landscape area where the road density is 3 mi/mi^2 (not an uncommon road density in national forests) and where the edge-affected zone is estimated to be 500 ft from the center of the road to each side, the edge-affected zone is 56% of the total acreage.

Table 2: A summary of some documented road-avoidance zones for various species (adapted from Robinson et al. 2010).

Species	Avoidance zone m (ft)	Type of disturbance	Reference
Snakes	650 (2133)	Forestry roads	Bowles (1997)
Salamander	35 (115)	Narrow forestry road, light traffic	Semlitsch (2003)
Woodland birds	150 (492)	Unpaved roads	Ortega and Capen (2002)
Spotted owl	400 (1312)	Forestry roads, light traffic	Wasser et al. (1997)
Marten	<100 (<328)	Any forest opening	Hargis et al. (1999)
Elk	500–1000 (1640-3281)	Logging roads, light traffic	Edge and Marcum (1985)
	100–300 (328-984)	Mountain roads depending on traffic volume	Rost and Bailey (1979)
Grizzly bear	3000 (9840)	Fall	Mattson et al. (1996)
	500 (1640)	Spring and summer	
	883 (2897)	Heavily traveled trail	Kasworm and Manley (1990)
	274 (899)	Lightly traveled trail	
	1122 (3681)	Open road	Kasworm and Manley (1990)
	665 (2182)	Closed road	
Black bear	274 (899)	Spring, unpaved roads	Kasworm and Manley (1990)
	914 (2999)	Fall, unpaved roads	

Roads and trails also affect ecosystems and habitats because they are also a major vector of non-native plant and animal species. This can have significant ecological and economic impacts when the invading species are aggressive and can overwhelm or significantly alter native species and systems. In addition, roads can increase harassment, poaching and collisions with vehicles, all of which lead to stress or mortality (Wisdom et al. 2000).

Recent reviews have synthesized the impacts of roads on animal abundance and distribution. Fahrig and Rytwinski (2009) did a complete review of the empirical literature on effects of roads and traffic on animal abundance and distribution looking at 79 studies that addressed 131 species and 30 species groups. They found that the number of documented negative effects of roads on animal abundance outnumbered the number of positive effects by a factor of 5. Amphibians, reptiles, most birds tended to show negative effects. Small mammals generally showed either positive effects or no effect, mid-sized mammals showed either negative effects or no effect, and large mammals showed predominantly negative effects. Benítez-López et al. (2010) conducted a meta-analysis on the effects of roads and infrastructure proximity on mammal and bird populations. They found a significant pattern of avoidance and a reduction in bird and mammal populations in the vicinity of infrastructure.

Road density[3] thresholds for fish and wildlife

[3] We intend the term “road density” to refer to the density all roads within national forests, including system roads, closed roads, non-system roads administered by other jurisdictions (private, county, state), temporary roads and motorized trails. Please see Attachment 2 for the relevant existing scientific information supporting this approach.

It is well documented that beyond specific road density thresholds, certain species will be negatively affected, and some will be extirpated. Most studies that look into the relationship between road density and wildlife focus on the impacts to large endangered carnivores or hunted game species, although high road densities certainly affect other species – for instance, reptiles and amphibians. Gray wolves (*Canis lupus*) in the Great Lakes region and elk in Montana and Idaho have undergone the most long-term and in depth analysis. Forman and Hersperger (1996) found that in order to maintain a naturally functioning landscape with sustained populations of large mammals, road density must be below 0.6 km/km^2 (1.0 mi/mi^2). Several studies have since substantiated their claim (Robinson et al. 2010, Table 3).

A number of studies at broad scales have also shown that higher road densities generally lead to greater impacts to aquatic habitats and fish density (Table 3). Carnefix and Frissell (2009) provide a concise review of studies that correlate cold water fish abundance and road density, and from the cited evidence concluded that "1) no truly "safe" threshold road density exists, but rather negative impacts begin to accrue and be expressed with incursion of the very first road segment; and 2) highly significant impacts (e.g., threat of extirpation of sensitive species) are already apparent at road densities on the order of 0.6 km/km^2 (1.0 mi/mi^2) or less" (p. 1).

Table 3: A summary of some road-density thresholds and correlations for terrestrial and aquatic species and ecosystems (reprinted from Robinson et al. 2010).

Species (Location)	Road density (mean, guideline, threshold, correlation)	Reference
Wolf (Minnesota)	0.36 km/km2 (mean road density in primary range); 0.54 km/km^2 (mean road density in peripheral range)	Mech et al. (1988)
Wolf	>0.6 km/km^2 (absent at this density)	Jalkotzy et al. (1997)
Wolf (Northern Great Lakes region)	>0.45 km/km^2 (few packs exist above this threshold); >1.0 km/km^2 (no pack exist above this threshold)	Mladenoff et al. (1995)
Wolf (Wisconsin)	0.63 km/km^2 (increasing due to greater human tolerance	Wydeven et al. (2001)
Wolf, mountain lion (Minnesota, Wisconsin, Michigan)	0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained populations)	Thiel (1985); van Dyke et al. (1986); Jensen et al. (1986); Mech et al. (1988); Mech (1989)
Elk (Idaho)	1.9 km/km^2 (density standard for habitat effectiveness)	Woodley 2000 cited in Beazley et al. 2004
Elk (Northern US)	1.24 km/km^2 (habitat effectiveness decline by at least 50%)	Lyon (1983)
Elk, bear, wolverine, lynx, and others	0.63 km/km^2 (reduced habitat security and increased mortality)	Wisdom et al. (2000)
Moose (Ontario)	0.2-0.4 km/km2 (threshold for pronounced response)	Beyer et al. (2013)
Grizzly bear (Montana)	>0.6 km/km^2	Mace et al. (1996); Mattson et al. (1996)
Black bear (North Carolina)	>1.25 km/km^2 (open roads); >0.5 km/km2 (logging roads); (interference with use of habitat)	Brody and Pelton (1989)
Black bear	0.25 km/km^2 (road density should not exceed)	Jalkotzy et al. (1997)
Bobcat (Wisconsin)	1.5 km/km^2 (density of all road types in home range)	Jalkotzy et al. (1997)

Large mammals	>0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained populations)	Forman and Hersperger (1996)
Bull trout (Montana)	Inverse relationship of population and road density	Rieman et al. (1997); Baxter et al. (1999)
Fish populations (Medicine Bow National Forest)	(1) Positive correlation of numbers of culverts and stream crossings and amount of fine sediment in stream channels (2) Negative correlation of fish density and numbers of culverts	Eaglin and Hubert (1993) cited in Gucinski et al. (2001)
Macroinvertebrates	Species richness negatively correlated with an index of road density	McGurk and Fong (1995)
Non-anadromous salmonids (Upper Columbia River basin)	(1) Negative correlation likelihood of spawning and rearing and road density (2) Negative correlation of fish density and road density	Lee et al. (1997)

Where both stream and road densities are high, the incidence of connections between roads and streams can also be expected to be high, resulting in more common and pronounced effects of roads on streams (Gucinski et al. 2000). For example, a study on the Medicine Bow National Forest (WY) found as the number of culverts and stream crossings increased, so did the amount of sediment in stream channels (Eaglin and Hubert 1993). They also found a negative correlation with fish density and the number of culverts. Invertebrate communities can also be impacted. McGurk and Fong (1995) report a negative correlation between an index of road density with macroinvertebrate diversity.

The U.S. Fish and Wildlife Service's Final Rule listing bull trout as threatened (USDI Fish and Wildlife Service 1999) addressed road density, stating:

> "… assessment of the interior Columbia Basin ecosystem revealed that increasing road densities were associated with declines in four non-anadromous salmonid species (bull trout, Yellowstone cutthroat trout, westslope cutthroat trout, and redband trout) within the Columbia River Basin, likely through a variety of factors associated with roads (Quigley & Arbelbide 1997). Bull trout were less likely to use highly roaded basins for spawning and rearing, and if present, were likely to be at lower population levels (Quigley and Arbelbide 1997). Quigley et al. (1996) demonstrated that when average road densities were between 0.4 to 1.1 km/km^2 (0.7 and 1.7 mi/mi^2) on USFS lands, the proportion of subwatersheds supporting "strong" populations of key salmonids dropped substantially. Higher road densities were associated with further declines" (USDI Fish and Wildlife Service 1999, p. 58922).

Anderson et al. (2012) also showed that watershed conditions tend to be best in areas protected from road construction and development. Using the US Forest Service's Watershed Condition Framework assessment data, they showed that National Forest lands that are protected under the Wilderness Act, which provides the strongest safeguards, tend to have the healthiest watersheds. Watersheds in Inventoried Roadless Areas – which are protected from road building and logging by the Roadless Area Conservation Rule – tend to be less healthy than watersheds in designated Wilderness, but they are considerably healthier than watersheds in the managed landscape.

Impacts on other resources
Roads and motorized trails also play a role in affecting wildfire occurrence. Research shows that human-ignited wildfires, which account for more than 90% of fires on national lands, is almost five times more likely in areas with roads (USDA Forest Service 1996a; USDA Forest Service 1998). Furthermore, Baxter (2002) found that off-road vehicles (ORVs) can be a significant source of fire ignitions on forestlands. Roads can affect where and how forests burn and, by extension, the vegetative condition of the forest. See Attachment 1 for more information documenting the relationship between roads and wildfire occurrence.

Finally, access allowed by roads and trails can increase of ORV and motorized use in remote areas threatening archaeological and historic sites. Increased visitation has resulted in intentional and unintentional damage to many cultural sites (USDI Bureau of Land Management 2000, Schiffman 2005).

II. Climate Change and Transportation Infrastructure including the value of roadless areas for climate change adaptation

As climate change impacts grow more profound, forest managers must consider the impacts on the transportation system as well as from the transportation system. In terms of the former, changes in precipitation and hydrologic patterns will strain infrastructure at times to the breaking point resulting in damage to streams, fish habitat, and water quality as well as threats to public safety. In terms of the latter, the fragmenting effect of roads on habitat will impede the movement of species which is a fundamental element of adaptation. Through planning, forest managers can proactively address threats to infrastructure, and can actually enhance forest resilience by removing unneeded roads to create larger patches of connected habitat.

Impact of climate change and roads on transportation infrastructure
It is expected that climate change will be responsible for more extreme weather events, leading to increasing flood severity, more frequent landslides, changing hydrographs (peak, annual mean flows, etc.), and changes in erosion and sedimentation rates and delivery processes. Roads and trails in national forests, if designed by an engineering standard at all, were designed for storms and water flows typical of past decades, and hence may not be designed for the storms in future decades. Hence, climate driven changes may cause transportation infrastructure to malfunction or fail (ASHTO 2012, USDA Forest Service 2010). The likelihood is higher for facilities in high-risk settings—such as rain-on-snow zones, coastal areas, and landscapes with unstable geology (USDA Forest Service 2010).

Forests fragmented by roads will likely demonstrate less resistance and resilience to stressors, like those associated with climate change (Noss 2001). First, the more a forest is fragmented (and therefore the higher the edge/interior ratio), the more the forest loses its inertia characteristic, and becoming less resilient and resistant to climate change. Second, the more a forest is fragmented characterized by isolated patches, the more likely the fragmentation will interfere with the ability of species to track shifting climatic conditions over time and space. Noss (2001) predicts that weedy species with effective dispersal mechanisms might benefit from fragmentation at the expense of native species.

Modifying infrastructure to increase resilience
To prevent or reduce road failures, culvert blow-outs, and other associated hazards, forest managers will need to take a series of actions. These include replacing undersized culverts with larger ones, prioritizing maintenance and upgrades (e.g., installing drivable dips and more outflow structures), and obliterating roads that are no longer needed and pose erosion hazards (USDA Forest Service 2010, USDA Forest Service 2012a, USDA Forest Service 2011, Table 4).

Olympic National Forest has developed a number of documents oriented at oriented at protecting watershed health and species in the face of climate change, including a 2003 travel management strategy and a report entitled Adapting to Climate Change in Olympic National Park and National Forest. In the travel management strategy, Olympic National Forest recommended that 1/3rd of its road system be decommissioned and obliterated (USDA Forest Service 2011a). In addition, the plan called for addressing fish migration barriers in a prioritized and strategic way – most of these are associated with roads. The report calls for road decommissioning, relocation of roads away from streams, enlarging culverts as well as replacing culverts with fish-friendly crossings (USDA Forest Service 2011a, Table 4).

Table 4: Current and expected sensitivities of fish to climate change on the Olympic Peninsula, associated adaptation strategies and action for fisheries and fish habitat management and relevant to transportation management at Olympic National Forest and Olympic National Park (excerpt reprinted from USDA Forest Service 2011a).

Current and expected sensitivites	Adaptation strategies and actions
Changes in habitat quantity and quality	• Implement habitat restoration projects that focus on re-creating watershed processes and functions and that create diverse, resilient habitat.
Increase in culvert failures, fill-slope failures, stream adjacent road failures, and encroachment from stream-adjacent road segments	• Decommission unneeded roads. • Remove sidecast, improve drainage, and increase culvert sizing on remaining roads. • Relocate stream-adjacent roads.
Greater difficulty disconnecting roads from stream channels	• Design more resilient stream crossing structures.
Major changes in quantity and timing of streamflow in transitional watersheds	• Make road and culvert designs more conservative in transitional watersheds to accommodate expected changes.
Decrease in area of headwater streams	• Continue to correct culvert fish passage barriers. • Consider re-prioritizing culvert fish barrier correction projects.
Decrease in habitat quantity and connectivity for species that use headwater streams	• Restore habitat in degraded headwater streams that are expected to retain adequate summer streamflow (ONF).

In December 2012, the USDA Forest Service published a report entitled "Assessing the Vulnerability of Watersheds to Climate Change." This document reinforces the concept expressed by Olympic National Forest that forest managers need to be proactive in reducing erosion potential from roads:

> "Road improvements were identified as a key action to improve condition and resilience of watersheds on all the pilot Forests. In addition to treatments that reduce erosion, road improvements can reduce the delivery of runoff from road segments to channels, prevent diversion of flow during large events, and restore aquatic habitat connectivity by providing for passage of aquatic organisms. As stated previously, watershed sensitivity is determined by both inherent and management-related factors. Managers have no control over the inherent factors, so to improve resilience, efforts must be directed at anthropogenic influences such as instream flows, roads, rangeland, and vegetation management….
>
> [Watershed Vulnerability Analysis] results can also help guide implementation of travel management planning by informing priority setting for decommissioning roads and road reconstruction/maintenance. As with the Ouachita NF example, disconnecting roads from the stream network is a key objective of such work. Similarly, WVA analysis could also help prioritize aquatic organism passage projects at road-stream crossings to allow migration by aquatic residents to suitable habitat as streamflow and temperatures change" (USDA Forest Service 2012a, p. 22-23).

Reducing fragmentation to enhance aquatic and terrestrial species adaptation
Decommissioning and upgrading roads and thus reducing the amount of fine sediment deposited on salmonid nests can increase the likelihood of egg survival and spawning success (McCaffery et al. 2007). In addition, this would reconnect stream channels and remove barriers such as culverts. Decommissioning roads in riparian areas may provide further benefits to salmon and other aquatic organisms by permitting reestablishment of streamside vegetation, which provides shade and maintains a cooler, more moderated microclimate over the stream (Battin et al. 2007).

One of the most well documented impacts of climate change on wildlife is a shift in the ranges of species (Parmesan 2006). As animals migrate, landscape connectivity will be increasingly important (Holman et al. 2005). Decommissioning roads in key wildlife corridors will improve connectivity and be an important mitigation measure to increase resiliency of wildlife to climate change. For wildlife, road decommissioning can reduce the many stressors associated with roads. Road decommissioning restores habitat by providing security and food such as grasses and fruiting shrubs for wildlife (Switalski and Nelson 2011).

Forests fragmented by roads and motorized trail networks will likely demonstrate less resistance and resilience to stressors, such as weeds. As a forest is fragmented and there is more edge habitat, Noss (2001) predicts that weedy species with effective dispersal mechanisms will increasingly benefit at the expense of native species. However, decommissioned roads when seeded with native species can reduce the spread of invasive species (Grant et al. 2011), and help restore fragmented forestlands. Off-road vehicles with large knobby tires and large undercarriages are also a key vector for weed spread (e.g., Rooney 2006). Strategically closing and decommissioning motorized routes, especially in roadless areas, will reduce the spread of weeds on forestlands (Gelbard and Harrison 2003).

Transportation infrastructure and carbon sequestration
The topic of the relationship of road restoration and carbon has only recently been explored. There is the potential for large amounts of carbon (C) to be sequestered by reclaiming roads. When roads are decompacted during reclamation, vegetation and soils can develop more

rapidly and sequester large amounts of carbon. A recent study estimated total soil C storage increased 6 fold to 6.5 x 107g C/km (to 25 cm depth) in the northwestern US compared to untreated abandoned roads (Lloyd et al. 2013). Another recent study concluded that reclaiming 425 km of logging roads over the last 30 years in Redwood National Park in Northern California resulted in net carbon savings of 49,000 Mg carbon to date (Madej et al. 2013, Table 5).

Kerekvliet et al. (2008) published a Wilderness Society briefing memo on the impact to carbon sequestration from road decommissioning. Using Forest Service estimates of the fraction of road miles that are unneeded, the authors calculated that restoring 126,000 miles of roads to a natural state would be equivalent to revegetating an area larger than Rhode Island. In addition, they calculate that the net economic benefit of road treatments are always positive and range from US$0.925-1.444 billion.

Table 5. Carbon budget implications in road decommissioning projects (reprinted from Madej et al. 2013).

Road Decommissioning Activities and Processes	Carbon Cost	Carbon Savings
Transportation of staff to restoration sites (fuel emissions)	X	
Use of heavy equipment in excavations (fuel emissions)	X	
Cutting trees along road alignment during hillslope recontouring	X	
Excavation of road fill from stream crossings		X
Removal of road fill from unstable locations		X
Reduces risk of mass movement		X
Post-restoration channel erosion at excavation sites	X	
Natural revegetation following road decompaction		X
Replanting trees		X
Soil development following decompaction		X

Benefits of roadless areas and roadless area networks to climate change adaptation
Undeveloped natural lands provide numerous ecological benefits. They contribute to biodiversity, enhance ecosystem representation, and facilitate connectivity (Loucks et al. 2003; Crist and Wilmer 2002, Wilcove 1990, The Wilderness Society 2004, Strittholt and Dellasala 2001, DeVelice and Martin 2001), and provide high quality or undisturbed water, soil and air (Anderson et al. 2012, Dellasalla et al. 2011). They also can serve as ecological baselines to help us better understand our impacts to other landscapes, and contribute to landscape resilience to climate change.

Forest Service roadless lands, in particular, are heralded for the conservation values they provide. These are described at length in the preamble of the Roadless Area Conservation Rule (RACR)[4] as well as in the Final Environmental Impact Statement (FEIS) for the RACR[5], and

[4] Federal Register .Vol. 66, No. 9. January 12, 2001. Pages 3245-3247.

include: high quality or undisturbed soil, water, and air; sources of public drinking water; diversity of plant and animal communities; habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land; primitive, semi-primitive non- motorized, and semi-primitive motorized classes of dispersed recreation; reference landscapes; natural appearing landscapes with high scenic quality; traditional cultural properties and sacred sites; and other locally identified unique characteristics (e.g., include uncommon geological formations, unique wetland complexes, exceptional hunting and fishing opportunities).

The Forest Service, National Park Service, and US Fish and Wildlife Service recognize that protecting and connecting roadless or lightly roaded areas is an important action agencies can take to enhance climate change adaptation. For example, the Forest Service National Roadmap for Responding to Climate Change (USDA Forest Service 2011b) establishes that increasing connectivity and reducing fragmentation are short and long term actions the Forest Service should take to facilitate adaptation to climate change.[6] The National Park Service also identifies connectivity as a key factor for climate change adaptation along with establishing "blocks of natural landscape large enough to be resilient to large-scale disturbances and long-term changes" and other factors. The agency states that: "The success of adaptation strategies will be enhanced by taking a broad approach that identifies connections and barriers across the landscape. Networks of protected areas within a larger mixed landscape can provide the highest level of resilience to climate change."[7] Similarly, the National Fish, Wildlife and Plants Climate Adaptation Partnership's Adaptation Strategy (2012) calls for creating an ecologically-connected network of conservation areas.[8]

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[5] Final Environmental Impact Statement, Vol. 1, 3–3 to 3–7

[6] Forest Service, 2011. *National Roadmap for Responding to Climate Change*. US Department of Agriculture. FS-957b. Page 26.

[7] National Park Service. *Climate Change Response Program Brief.* http://www.nature.nps.gov/climatechange/adaptationplanning.cfm. Also see: National Park Service, 2010. *Climate Change Response Strategy*. http://www.nature.nps.gov/climatechange/docs/NPS_CCRS.pdf. Objective 6.3 is to "Collaborate to develop cross-jurisdictional conservation plans to protect and restore connectivity and other landscape-scale components of resilience."

[8] See http://www.wildlifeadaptationstrategy.gov/pdf/NFWPCAS-Chapter-3.pdf. Pages 55- 59. The first goal and related strategies are:

> Goal 1: Conserve habitat to support healthy fish, wildlife, and plant populations and ecosystem functions in a changing climate.
>
> > Strategy 1.1: identify areas for an ecologically-connected network of terrestrial, freshwater, coastal, and marine conservation areas that are likely to be resilient to climate change and to support a broad range of fish, wildlife, and plants under changed conditions.
> >
> > Strategy 1.2: Secure appropriate conservation status on areas identified in Strategy 1.1 to complete an ecologically-connected network of public and private conservation areas that will be resilient to climate change and support a broad range of species under changed conditions.
> >
> > Strategy 1.4: Conserve, restore, and as appropriate and practicable, establish new ecological connections among conservation areas to facilitate fish, wildlife, and plant migration, range shifts, and other transitions caused by climate change.

Crist and Wilmer (2002) looked at the ecological value of roadless lands in the Northern Rockies and found that protection of national forest roadless areas, when added to existing federal conservation lands in the study area, would 1) increase the representation of virtually all land cover types on conservation lands at both the regional and ecosystem scales, some by more than 100%; 2) help protect rare, species-rich, and often-declining vegetation communities; and 3) connect conservation units to create bigger and more cohesive habitat "patches."

Roadless lands also are responsible for higher quality water and watersheds. Anderson et al. (2012) assessed the relationship of watershed condition and land management status and found a strong spatial association between watershed health and protective designations. Dellasalla et al. (2011) found that undeveloped and roadless watersheds are important for supplying downstream users with high-quality drinking water, and developing these watersheds comes at significant costs associated with declining water quality and availability. The authors recommend a light-touch ecological footprint to sustain the many values that derive from roadless areas including healthy watersheds.

III. Sustainable Transportation Management in National Forests as Part of Ecological Restoration

At 375,000 miles strong, the Forest Service road system is one of the largest in the world – it is eight times the size of the National Highway System. It is also indisputably unsustainable – that is, roads are not designed, located, or maintained according to best management practices, and environmental impacts are not minimized. It is largely recognized that forest roads, especially unpaved ones, are a primary source of sediment pollution to surface waters (Endicott 2008, Gucinski et al. 2000), and that the system has about 1/3rd more miles than it needs (USDA Forest Service 2001). In addition, the majority of the roads were constructed decades ago when road design and management techniques did not meet current standards (Gucinski et al. 2000, Endicott 2008), making them more vulnerable to erosion and decay than if they had been designed today. Road densities in national forests often exceed accepted thresholds for wildlife.

Only a small portion of the road system is regularly used. All but 18% of the road system is inaccessible to passenger vehicles. Fifty-five percent of the roads are accessible only by high clearance vehicles and 27% are closed. The 18% that is accessible to cars is used for about 80% of the trips made within National Forests.[9] Most of the road maintenance funding is directed to the passenger car roads, while the remaining roads suffer from neglect. As a result, the Forest Service currently has a $3.7 billion road maintenance backlog that grows every year. In other words, only about 1/5th of the roads in the national forest system are used most of the time, and the fraction that is used often is the best designed and maintained because they are higher level access roads. The remaining roads sit generally unneeded and under-maintained – arguably a growing ecological and fiscal liability.

Current Forest Service management direction is to identify and implement a sustainable transportation system.[10] The challenge for forest managers is figuring out what is a sustainable road system and how to achieve it – a challenge that is exacerbated by climate change. It is

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[9] USDA Forest Service. Road Management Website Q&As. Available online at http://www.fs.fed.us/eng/road_mgt/qanda.shtml.

[10] See Forest Service directive memo dated March 29, 2012 entitled "Travel Management, Implementation of 36 CFR, Part 202, Subpart A (36 CFR 212.5(b))"

reasonable to define a sustainable transportation system as one where all the routes are constructed, located, and maintained with best management practices, and social and environmental impacts are minimized. This, of course, is easier said than done, since the reality is that even the best roads and trail networks can be problematic simply because they exist and usher in land uses that without the access would not occur (Trombulak and Frissell 2000, Carnefix and Frissell 2009, USDA Forest Service 1996b), and when they are not maintained to the designed level they result in environmental problems (Endicott 2008; Gucinski et al. 2000). Moreover, what was sustainable may no longer be sustainable under climate change since roads designed to meet older climate criteria may no longer hold up under new climate scenarios (USDA Forest Service 2010, USDA Forest Service 2011b, USDA Forest Service 2012a, AASHTO 2012).

Forest Service efforts to move toward a more sustainable transportation system
The Forest Service has made efforts to make its transportation system more sustainable, but still has considerable work to do. In 2001, the Forest Service tried to address the issue by promulgating the Roads Rule[11] with the purpose of working toward a sustainable road system (USDA 2001). The Rule directed every national forest to identify a minimum necessary road system and identify unneeded roads for decommissioning. To do this, the Forest Service developed the Roads Analysis Process (RAP), and published Gucinski et al. (2000) to provide the scientific foundation to complement the RAP. In describing the RAP, Gucinski et al. (2000) writes:

> "Roads Analysis 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. Roads Analysis is to be 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" (p. 10).

Most national forests have completed RAPs, although most only looked at passenger vehicle roads which account for less than 20% of the system's miles. The Forest Service Washington Office in 2010 directed that forests complete a Travel Analysis Process (TAP) by the end of fiscal year 2015, which must address all roads and create a map and list of roads identifying which are likely needed and which are not. Completed TAPs will provide a blueprint for future road decommissioning and management, they will not constitute compliance with the Roads Rule, which clearly requires the identification of the minimum roads system and roads for decommissioning. Almost all forests have yet to comply with subpart A.

The Forest Service in 2005 then tried to address the off-road portion of this issue by promulgating subpart B of the Travel Managemenr Rule,[12] with the purpose of curbing the most serious impacts associated with off-road vehicle use. Without a doubt, securing summer-time travel management plans was an important step to curbing the worst damage. However, much work remains to be done to approach sustainability, especially since many national forests used the travel management planning process to simply freeze the footprint of motorized routes, and did not try to re-design the system to make it more ecologically or socially sustainable. Adams

[11] 36 CFR 215 subpart A
[12] 36 CFR 212 subpart B

and McCool (2009) considered this question of how to achieve sustainable motorized recreation and concluded that:

> As the agencies move to revise [off-road vehicle] allocations, they need to clearly define how they intend to locate routes so as to minimize impacts to natural resources and other recreationists in accordance with Executive Order 11644....[13]
>
> …As they proceed with designation, the FS and BLM need to acknowledge that current allocations are the product of agency failure to act, not design. Ideally, ORV routes would be allocated as if the map were currently empty of ORV routes. Reliance on the current baseline will encourage inefficient allocations that likely disproportionately impact natural resources and non-motorized recreationists. While acknowledging existing use, the agencies need to do their best to imagine the best possible arrangement of ORV routes, rather than simply tinkering around the edges of the current allocations.[14]

The Forest Service only now is contemplating addressing the winter portion of the issue, forced by a lawsuit challenging the Forest Service's inadequate management of snowmobiles. The agency is expected to issue a third rule in the fall of 2014 that will trigger winter travel management planning.

Strategies for identifying a minimum road system and prioritizing restoration
Transportation Management plays an integral role in the restoration of Forestlands. Reclaiming and obliterating roads is key to developing a sustainable transportation system. Numerous authors have suggested removing roads 1) to restore water quality and aquatic habitats Gucinski et al. 2000), and 2) to improve habitat security and restore terrestrial habitat (e.g., USDI USFWS 1993, Hebblewhite et al. 2009).

Creating a minimum road system through road removal will increase connectivity and decrease fragmentation across the entire forest system. However, at a landscape scale, certain roads and road segments pose greater risks to terrestrial and aquatic integrity than others. Hence, restoration strategies must focus on identifying and removing/mitigating the higher risk roads. Additionally, areas with the highest ecological values, such as being adjacent to a roadless area, may also be prioritized for restoration efforts. Several methods have been developed to help prioritize road reclamation efforts including GIS-based tools and best management practices (BMPs). It is our hope that even with limited resources, restoration efforts can be prioritized and a more sustainable transportation system created.

GIS-based tools

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[13] Recent court decisions have made it clear that the minimization requirements in the Executive Orders are not discretionary and that the Executive Orders are enforceable. See

- *Idaho Conservation League v. Guzman ,* 766 F. Supp. 2d 1056 (D. Idaho 2011) (Salmon-Challis National Forest TMP) .
- *The Wilderness Society v. U.S. Forest Service*, CV 08-363 (D. Idaho 2012) (Sawtooth-Minidoka district National Forest TMP).
- *Central Sierra Environmental Resource Center v. US Forest Service*, CV 10-2172 (E.D. CA 2012) (Stanislaus National Forest TMP).

[14] Page 105.

Girvetz and Shilling (2003) developed a novel and inexpensive way to analyze environmental impacts from road systems using the Ecosystem Management Decision Support program (EMDS). EMDS was originally developed by the United States Forest Service, as a GIS-based decision support tool to conduct ecological analysis and planning (Reynolds 1999). Working in conjunction with Tahoe National Forest managers, Girvetz and Shilling (2003) used spatial data on a number of aquatic and terrestrial variables and modeled the impact of the forest's road network. The network analysis showed that out of 8233 km of road analyzed, only 3483 km (42%) was needed to ensure current and future access to key points. They found that the modified network had improved patch characteristics, such as significantly fewer "cherry stem" roads intruding into patches, and larger roadlessness.

Shilling et al. (2012) later developed a recreational route optimization model using a similar methodology and with the goal of identifying a sustainable motorized transportation system for the Tahoe National Forest (Figure 2). Again using a variety of environmental factors, the model identified routes with high recreational benefits, lower conflict, lower maintenance and management requirements, and lower potential for environmental impact operating under the presumption that such routes would be more sustainable and preferable in the long term. The authors combined the impact and benefit analyses into a recreation system analysis "that was effectively a cost-benefit accounting, consistent with requirements of both the federal Travel Management Rule (TMR) and the National Environmental Policy Act" (p. 392**).**

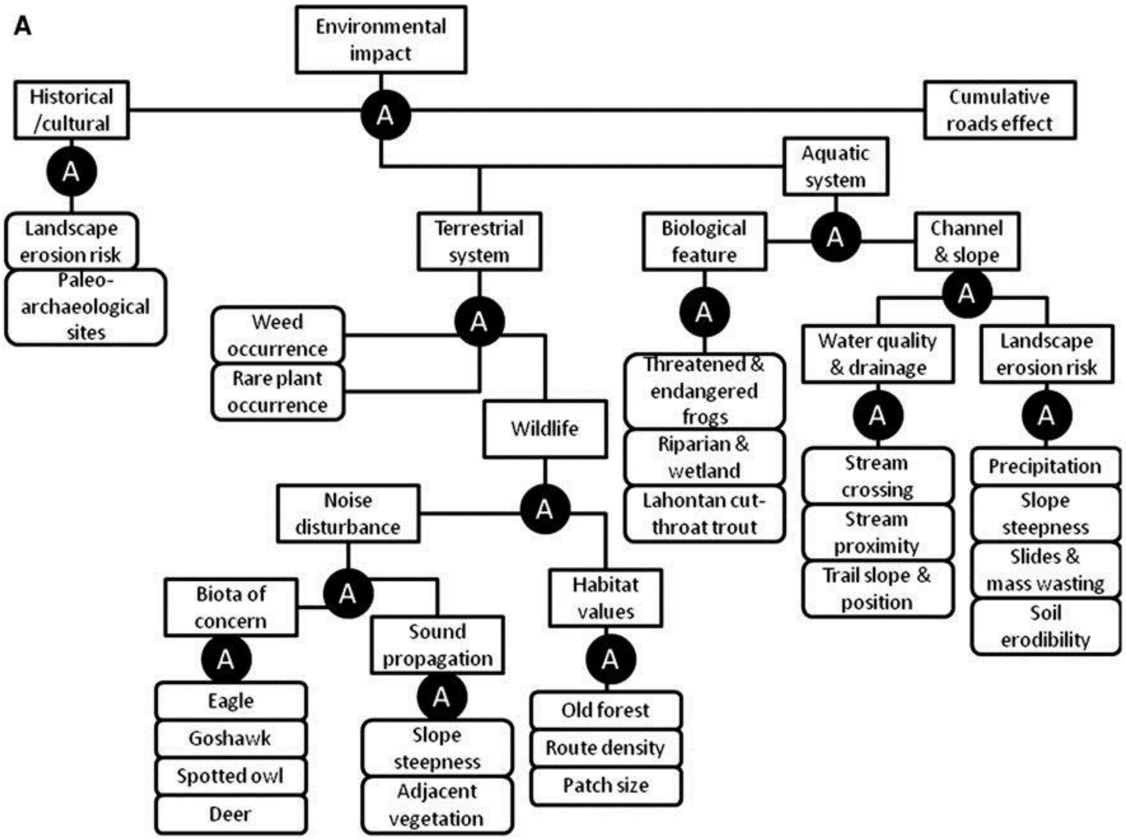

Figure 2: A knowledge base of contributions of various environmental conditions to the concept ‘‘environmental impact’’ [of motorized trails]. Rectangles indicate concepts, circles indicate Boolean logic operators, and rounded rectangles indicate sources of environmental data. (Reprinted from Shilling et al. 2012)

The Wilderness Society in 2012 also developed a GIS decision support tool called "RoadRight" that identifies high risk road segments to a variety of forest resources including water, wildlife, and roadlessness (The Wilderness Society 2012, The Wilderness Society 2013). The GIS system is designed to provide information that will help forest planners identify and minimize road related environmental risks. See the summary of and user guide for RoadRight that provides more information including where to access the open source software.[15]

[15] The Wilderness Society, 2012. Rightsizing the National Forest Road System: A Decision Support Tool. Available at http://www.landscapecollaborative.org/download/attachments/12747016/Road+decommissioning+model+-overview+2012-02-29.pdf?version=1&modificationDate=1331595972330.

The Wilderness Society, 2013.
RoadRight: A Spatial Decision Support System to Prioritize Decommissioning and Repairing Roads in

Best management practices (BMPs)
BMPs have also been developed to help create more sustainable transportation systems and identify restoration opportunities. BMPs provide science-based criteria and standards that land managers follow in making and implementing decisions about human uses and projects that affect natural resources. Several states have developed BMPs for road construction, maintenance and decommissioning practices (e.g., Logan 2001, Merrill and Cassaday 2003, USDA Forest Service 2012b).

Recently, BMPs have been developed for addressing motorized recreation. Switalski and Jones (2012) published, "*Off-Road Vehicle Best Management Practices for Forestlands: A Review of Scientific Literature and Guidance for Managers*." This document reviews the current literature on the environmental and social impacts of off-road vehicles (ORVs), and establishes a set of Best Management Practices (BMPs) for the planning and management of ORV routes on forestlands. The BMPs were designed to be used by land managers on all forestlands, and is consistent with current forest management policy and regulations. They give guidance to transportation planners on where how to place ORV routes in areas where they will reduce use conflicts and cause as little harm to the environment as possible. These BMPs also help guide managers on how to best remove and restore routes that are redundant or where there is an unacceptable environmental or social cost.

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Attachments

Attachment 1: Wildfire and Roads Fact Sheet

Attachment 2: Using Road Density as a Metric for Ecological Health in National Forests: What Roads and Routes should be Included? Summary of Scientific Information

Roads and Fire: A Proven Relationship

Photo: Lou Anegli Digital

Roaded Forests Are at a Greater Risk of Experiencing Wildfires than Unroaded Forests

- A wildland fire ignition is almost twice as likely to occur in a roaded area than in a roadless area. (USDA 2000, Table 3-18)
- The location of large wildfires is often correlated with proximity to busy roads. (Sierra Nevada Ecosystem Project, 1996)
- High road density increases the probability of fire occurrence due to human-caused ignitions. (Hann, W.J., et al. 1997)
- Unroaded areas have lower potential for high-intensity fires than roaded areas because they are less prone to human-caused ignitions. (DellaSala, et al. 1995)
- The median size of large fires on national forests is greater outside of roadless areas. (USDA 2000, Table 3-22)
- A positive correlation exists between lightning fire frequency and road density due to increased availability of flammable fine fuels near roads. (Arienti, M. Cecilia, et al. 2009)
- Human caused wildfires are strongly associated with access to natural landscapes, with the proximity to urban areas and roads being the most important factor (Romero-Calcerrada, et al. 2008)

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Human Activity and Wildfire

- Sparks from cars, off-road vehicles, and neglected campfires caused nearly 50,000 wildfire ignitions in 2000. (USDA 2000, Fuel Management and Fire Suppression Specialist Report, Table 4.)
- More than 90% of fires on national lands are caused by humans (USDA 1996 and 1998)
- Human-ignited wildfire is almost 5 times more likely to occur in a roaded area than in a roadless area (USDA 2000, Table 3-19).

There are 375,000 miles of roads in our national forests.

The Wilderness Society
1615 M St. N.W.
Washington, D.C. 20036
(202) 833-2300
wilderness.org

Roads and Fire: A Proven Relationship

Exhibit XII.1-30

Photo: USDA Forest Service, Coconino National Forest

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Attachment 2: Using Road Density as a Metric for Ecological Health in National Forests: What Roads and Routes should be Included?
Summary of Scientific Information
Last Updated, November 22, 2012

I. Density analysis should include closed roads, non-system roads administered by other jurisdictions (private, county, state), temporary roads and motorized trails.

Typically, the Forest Service has calculated road density by looking only at open system road density. From an ecological standpoint, this approach may be flawed since it leaves out of the density calculations a significant percent of the total motorized routes on the landscape. For instance, the motorized route system in the entire National Forest System measures well over 549,000 miles.[1] By our calculation, a density analysis limited to open system roads would consider less than 260,000 miles of road, which accounts for less than half of the entire motorized transportation system estimated to exist on our national forests.[2] These additional roads and motorized trails impact fish, wildlife, and water quality, just as open system roads do. In this section, we provide justification for why a road density analysis used for the purposes of assessing ecological health and the effects of proposed alternatives in a planning document should include closed system roads, non-system roads administered by other jurisdictions, temporary roads, and motorized trails.

Impacts of closed roads

It is crucial to distinguish the density of roads physically present on the landscape, whether closed to vehicle use or not, from "open-road density" (Pacific Rivers Council, 2010). An open-road density of 1.5 mi/mi² has been established as a standard in some national forests as protective of some terrestrial wildlife species. However, many areas with an open road density of 1.5 mi/mi² have a much higher inventoried or extant hydrologically effective road density, which may be several-fold as high with significant aquatic impacts. This higher density occurs because many road "closures" block vehicle access, but do nothing to mitigate the hydrologic alterations that the road causes. The problem is

[1] The National Forest System has about 372,000 miles of system roads. The forest service also has an estimated 47,000 miles of motorized trails. As of 1998, there were approximately 130,000 miles of non-system roads in our forests. Non-system roads include public roads such as state, county, and local jurisdiction and private roads. (USFS, 1998) The Forest Service does not track temporary roads but is reasonable to assume that there are likely several thousand miles located on National Forest System lands.

[2] About 30% of system roads, or 116,108 miles, are in Maintenance Level 1 status, meaning they are closed to all motorized use. (372,000 miles of NFS roads - 116,108 miles of ML 1 roads = 255,892). This number is likely conservative given that thousands of more miles of system roads are closed to public motorized use but categorized in other Maintenance Levels.

further compounded in many places by the existence of "ghost" roads that are not captured in agency inventories, but that are nevertheless physically present and causing hydrologic alteration (Pacific Watershed Associates, 2005).

Closing a road to public motorized use can mitigate the impacts on water, wildlife, and soils only if proper closure and storage technique is followed. Flow diversions, sediment runoff, and illegal incursions will continue unabated if necessary measures are not taken. The Forest Service's National Best Management Practices for non-point source pollution recommends the following management techniques for minimizing the aquatic impacts from closed system roads: eliminate flow diversion onto the road surface, reshape the channel and streambanks at the crossing-site to pass expected flows without scouring or ponding, maintain continuation of channel dimensions and longitudinal profile through the crossing site, and remove culverts, fill material, and other structures that present a risk of failure or diversion. Despite good intentions, it is unlikely given our current fiscal situation and past history that the Forest Service is able to apply best management practices to all stored roads,[3] and that these roads continue to have impacts. This reality argues for assuming that roads closed to the public continue to have some level of impact on water quality, and therefore, should be included in road density calculations.

As noted above, many species benefit when roads are closed to public use. However, the fact remains that closed system roads are often breached resulting in impacts to wildlife. Research shows that a significant portion of off-road vehicle (ORV) users violates rules even when they know what they are (Lewis, M.S., and R. Paige, 2006; Frueh, LM, 2001; Fischer, A.L., et. al, 2002; USFWS, 2007.). For instance, the Rio Grande National Forest's Roads Analysis Report notes that a common travel management violation occurs when people drive around road closures on Level 1 roads (USDA Forest Service, 1994). Similarly, in a recent legal decision from the Utah District Court , *Sierra Club v. USFS*, Case No. 1:09-cv-131 CW (D. Utah March 7, 2012), the court found that, as part of analyzing alternatives in a proposed travel management plan, the Forest Service failed to take a hard look at the impact of continued illegal use. In part, the court based its decision on the Forest Service's acknowledgement that illegal motorized use is a significant problem and that the mere presence of roads is likely to result in illegal use.

In addition to the disturbance to wildlife from ORVs, incursions and the accompanying human access can also result in illegal hunting and trapping of animals. The Tongass National Forest refers to this in its EIS to amend the Land and Resources Management Plan. Specifically, the Forest Service notes in the EIS that Alexander Archipelego wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi^2 or less may be necessary (USDA Forest Service, 2008).

As described below, a number of scientific studies have found that ORV use on roads and trails can have serious impacts on water, soil and wildlife resources. It should be expected that ORV use will continue to

[3] The Forest Service generally reports that it can maintain 20-30% of its open road system to standard.

some degree to occur illegally on closed routes and that this use will affect forest resources. Given this, roads closed to the general public should be considered in the density analysis.

Impacts of non-system roads administered by other jurisdictions (private, county, state)

As of 1998, there were approximately 130,000 miles of non-system roads in national forests (USDA Forest Service, 1998). These roads contribute to the environmental impacts of the transportation system on forest resources, just as forest system roads do. Because the purpose of a road density analysis is to measure the impacts of roads at a landscape level, the Forest Service should include all roads, including non-system, when measuring impacts on water and wildlife. An all-inclusive analysis will provide a more accurate representation of the environmental impacts of the road network within the analysis area.

Impacts of temporary roads

Temporary roads are not considered system roads. Most often they are constructed in conjunction with timber sales. Temporary roads have the same types environmental impacts as system roads, although at times the impacts can be worse if the road persists on the landscape because they are not built to last.

It is important to note that although they are termed temporary roads, their impacts are not temporary. According to Forest Service Manual (FSM) 7703.1, the agency is required to "Reestablish vegetative cover on any unnecessary roadway or area disturbed by road construction on National Forest System lands within 10 years after the termination of the activity that required its use and construction." Regardless of the FSM 10-year rule, temporary roads can remain for much longer. For example, timber sales typically last 3-5 years or more. If a temporary road is built in the first year of a six year timber sale, its intended use does not end until the sale is complete. The timber contract often requires the purchaser to close and obliterate the road a few years after the Forest Service completes revegetation work. The temporary road, therefore, could remain open 8-9 years before the ten year clock starts ticking per the FSM. Therefore, temporary roads can legally remain on the ground for up to 20 years or more, yet they are constructed with less environmental safeguards than modern system roads.

Impacts of motorized trails

Scientific research and agency publications generally do not decipher between the impacts from motorized trails and roads, often collapsing the assessment of impacts from unmanaged ORV use with those of the designated system of roads and trails. The following section summarizes potential impacts resulting from roads and motorized trails and the ORV use that occurs on them.

Aquatic Resources
While driving on roads has long been identified as a major contributor to stream sedimentation (for review, see Gucinski, 2001), recent studies have identified ORV routes as a significant cause of stream sedimentation as well (Sack and da Luz, 2004; Chin et al.; 2004, Ayala et al.; 2005, Welsh et al;. 2006). It has been demonstrated that sediment loss increases with increased ORV traffic (Foltz, 2006). A study by

Sack and da Luz (2004) found that ORV use resulted in a loss of more than 200 pounds of soil off of every 100 feet of trail each year. Another study (Welsh et al., 2006) found that ORV trails produced five times more sediment than unpaved roads. Chin et al. (2004) found that watersheds with ORV use as opposed to those without exhibited higher percentages of channel sands and fines, lower depths, and lower volume – all characteristics of degraded stream habitat.

Soil Resources [4]

Ouren, et al. (2007), in an extensive literature review, suggests ORV use causes soil compaction and accelerated erosion rates, and may cause compaction with very few passes. Weighing several hundred pounds, ORVs can compress and compact soil (Nakata et al., 1976; Snyder et al., 1976; Vollmer et al., 1976; Wilshire and Nakata, 1976), reducing its ability to absorb and retain water (Dregne, 1983), and decreasing soil fertility by harming the microscopic organisms that would otherwise break down the soil and produce nutrients important for plant growth (Wilshire et al., 1977). An increase in compaction decreases soil permeability, resulting in increased flow of water across the ground and reduced absorption of water into the soil. This increase in surface flow concentrates water and increases erosion of soils (Wilshire, 1980; Webb, 1983; Misak et al., 2002).

Erosion of soil is accelerated in ORV-use areas directly by the vehicles, and indirectly by increased runoff of precipitation and the creation of conditions favorable to wind erosion (Wilshire, 1980). Knobby and cup-shaped protrusions from ORV tires that aid the vehicles in traversing steep slopes are responsible for major direct erosional losses of soil. As the tire protrusions dig into the soil, forces far exceeding the strength of the soil are exerted to allow the vehicles to climb slopes. The result is that the soil and small plants are thrown downslope in a "rooster tail" behind the vehicle. This is known as mechanical erosion, which on steep slopes (about 15° or more) with soft soils may erode as much as 40 tons/mi (Wilshire, 1992). The rates of erosion measured on ORV trails on moderate slopes exceed natural rates by factors of 10 to 20 (Iverson et al., 1981; Hinckley et al., 1983), whereas use on steep slopes has commonly removed the entire soil mantle exposing bedrock. Measured erosional losses in high use ORV areas range from 1.4-242 lbs/ft^2 (Wilshire et al., 1978) and 102-614 lbs/ft^2 (Webb et al., 1978). A more recent study by Sack and da Luz (2003) found that ORV use resulted in a loss of more than 200 lbs of soil off of every 100 feet of trail each year.

Furthermore, the destruction of cryptobiotic soils by ORVs can reduce nitrogen fixation by cyanobacteria, and set the nitrogen economy of nitrogen-limited arid ecosystems back decades. Even small reductions in crust can lead to diminished productivity and health of the associated plant community, with cascading effects on plant consumers (Davidson et al., 1996). In general, the deleterious effects of ORV use on cryptobiotic crusts is not easily repaired or regenerated. The recovery time for the lichen component of crusts has been estimated at about 45 years (Belnap, 1993). After this time the crusts may appear to have regenerated to the untrained eye. However, careful observation will reveal that the 45 year-old crusts will not have recovered their moss component, which will take an additional 200 years to fully come back (Belnap and Gillette, 1997).

[4] For a full review see Switalski, T. A. and A. Jones (2012).

Wildlife Resources [5]
Studies have shown a variety of possible wildlife disturbance vectors from ORVs. While these impacts are difficult to measure, repeated harassment of wildlife can result in increased energy expenditure and reduced reproduction. Noise and disturbance from ORVs can result in a range of impacts including increased stress (Nash et al., 1970; Millspaugh et al., 2001), loss of hearing (Brattstrom and Bondello, 1979), altered movement patterns (e.g., Wisdom et al. 2004; Preisler et al. 2006), avoidance of high-use areas or routes (Janis and Clark 2002; Wisdom 2007), and disrupted nesting activities (e.g., Strauss 1990).

Wisdom et al. (2004) found that elk moved when ORVs passed within 2,000 yards but tolerated hikers within 500 ft. Wisdom (2007) reported preliminary results suggesting that ORVs are causing a shift in the spatial distribution of elk that could increase energy expenditures and decrease foraging opportunities for the herd. Elk have been found to readily avoid and be displaced from roaded areas (Irwin and Peek, 1979; Hershey and Leege, 1982; Millspaugh, 1995). Additional concomitant effects can occur, such as major declines in survival of elk calves due to repeated displacement of elk during the calving season (Phillips, 1998). Alternatively, closing or decommissioning roads has been found to decrease elk disturbance (Millspaugh et al., 2000; Rowland et al., 2005).

Disruption of breeding and nesting birds is particularly well-documented. Several species are sensitive to human disturbance with the potential disruption of courtship activities, over-exposure of eggs or young birds to weather, and premature fledging of juveniles (Hamann et al., 1999). Repeated disturbance can eventually lead to nest abandonment. These short-term disturbances can lead to long-term bird community changes (Anderson et al., 1990). However when road densities decrease, there is an observable benefit. For example, on the Loa Ranger District of the Fishlake National Forest in southern Utah, successful goshawk nests occur in areas where the localized road density is at or below 2-3 mi/mi^2 (USDA, 2005).

Examples of Forest Service planning documents that use total motorized route density or a variant

Below, we offer examples of where total motorized route density or a variant has been used by the Forest Service in planning documents.

- The Mt. Taylor RD of the Cibola NF analyzed open and closed system roads and motorized trails together in a single motorized *route* density analysis. Cibola NF: Mt. Taylor RD Environmental Assessment for Travel Management Planning, Ch.3, p 55. http://prdp2fs.ess.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5282504.pdf.

- The Grizzly Bear Record of Decision (ROD) for the Forest Plan Amendments for Motorized Access

[5] For a full review see:Switalski, T. A. and A. Jones (2012).

Management within the Selkirk and Cabinet-Yaak Grizzly Bear Recovery Zones (Kootenai, Lolo, and Idaho Panhandle National Forests) assigned route densities for the designated recovery zones. One of the three densities was for Total Motorized Route Density (TMRD) which includes open roads, restricted roads, roads not meeting all reclaimed criteria, and open motorized trails. The agency's decision to use TMRD was based on the Endangered Species Act's requirement to use best available science, and monitoring showed that both open and closed roads and motorized trails were impacting grizzly. Grizzly Bear Plan Amendment ROD. Online at cache.ecosystem-management.org/48536_FSPLT1_009720.pdf.

- The Chequamegon-Nicolet National Forest set forest-wide goals in its forest plan for both open road density and total road density to improve water quality and wildlife habitat.

> *I decided to continue reducing the amount of total roads and the amount of open road to resolve conflict with quieter forms of recreation, impacts on streams, and effects on some wildlife species.* ROD, p 13.

Chequamegon-Nicolet National Forest Land and Resource Management Plan Record of Decision. Online at http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5117609.pdf.

- The Tongass National Forest's EIS to amend the forest plan notes that Alexander Archipelago wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi^2 or less may be necessary.

> *Another concern in some areas is the potentially unsustainable level of hunting and trapping of wolves, when both legal and illegal harvest is considered. The 1997 Forest Plan EIS acknowledged that open road access contributes to excessive mortality by facilitating access for hunters and trappers. Landscapes with open-road densities of 0.7 to 1.0 mile of road per square mile were identified as places where human-induced mortality may pose risks to wolf conservation. The amended Forest Plan requires participation in cooperative interagency monitoring and analysis to identify areas where wolf mortality is excessive, determine whether the mortality is unsustainable, and identify the probable causes of the excessive mortality.*
>
> *More recent information indicates that wolf mortality is related not only to roads open to motorized access, but to all roads, because hunters and trappers use all roads to access wolf habitat, by vehicle or on foot. Consequently, this decision amends the pertinent standard and guideline contained in Alternative 6 as displayed in the Final EIS in areas where road access and associated human caused mortality has been determined to be the significant contributing factor to unsustainable wolf mortality. The standard and guideline has been modified to ensure that a range of options to reduce mortality risk will be considered in these areas, and to specify that total road densities of 0.7 to 1.0 mile per square mile or less may be necessary.* ROD, p 24.

Tongass National Forest Amendment to the Land and Resource Management Plan Record of Decision and Final EIS. January 2008. http://tongass-fpadjust.net/Documents/Record_of_Decision.pdf

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Forest Service | Washington Office | 1400 Independence Avenue, SW
Washington, DC 20250

File Code: 2300/2500/7700 **Date:** November 10, 2010
Route To:

Subject: Travel Management, Implementation of 36 CFR, Part 212, Subpart A (36 CFR 212.5(b)

To: Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors

Travel planning is intended to identify opportunities for the forest transportation system to meet current or future management objectives, based on ecological, social, cultural, and economic concerns. As you know, the Forest Service *Travel Management Rule*, promulgated in 2005, has three parts:

- Subpart A – Administration of the Forest Transportation System;
- Subpart B – Designation of roads, trails, and areas for motor vehicle use; and
- Subpart C – Use by over-snow vehicles.

Over the past 5 years, the Agency has made great strides in completing Subpart B of the *Travel Management Rule* (rule), which was prioritized in order to stop uncontrolled cross-country motor vehicle use. Approximately sixty-seven percent of National Forest System (NFS) lands are covered by a motor vehicle use map. It is anticipated that 93 percent of NFS lands will be covered by December 31, 2010.

Subpart A of the *Travel Management Rule*

This letter is to reaffirm agency commitment to completing those sections of Subpart A of the rule which requires each unit of the NFS to:

- Identify the minimum road system needed for safe and efficient travel and for the protection, management, and use of NFS lands; and
- Identify roads that are no longer needed to meet forest resource management objectives and; therefore, scheduled for decommissioning or considered for other uses (36 CFR 212.5(b)).

By completing the applicable sections of Subpart A, the Agency expects to identify and maintain an appropriately sized and environmentally sustainable road system that is responsive to ecological, economic, and social concerns. Though this process points to a smaller road system than our current one, the national forest road system of the future must provide needed access for recreation and resource management and support watershed restoration and resource protection to sustain healthy ecosystems and ecological connectivity.

Process

Identifying the minimum road system and unneeded roads requires a travel analysis process that is dynamic, interdisciplinary, and integrated with all resource areas. With this letter, I am directing the use of the travel analysis process (TAP) described in Forest Service Manual 7712 and Forest Service Handbook (FSH) 7709.55, Chapter 20, to complete the applicable sections of Subpart A. The TAP is a science-based process that will ensure future travel-management decisions are based on the consideration of environmental, social, and economic impacts. All NFS roads, maintenance levels 1-5, must be included in the analysis.

For units that have previously conducted travel analysis or roads analyses (RAPs), the appropriate line officer should review the prior report to: 1) assess the adequacy of the analysis and the relevance of any recommendations to the process for complying with Subpart A; 2) help determine the appropriate scope and scale for any new analysis; and 3) build on previous work. A RAP completed in accordance with publication FS-643, "Roads Analysis: Informing Decisions about Managing the National Forest Transportation System," will also satisfy the roads analysis requirement of Subpart A.

Although the TAP does not include a National Environmental Policy Act (NEPA) decision, we expect line officers to engage the public in the process, which should involve a broad spectrum of interested and affected citizens, other State and Federal agencies, and tribal governments.

Results from the TAP must be documented in a **travel analysis report**, which should include:

- Information about the analysis and recommendations;
- A map displaying the recommended minimum road system;
- A list of recommended unneeded roads; and
- Further reporting requirements identified in Step 6 of FSH 7709.55, Chapter 20.

Each regional forester must certify that TAP reports for units within their region comply with this direction and are consistent with national policy.

In complying with this direction, units should seek to integrate the steps contained in the Watershed Condition Framework (WCF) with the six TAP steps contained in FSH 7709.55, Chapter 20, to eliminate redundancy and ensure an iterative and adaptive approach for both processes. We expect that the WCF process, and especially the initial watershed condition assessment (Step A) to be completed by March 31, 2011, will provide important information for your work on Subpart A, while the TAP process will likewise provide information for the WCF process. The intent is for each process to inform the other so that they can be integrated and updated with new information or where conditions change. However, the Agency expectation is that each process will move forward: units should not halt one process to wait for the other.

Timing

The travel analysis report **must be completed by the end of FY 2015**. Beyond FY 2015, no Capital Improvement and Maintenance (CMCM) funds may be expended on NFS roads (maintenance levels 1-5) that have not been included in a TAP or RAP.

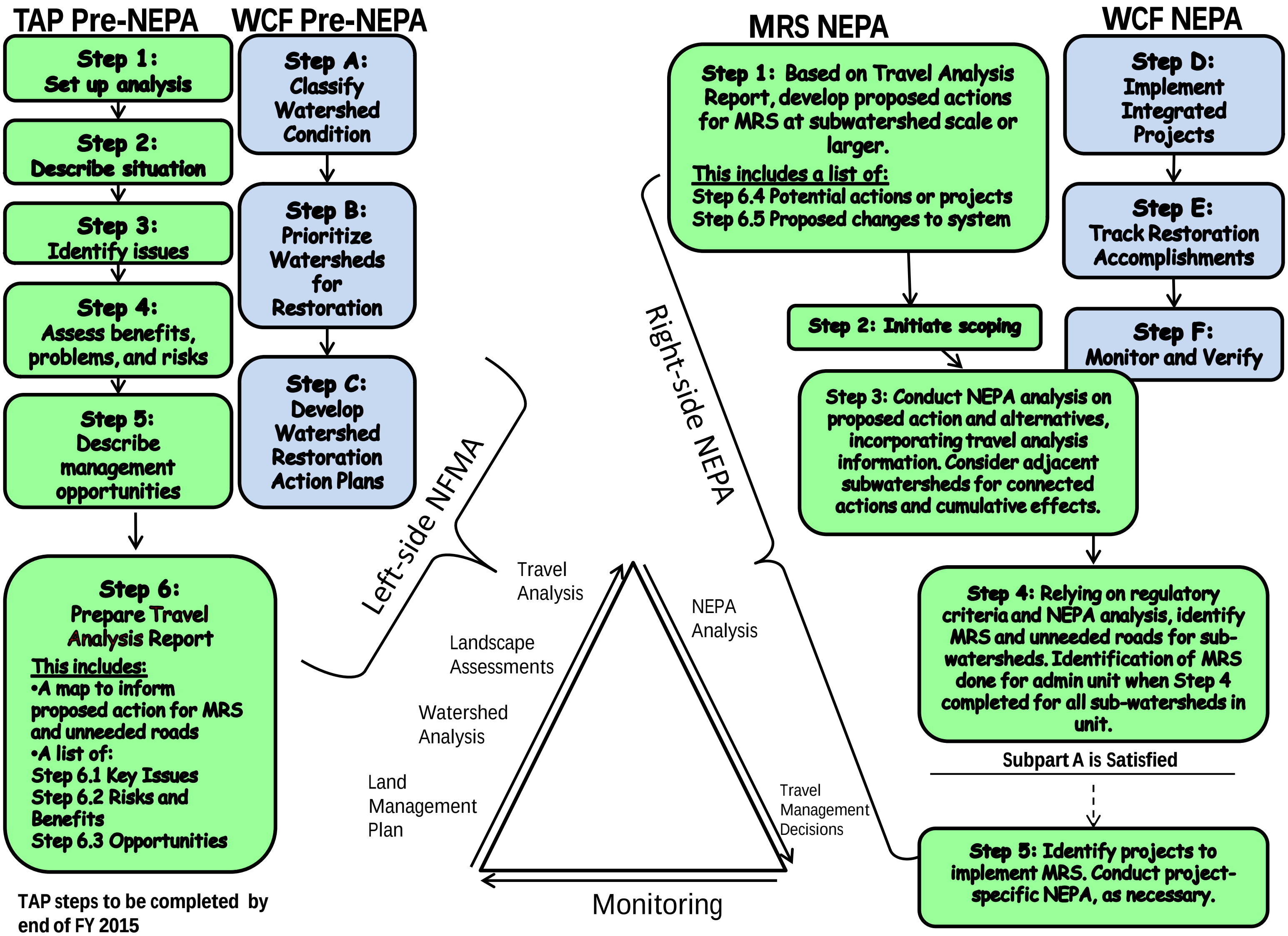
Once certified by the regional forester, units are directed to immediately use the TAP reports to inform resource assessments, project and forest plan NEPA decisions to achieve the TAP recommendations.

Leadership

The Washington Office lead for Subpart A is Anne Zimmermann, Director of Watershed, Fish, Wildlife, Air and Rare Plants. Working with her on the Washington Office Steering Team are Jim Bedwell, Director of Recreation, Heritage, and Volunteer Resources, and Richard Sowa, Director of Engineering. I expect regions to create a similar leadership structure to lead this integrated effort.

This work will require significant financial and human resources. Your leadership and commitment to this component of the *Travel Management Rule* is important. Together, we will move towards an ecologic, economic, and socially sustainable and responsible national road system of the future.

/s/ James M. Pena (for) Joel D. Holtrop
JOEL D. HOLTROP
Deputy Chief, National Forest System



SFL et al. Comments on the DEIS for draft forest plans on the Inyo, Sequoia, and Sierra national forests (August 25, 2016)

Forest Service | Washington Office | 1400 Independence Avenue, SW
Washington, DC 20250

File Code: 2300/2500/7700 **Date:** March 29, 2012
Route To:

Subject: Travel Management, Implementation of 36 CFR, Part 202, Subpart A (36 CFR 212.5(b))

To: Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors

This letter is to reaffirm agency commitment to completing a travel analysis report for Subpart A of the travel management rule by 2015 and update and clarify Agency guidance. This letter replaces the November 10, 2010, letter on the same topic.

The Agency expects to maintain an appropriately sized and environmentally sustainable road system that is responsive to ecological, economic, and social concerns. The national forest road system of the future must continue to provide needed access for recreation and resource management, as well as support watershed restoration and resource protection to sustain healthy ecosystems.

Forest Service regulations at 36 CFR 212.5(b)(1) require the Forest Service to identify the minimum road system needed for safe and efficient travel and for administration, utilization, and protection of National Forest System (NFS) lands. In determining the minimum road system, the responsible official must incorporate a science-based roads analysis at the appropriate scale. Forest Service regulations at 36 CFR 212.5(b)(2) require the Forest Service to identify NFS roads that are no longer needed to meet forest resource management objectives.

Process

Travel analysis requires a process that is dynamic, interdisciplinary, and integrated with all resource areas. With this letter, I am directing the use of the travel analysis process (TAP) described in Forest Service Manual 7712 and Forest Service Handbook (FSH) 7709.55, Chapter 20. The TAP is a science-based process that will inform future travel management decisions. Travel analysis serves as the basis for developing proposed actions, but does not result in decisions. Therefore, travel analysis does not trigger the National Environmental Policy Act (NEPA). The completion of the TAP is an important first step towards the development of the future minimum road system (MRS). All NFS roads, maintenance levels 1-5, must be included in the analysis.

For units that have previously conducted their travel or roads analysis process (RAP), the appropriate line officer should review the prior report to assess the adequacy and the relevance of their analysis as it complies with Subpart A. This analysis will help determine the appropriate scope and scale for any new analysis and can build on previous work. A RAP completed in accordance with publication FS-643, “Roads Analysis: Informing Decisions about Managing the

National Forest Transportation System," will also satisfy the roads analysis requirement of Subpart A.

Results from the TAP must be documented in a **travel analysis report**, which shall include:

- A map displaying the roads that can be used to inform the proposed action for identifying the MRS and unneeded roads.
- Information about the analysis as it relates to the criteria found in 36 CFR 212.5(b)(1).

Units should seek to integrate the steps contained in the Watershed Condition Framework (WCF) with the six TAP steps contained in FSH 7709.55, Chapter 20, to eliminate redundancy and ensure an iterative and adaptive approach for both processes. We expect the WCF process and the TAP will complement each other. The intent is for each process to inform the other so that they can be integrated and updated with new information or where conditions change. The travel analysis report described above must be completed by the end of FY 2015.

The next step in identification of the MRS is to use the travel analysis report to develop proposed actions to identify the MRS. These proposed actions generally should be developed at the scale of a 6th code subwatershed or larger. Proposed actions and alternatives are subject to environmental analysis under NEPA. Travel analysis should be used to inform the environmental analysis.

The administrative unit must analyze the proposed action and alternatives in terms of whether, per 36 CFR 212.5(b)(1), the resulting road system is needed to:

- Meet resource and other management objectives adopted in the relevant land and resource management plan;
- Meet applicable statutory and regulatory requirements;
- Reflect long-term funding expectations;
- Ensure that the identified system minimizes adverse environmental impacts associated with road construction, reconstruction, decommissioning, and maintenance.

The resulting decision identifies the MRS and unneeded roads for each subwatershed or larger scale. The NEPA analysis for each subwatershed must consider adjacent subwatersheds for connected actions and cumulative effects. The MRS for the administrative unit is complete when the MRS for each subwatershed has been identified, thus satisfying Subpart A. To the extent that the subwatershed NEPA analysis covers specific road decisions, no further NEPA analysis will be needed. To the extent that further smaller-scale, project-specific decisions are needed, more NEPA analysis may be required.

A flowchart displaying the process for identification of the MRS is enclosed with this letter.

Timing

The travel analysis report **must be completed by the end of FY 2015**. Beyond FY 2015, no Capital Improvement and Maintenance (CMCM) funds may be expended on NFS roads (maintenance levels 1-5) that have not been included in a TAP or RAP.

Leadership

The Washington Office lead for Subpart A is Anne Zimmermann, Director of Watershed, Fish, Wildlife, Air and Rare Plants. Working with her on the Washington Office Steering Team are Jim Bedwell, Director of Recreation, Heritage, and Volunteer Resources, and Emilee Blount, Director of Engineering. I expect the Regions to continue with the similar leadership structures which have been established.

Your leadership and commitment to this component of the travel management rule is important. Together, we will move towards an ecologic, economic, and socially sustainable and responsible national road system of the future.

/s/ James M. Pena (for):
LESLIE A. C. WELDON
Deputy Chief, National Forest System

File Code: 2300/2500/7700 **Date:** December 17, 2013
Route To:

Subject: Travel Management Implementation

To: Regional Foresters, Station Directors, Area Director, IITF Director, Deputy Chiefs and WO Directors

This letter supplements and reaffirms the direction provided in my 2300 /2500/7700 March 29, 2012, letter regarding the implementation of Subpart A of the Travel Management Rule, and the subsequent September 2012 communication materials.

Continued shared understanding is needed between the Washington Office and the regions regarding the Subpart A travel analysis report (TAR) and supporting map and completion expectations by the September 30, 2015, date.

The March 29, 2012, letter outlined a process for identifying the minimum road system (MRS) and clarified the TAR that must be completed by the end of fiscal year (FY) 2015. Beyond FY 2015, no Capital Improvement and Maintenance (CMCM) funds may be expended on National Forest System (NFS) roads (maintenance levels 1-5) that have not been included in a travel analysis process (TAP) or roads analysis process (RAP).

In line with this, two video teleconferences (VTCs) were held with the Regional Foresters (July 15, 2013, and August 9, 2013) to discuss progress toward completing the TAR and a supporting map by September 30, 2015, to share lessons learned, to clarify expectations for public involvement, and to discuss the final deliverables.

All regions stated they were on track to meet the September 2015 deadline. We were able to reach agreement on what needs to be completed by the deadline. Each forest will produce a TAR , a list of roads “likely not needed for future use” and a map displaying the roads. Forests which have completed their TAR will need to ensure their maps conform to standard.

Enclosed is the map template to use with your completed TAR and the associated steps for producing the map. A forest must complete the necessary analysis, produce a report summarizing this analysis (TAR), a list of roads likely not needed for future use, and synthesize these results in a map that displays roads that are likely needed and likely not needed in the future aligned with the following map example to meet the September 30, 2015 deliverable.

We appreciate the feedback received from the two VTCs and the opportunity to make sure we have a shared understanding of the deliverables. Please contact our WO NFS Director's Steering Team (Rob Harper, Joe Meade, or Emilee Blount) should you have questions on the process or final deliverables.

/s/ James M. Pena (for)
LESLIE A. C. WELDON
Deputy Chief, National Forest System

Enclosures

Forest Service Washington Office 1400 Independence Avenue, SW
Washington, DC 20250

File Code: 7700 **Date:** September 24, 2015
Route To:

Subject: Completion of Travel Management and Next Steps

To: Regional Foresters

As a result of the teleconference held August 17, 2015, and the deadline for completing your Travel Analysis Reports (TARs) September 30, 2015, I want to re-emphasize the Chief's expectations and next steps. Prior to considering the TAR final, review each to ensure the intent has been met and the reports are complete. As required by Subpart A of the Travel Management rule; each unit of the National Forest System must:

- Identify the minimum road system needed for safe and efficient travel and for administration, utilization, and protection of National Forest System lands;
- Identify the roads on lands under Forest Service jurisdiction that are no longer needed to meet forest recreation and resource management objectives and reflect long-term funding expectations; and,
- Decommission or consider other uses of those roads identified as unneeded.

As you are aware, completion of the TAR involves three parts:

1. Travel Analysis Process (TAP), a map displaying all system roads that differentiates between those roads which are likely needed from those roads which are likely not needed;
2. List of each road clearly showing the relationship to your TAP, integrated with your analysis, your rationale; and,
3. Clarification of proposed changes to your system roads.

Once your review is complete, please send the link where your TAR is located to Leslie Boak, Acting National Transportation Program Manager at ljboak@fs.fed.us for posting on Forest Service internal Web site at http://fsweb.wo.fed.us/eng/. The Washington Office (WO) travel management leadership team comprised of the Directors for Engineering, Technology and Geospatial Services; Watershed, Fish, Wildlife, Air and Rare Plants; and Recreation, Heritage and Volunteer Resources will monitor your progress and will provide a National WO Review. The TARs are not considered final until both reviews are complete, at which time, the TARs will be available to the public.

If you have any questions, please contact Brian Ferebee, Associate Deputy Chief, National Forest System, at (202) 205-0824, or by email at bferebee@fs.fed.us.

/s/ Brian Ferebee (for)

LESLIE A. C. WELDON
Deputy Chief, National Forest System

cc: Glenn P. Casamassa

Examples of road plan components from existing National Forest Land Management Plans

Last Updated: August 2016

Topic	Forest	Example of Road Component	LRMP Date
Road density	San Juan National Forest	Road Density Guideline for Water Quality and Watershed Health on SJNF Lands: In order to protect water quality and watershed function, road densities on SJNF lands should not exceed 2 miles/square mile within any U.S. Geological Survey (USGS) 6th level Hydrologic Unit Code (HUC) watershed. In order to protect major surface source water protection areas for municipalities within USGS 6th level HUC watersheds, road densities on NFS lands should not exceed 1.5 miles/square mile. If new road construction is necessary on NFS lands within an area exceeding this density guideline, management actions should be considered that would result in post-construction road densities that are equal to or less than the pre-construction density. The following parameters and constraints will be used to calculate road density for water quality and watershed health: 2.13.27a: Roads used to develop road density calculations include those roads on NFS lands only, regardless of road ownership, that are a) open year-long or seasonally to public use and b) closed to public use, but are used for administrative access or are authorized by contract, permit, or other written authorization. Included in these calculations are NFS maintenance level 2–5 roads. Non-motorized and motorized trails and those roads that are closed to all motorized use and/or are in storage are not used for road density calculations. Temporary roads to be used for 5 years or less are not included in these calculations. 2.13.27b: Road densities will be calculated within USGS 6th level HUC watersheds on NFS lands only. 2.13.27c: Municipal watersheds are USGS 6th level HUC watersheds where the surface source water intake exists for an incorporated town, city, or other municipality with a public water supply. The MOU between the USFS Region 2 and the CDPHE states, "Revised Forest Plans will provide direction and desired conditions for municipal supply watersheds/source water areas to protect water quality while allowing for multiple use outputs (per 36 CFR	2013

		251.9 and FSM 2542)." 2.13.27d: Data used for density calculations will be based on the best available information at the time of analysis. Road and Motorized Trail Density Guideline for Ungulate Production Areas, Winter Concentration Areas, Severe Winter Range, and Critical Winter Range on SJNF Lands: The intent of this guideline is to ensure no net loss of existing habitat effectiveness within the areas listed below. In order to maintain wildlife habitat effectiveness of SJNF lands, road and motorized trail densities should be addressed when analyzing and approving management actions that affect motorized routes. Where management actions would result in road and motorized trail densities exceeding 1 mile/square mile on SJNF lands in the areas listed below, actions should be designed to maintain habitat effectiveness on SJNF lands throughout each mapped polygon. Habitat effectiveness for this guideline is considered maintained when road densities within the CPW mapped areas on SJNF lands listed below are less than or equal to 1 mile/square mile. When road densities exceed 1 mile/square mile within the CPW mapped areas on SJNF lands listed below, densities should not be increased without mitigation designed to maintain habitat effectiveness. - Big game production areas (calving or lambing areas) - Elk and deer severe winter range - Elk and deer winter concentration areas - Deer critical winter range The following parameters and constraints will be used to calculate road and motorized trail density for wildlife: 2.13.29a: Roads used to develop route density calculations include roads on NFS lands only, regardless of road ownership, that are a) open year-long or seasonally to public use and b) closed to public use, but are used for administrative access or are authorized by contract, permit, or other written authorization. Included in these calculations are maintenance level 2–5 NFS roads. Also included for this calculation are NFS trails that are designated for motorized use. Roads and motorized trails with design features sufficient to maintain habitat effectiveness (such as seasonal closures that are determined to be sufficient mitigation), as determined by the USFS biologist, should not be used for final density calculations. Non-motorized trails and those roads that are closed to all motorized use and/or are in storage are not used	

		for route density calculations. Temporary roads to be used for 5 years or less are not included in these calculations. 2.13.29b: Data used for density calculations will be based on the best available information at the time of analysis. 2.13.31: Road and Motorized Trail Density Guideline for Deer and Elk General Winter Range on SJNF Lands: Where management actions would result in road and motorized trail densities exceeding 1 mile/square mile and where CPW analysis determines that road and motorized trail densities inhibit the state's ability to meet population objectives, SJNF management actions should be designed to reduce the impacts of road density on habitat effectiveness throughout each mapped general winter range polygon. This guideline applies to the portions of each mapped general winter range polygon not covered under Guideline 2.13.29. The following parameters and constraints will be used to calculate road and motorized trail density for wildlife: 2.13.31a: Roads used to develop route density calculations include roads on NFS lands only, regardless of road ownership, that are a) open year-long or seasonally to public use and b) closed to public use, but are used for administrative access or are authorized by contract, permit, or other written authorization. Included in these calculations are maintenance level 2–5 NFS roads. Also included for this calculation are NFS trails that are designated for motorized use. Roads and motorized trails with design features sufficient to maintain habitat effectiveness (such as seasonal closures that are determined to be sufficient mitigation), as determined by the USFS biologist, should not be used for final density calculations. Non-motorized trails and those roads that are closed to all motorized use and/or are in storage are not used for route density calculations. Temporary roads to be used for 5 years or less are not included in these calculations. 2.13.31b: Data used for density calculations will be based on the best available information at the time of analysis.	
	Chequamegon-Nicolet National Forest	Goal 3.1 – Capital Infrastructure: Build and maintain safe, efficient, and effective infrastructure that supports public and administrative uses of National Forest System lands. Retain and progress toward the Forestwide average total road density goal of 3.0 miles per square mile established in 1986.	2004

		Objective 3.1: Reduce average open and total road density on the Chequamegon-Nicolet National Forests. Use Appendix BB, "Guide for Reducing Open and Total Road Density" and Road Density Map in Map Packet to focus efforts.	
Temporary Roads	San Juan National Forest	Standard 2.13.22: No temporary road shall be constructed . . . prior to the development of a project-specific plan that defines how the road shall be managed and constructed. The plan must define the road design, who are responsible parties and their roles in construction, maintenance and decommissioning, the funding source, a schedule for construction, maintenance and decommissioning, the method(s) for decommissioning, and post-decommissioning monitoring requirements for determining decommissioning success."	2013
	White Mountain National Forest	Standard: Temporary roads must be decommissioned upon completion of the activity for which they were authorized.	2005
Minimum Road System and Subpart A requirements	Monongahela National Forest	Goal, RF02: Provide developed roads to the density and maintenance level needed to meet resource and use objectives. During watershed or project-level planning: a) Update inventory of area transportation system. b) Determine the minimum transportation system necessary to achieve access management objectives. c) Incorporate cost efficiency into construction, reconstruction and maintenance needs. d) Identify roads to decommission, obliterate, replace, or improve that are causing resource damage. e) Integrate needs for off-road parking.	2006
	Beaverhead-Deerlodge National Forest	Goal: The minimum transportation system necessary is identified and managed…	2011
Decommissioning and sustainability	Coconino National Forest	Objective: Naturalize or decommission 200 to 800 miles of unauthorized roads and system roads to create a more cost effective road system and to restore natural resources impacted by roads during the 10 years following plan approval. Guideline: To maintain an efficient and sustainable road system, unneeded roads	2013

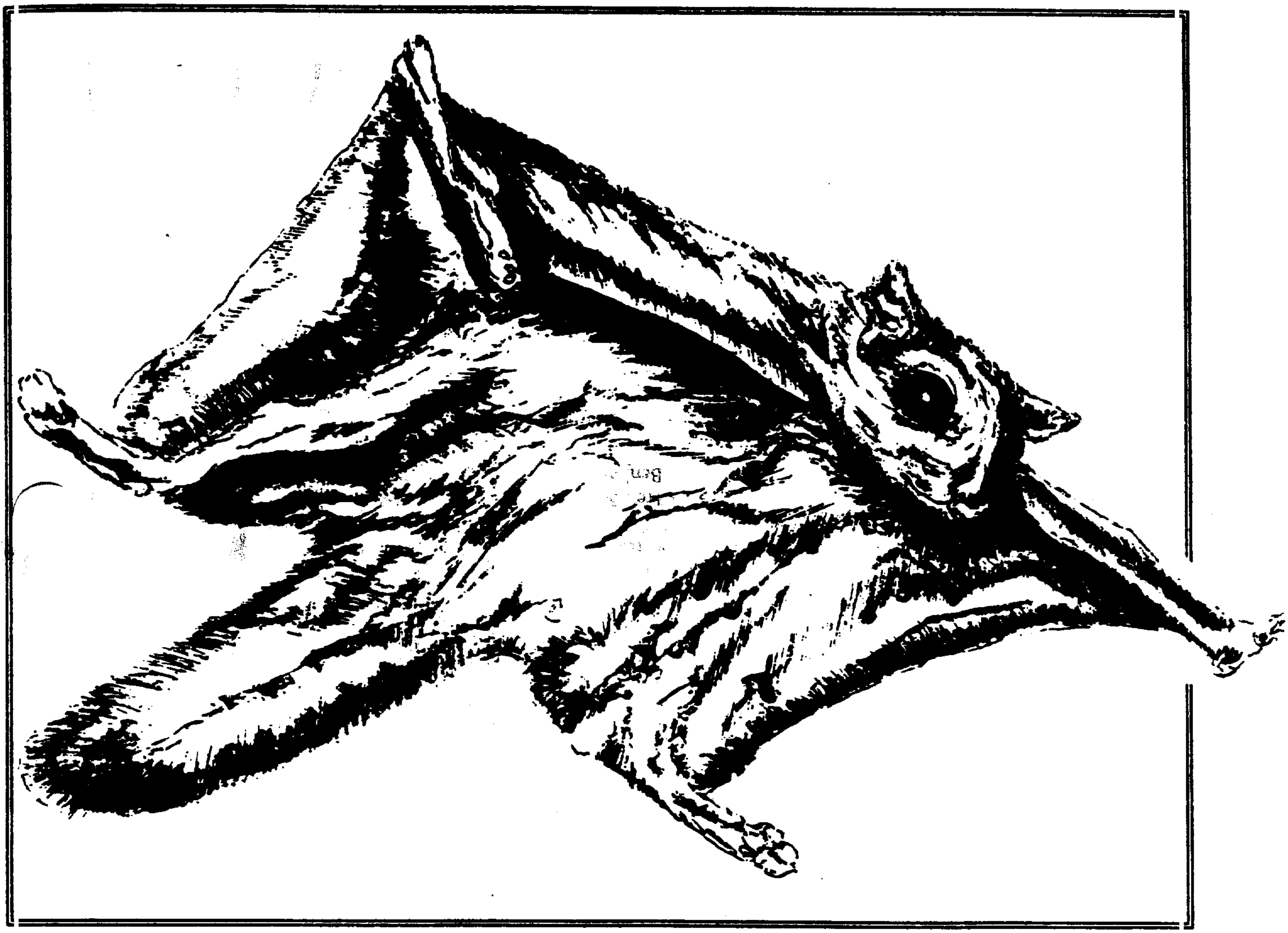
		should be decommissioned. Factors in prioritizing the naturalization of decommissioned and unauthorized roads should include the following: 1. Watershed Condition - Soils that are receiving, or are expected to receive, damage to the extent that soil productivity is or will be significantly impaired outside of the road prism. - Riparian areas (e.g., springs, wetlands, or stream reaches) that are impaired due to sedimentation or alterations to hydrology related to the road. - Meadows at the TES montane meadows polygon map unit scale that are likely to be or being damaged. - Poorly located, designed, or maintained roads connected to downstream impaired waters, where potential for increased runoff and sedimentation is high. 2. Wildlife, Fish, and Plants - Habitats for threatened, endangered, or sensitive species that are susceptible to roads as barriers or roads as mortality hazards. 3. Social and Cultural Values - Areas of high or very high scenic integrity. - Roads that provide undesirable access to archaeological sites and areas of traditional cultural use by local tribal members. - Areas where user conflict must be resolved or to ensure public safety. - Semiprimitive nonmotorized ROS objectives as set through environmental analysis. - Roads where use levels or road maintenance causes adverse noise effects to wildlife during key periods in their life cycle or to recreational experiences. - Redundant roads. - Roads that are not identified on the motor vehicle use map (MVUM), which are not needed for administrative purposes. - Roads that continue to be used for public access despite motorized restrictions.	
	Jefferson National	Objective 33.01. Analyze transportation system within one watershed per year	2004

	Forest	through watershed analysis, and identify roads to be decommissioned. (See also Objective 1.02). Objective 33.02. Priorities for decommissioning are roads causing resource damage and roads in areas where the desired condition is to reduce open road density.	
	Chequamegon-Nicolet National Forest	Guideline: Road decommissioning and restoration priorities: • Resource protection and (or) restoration. • Abandoned roadbeds and unneeded access roads associated with road relocation. • Meeting desired road densities within Wilderness study areas, Management Areas 6A and 6B (semi-primitive non-motorized areas), wild and scenic riverways, Moquah Barrens, and Riley Lake Wildlife Management Area. • Meeting desired road densities within Research Natural Areas, Special Management Areas, and Old Growth and Natural Feature Complexes. • Local roads that connect to arterial or collector roads scheduled for reconstruction. • Working towards desired total road density within areas not listed above and shown as 2.0 mile/square mile open road density on Road Density Map (See Map packet).	2004
Connectivity	Coconino National Forest	Management Approach: - Consider wildlife and plant habitat needs early in the transportation and development planning process. - Work closely with the Arizona Game and Fish Department, Arizona Wildlife Linkages Working Group, Arizona Department of Transportation, and others to identify linkages and potential barriers to wildlife movement and to mitigate such threats during project design.	2013
Cross-boundary integration	Coconino National Forest	Management Approach: - Cooperate with the National Park Service (NPS) to identify Forest Service roads near boundaries with national monuments that should be closed or decommissioned from the system to prevent trespass onto NPS land.	2013
Visitor experiences	Jefferson National Forest	Standard: Road construction is not allowed within Semi-Primitive Motorized or Non-Motorized areas except during an emergency or as subject to valid existing rights and leases. (See standards under Recreation Opportunity Spectrum.)	2004

80/9180006

APPALACHIAN NORTHERN FLYING SQUIRRELS

(Glaucomys sabrinus fuscus) *(Glaucomys sabrinus coloratus)*



RECOVERY PLAN

U.S. FISH & WILDLIFE SERVICE

DEPARTMENT OF THE INTERIOR

Region 5
U.S. Fish and Wildlife Service

62pp.

809180006

Appalachian Northern Flying Squirrels

(<u>Glaucomys</u> <u>sabrinus</u> <u>fuscus</u>)
(<u>Glaucomys</u> <u>sabrinus</u> <u>coloratus</u>)

Recovery Plan

Prepared by:

Annapolis Field Office
U. S. Fish and Wildlife Service
Annapolis, Maryland

in cooperation with the

Northern Flying Squirrel Recovery Team

For:

Region 5
U.S. Fish and Wildlife Service
Newton Corner, Massachusetts

Approved: Ronald E. Lambertson
Regional Director,
U.S. Fish and Wildlife Service

Date: September 24, 1990

62pp.

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EXECUTIVE SUMMARY -- Appalachian Northern Flying Squirrels

Species status: Endangered

Habitat requirements: High-elevation forests in southern Appalachians, usually in spruce/fir-hardwood ecotone

Recovery Objective: To de-list the Virginia and Carolina northern flying squirrels

Recovery criteria:

To down-list: (1) Populations are stable or expanding at ≥ 80% of designated Geographic Recovery Areas for 10 years; (2) Sufficient life history data are available to permit effective management; (3) GRAs are managed for squirrels in perpetuity.

To de-list: In addition to 1, 2, and 3, continued existence of high-elevation forests is assured.

Actions Needed:

1. Survey for new populations and monitor known populations.
2. Study habitat requirements.
3. Study diet, interactions with other squirrels and genetics.
4. Study effects of various land use practices (mining, logging, recreation).
5. Ensure implementation of appropriate habitat management guidelines, based on results of 1-3. (This would include periodic monitoring, even following de-listing.)

Total estimated cost of recovery: (X 1000)

Year	Need 1	Need 2	Need 3	Need 4	Need 5	Total
1991	80	60	30	30	25	225
1992	80	120	30	30	25	285
1993	80	120	40	30	25	285
1994	80	--	10	15	25	120
1995	70	--	--	10	25	105
1996	60	--	--	10	25	95
1997	40	--	--	10	25	75
1998	25	--	--	--	25	50
1999	25	--	--	--	25	50
2000	25	--	--	--	25	50
Total	565	300	110	135	250	1360

Down-listing may be initiated by the year 2000, depending on population status.

* * *

Recovery plans delineate reasonable actions believed to be required to recover and/or protect listed species. Plans are published by the U.S. Fish and Wildlife Service, sometimes prepared with the assistance of recovery teams, contractors, state agencies, and others. Objectives will be attained and any necessary funds made available subject to budgetary and other constraints affecting the parties involved, as well as the need to address other priorities. Recovery plans do not necessarily represent the views, official position, or approval of any individuals or agencies involved in plan formulation, other than the U.S. Fish and Wildlife Service. They represent the official position of the Service only after they have been signed by the Regional Director or Director as approved. Approved recovery plans are subject to modification as dictated by new findings, changes in species status, and the completion of recovery tasks.

Literature citations should read as follows:

U.S. Fish and Wildlife Service. 1990. Appalachian Northern Flying Squirrels (Glaucomys sabrinus fuscus and Glaucomys sabrinus coloratus) Recovery Plan. Newton Corner, Massachusetts. 53 pp.

Additional copies may be purchased from:

Fish and Wildlife Reference Service
5430 Grosvenor Lane, Suite 110
Bethesda, Maryland 20814
301-492-6403 or 1-800-582-3421

Fees vary depending on number of pages.

TABLE OF CONTENTS

List of Figures

PART I: INTRODUCTION

Description

The two endangered subspecies of northern flying squirrel, Glaucomys sabrinus fuscus Miller and Glaucomys sabrinus coloratus Handley, are small, nocturnal, gliding mammals 260-305 mm in total length and 90-140 g in weight. They possess a long, broad, flattened tail (80% of head and body length), prominent eyes, and dense, silky fur. The distinctive patagia (folds of skin between the wrists and ankles) are fully furred and supported by slender cartilages extending from the wrist bones; these plus the broad tail create a large gliding surface area and are the structural basis for the squirrel's characteristic gliding locomotion (Thorington and Heaney, 1981). Adults are dorsally gray with a brownish, tan, or reddish wash, and grayish white or buffy white ventrally. Juveniles have uniformly slate gray backs and off-white undersides. The more southern subspecies, G. s. coloratus, is larger (avg. 286 vs. 266 mm total length) than G. s. fuscus, with a longer tail (avg. 134 vs. 115 mm) and brighter coloration (Handley, 1980).

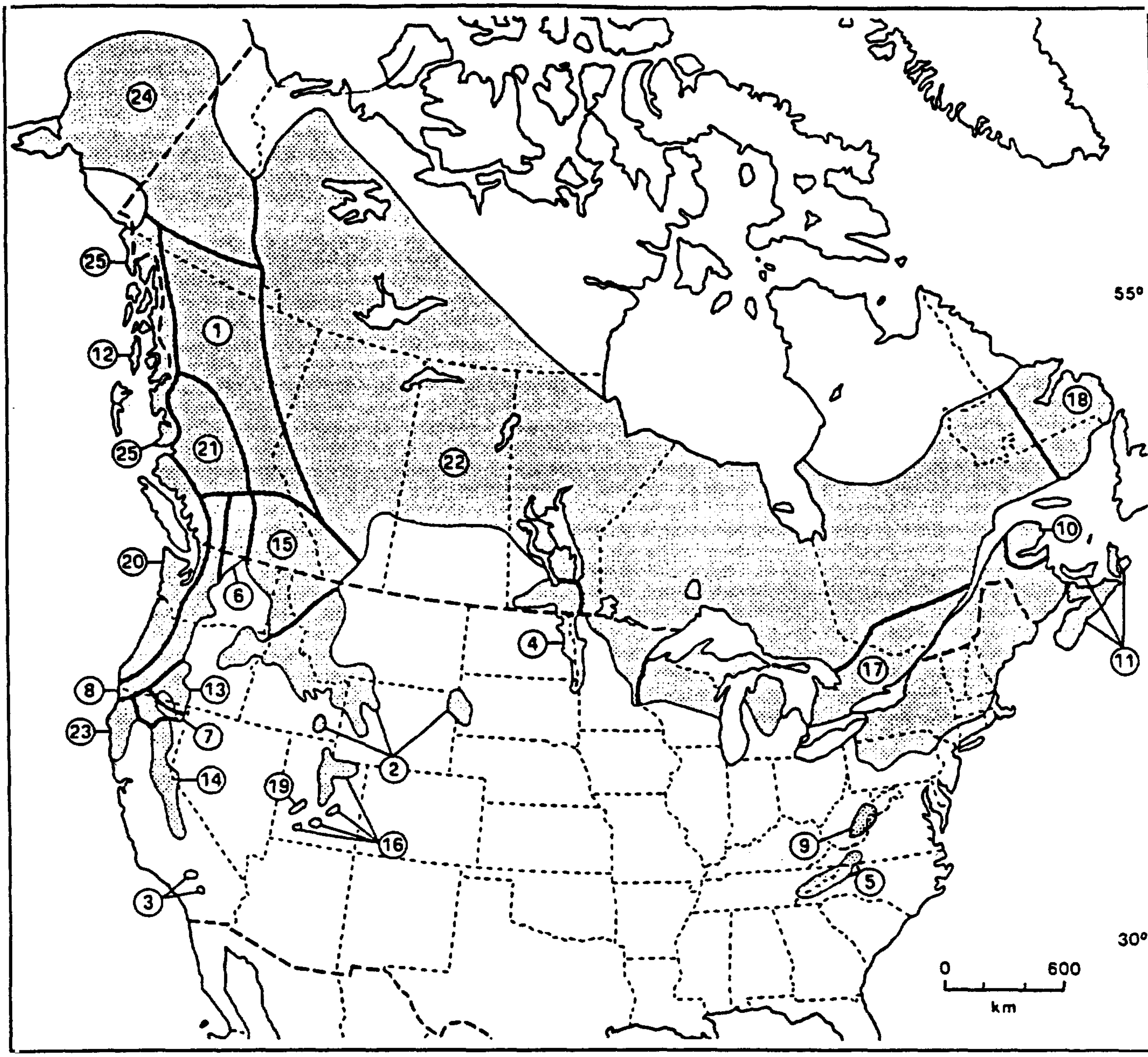
Glaucomys sabrinus can be distinguished from the southern flying squirrel, G. volans, by its larger size (e.g., hindfoot 33-41 mm vs. < 33 mm for G. volans); greater adult weight (90-140 g for G. sabrinus vs. 50-90 g for G. volans); the gray base of its ventral hairs as opposed to the white base in the southern species; the relatively longer upper tooth row; and the much shorter, stouter baculum (penis bone) of the males. A full account of the species' taxonomic history can be found in Howell (1918) and Wells-Gosling and Heaney (1984). The original descriptions of the two Appalachian subspecies are available in Miller (1936) and Handley (1953). Wells-Gosling (l985) provides numerous photographs of both Glaucomys species.

Distribution

A total of 25 subspecies of Glaucomys sabrinus occur in boreal coniferous and mixed coniferous/hardwood forests of the northern United States and Canada, th mountain ranges of the western United States, and certain highland areas of the southern Appalachian Mountains (Figure 1). The general distribution of the two endangered subspecies in the southern Appalachians is shown in Figure 2.

Prior to their Federal listing, these two subspecies were known from fewer than 3 specimens collected from eight localities (U.S. Fish and Wildlife Service, 1985). Since these subspecies were listed, intensive field work as well as apparent population increases in some localities have led to the capture of many additional animals and their discovery in new areas. Sites of capture prior to 1985 are considered historic localities.

The subspecies fuscus is now known from the following areas in West Virginia: (1) the Stuart Knob area (Randolph County); (2) the Cheat Bridge area (Pocahontas and Randolph Counties); (3) the Cranberry area (Greenbrier, Pocahontas, Randolph, and Webster Counties); (4) the Spruce Knob area (Pendleton and Randolph Counties); and (5) the Blackwater Falls area (Tucker County). At least 187 G. s. fuscus have been captured in these areas since intensive efforts to locate the animals began in 1985 (C. Stihler, WV DNR, pers. comm.). In Virginia G. sabrinus is known from three localities: (1) Highland County; (2) the Whitetop-Grayson Highlands area (Smyth and Grayson Counties); and (3) one site in Montgomery County (J. Cranford, Biology Dept., VPI&SU, pers comm., 1985). The habitat at this third site is atypical, and northern flying squirre have not been captured here since 1982. A total of 46 individuals have been captured in Virginia since 1985 (M. Fies, VA Department of Game and Inland Fisheries; J. Pagels, VA Commonwealth University, pers. comm., 1990).



Numbers 5 and 9 indicate G.s. coloratus and fuscus respectively. Other subspecies are as follows: 1) G.s. alpinus, 2) G.s. bangsi, 3) G.s. californicus, 4) G.s. canescens, 6) G.s. columbiensis, 7) G.s. flaviventris, 8) G.s. fuliginosus, 10) G.s. goodwini, 11) G.s. gouldi, 12) G.s. griseifrons, 13) G.s. klamathensis, 14) G.s. lascivus, 15) G.s. latipes, 16) G.s. lucifugus, 17) G.s. macrotis, 18) G.s. makkovikensis, 19) G.s. murinauralis, 20) G.s. oregonensis, 21) G.s. reductus, 22) G.s. sabrinus, 23) G.s. stephensi, 24) G.s. yukonensis, 25) G.s. zaphaeus. Modified from: Wells-Gosling and Heaney (1984).

Figure 1. General Distribution of Glaucomys sabrinus

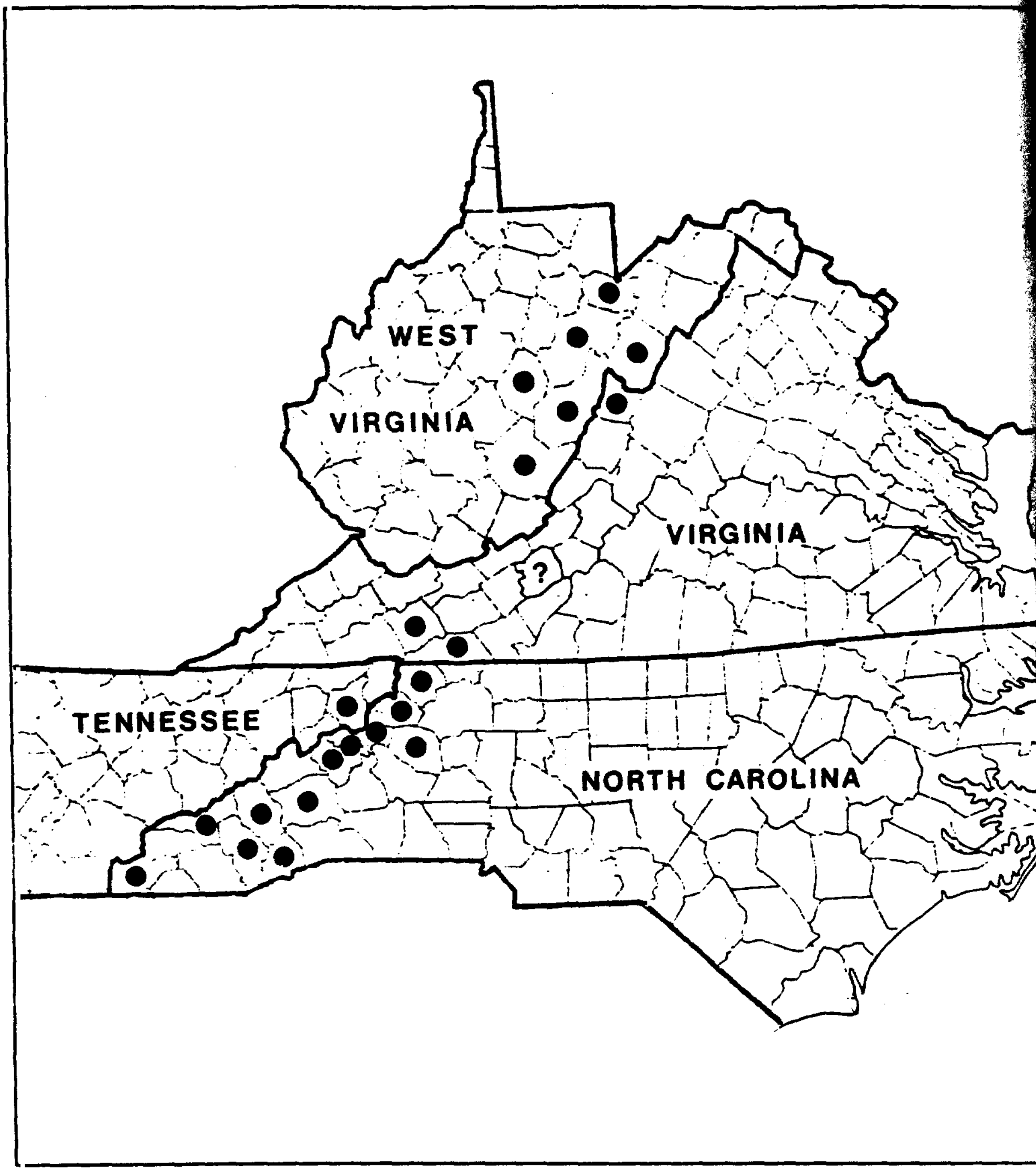


Figure 2. Distribution of Glaucomys sabrinus fu
and G. s. coloratus

Glaucomys sabrinus coloratus, the southernmost subspecies, has now been found in the following isolated localities in North Carolina and Tennessee: (1) the Roan Mountain area (Mitchell County, NC and Carter County, TN); (2) the Grandfather Mountain area (Avery, Caldwell, and Watauga Counties, NC); (3) the Black Mountains, including Mt. Mitchell (Buncombe and Yancey Counties, NC); (4) the Great Balsam Mountains (Haywood and Transylvania Counties, NC); (5) the Plott Balsam Mountains (Haywood and Jackson Counties, NC); (6) the Great Smoky Mountains (Jackson and Swain Counties, NC); (7) the Unicoi Mountains (Cherokee County, NC); and (8) the Long Hope Valley area (Ashe and Watauga Counties, NC). Approximately 150 northern flying squirrels have been captured in these areas since 1985 (P. Weigl, pers. comm.).

The taxonomic status of Glaucomys sabrinus in southern Virginia (Smyth and Grayson Counties) presents a special situation. Due to the geographic proximity of these populations to those in North Carolina and their apparent intermediate morphological characteristics (C. Handley, Jr., Smithsonian Institution, pers. comm., 1988), we are for the purposes of this plan including these squirrels within the subspecies coloratus. This is a management, not a taxonomic, decision. The true subspecific affiliation of these animals will be determined through analysis of additional specimens as they become available.

The pre-settlement distribution of Glaucomys sabrinus in the Southeast is unknown, but fossil remains indicate a much larger range during the late Pleistocene and early Holocene (Kurten and Anderson, 1980; Lundelius et al., 1983; Semken, 1983). The disjunct distribution of these subspecies in the southern Appalachians and their great distance from the center of the species' range in the northern United States and Canada suggest that they are relicts which have become isolated in small patches of suitable habitat by changing climatic and vegetational conditions since the last ice age.

Habitat

Throughout its eastern range the northern flying squirrel is usually associated wi boreal habitats, especially spruce-fir and northern hardwood forests. In the southern Appalachians G. sabrinus shows a relict distribution and tends to occu rather small and potentially vulnerable islands of high elevation habitat. The subspecies fuscus and coloratus are commonly captured in conifer-hardwood ecotones or mosaics consisting of red spruce (Picea rubens) and fir (Abies frase and A. balsamea) associated with mature beech (Fagus grandifolia), yellow birch (Betula alleganiensis), sugar maple (Acer saccharum) or red maple (Acer rubrum hemlock (Tsuga canadensis), and black cherry (Prunus serotina). A recent habit analysis of 13 capture sites in the southern Appalachians revealed that while species composition of the occupied forest may vary in different locations, some combination of hardwoods and conifers (particularly spruce and fir) appears essential to support these animals. Understory components did not appear to b significant indicators of G. sabrinus habitat (Payne et al., 1989). Studies with captives indicate that G. sabrinus will readily use both deciduous and coniferous environments. In contrast, both distributional data and experimental studies indicate that G. volans has a marked preference for hardwood forests (Weigl, 1978).

It could be argued that the capture of northern flying squirrels in conifer/hardwoo ecotonal areas may be partially an artifact of the elimination of large, contiguous spruce or stands in the southern Appalachians and the over-sampling of ecotonal habitats. However, trapping that has been conducted in the remaining stands of pure conifers has so far failed to yield any sabrinus (Weigl and Boynton, 1990).

Although conifers are clearly an important component of G. sabrinus habitat, northern flying squirrels have also been taken in deciduous areas some distance from spruce-fir forest in the central Appalachians and in New England (P. Weigl, pers. obs.). As mentioned above, Dr. J. Cranford of Virginia Polytechnic Institute and State University (pers. comm.) captured the species in April of 1978 and March of 1982 in a riparian hemlock-hardwood-rhododendron forest in Montgomery County, Virginia. Nest boxes placed at this site have so far revealed no additional northern flying squirrels. Recently, a juvenile female G. s. coloratus was captured in the Unicoi Mountains of North Carolina, 45 km away from the nearest natural spruce-fir stands (A. Boynton, NC Wildlife Resources Commission, pers. comm., 1989). Further study is needed to determine the importance of the spruce-fir forest component to the survival of northern flying squirrels in the southern Appalachians.

Northern flying squirrels have been captured in stands of varying age, understory density, and composition, but most have been taken from moist forest with at least some widely spaced, mature trees and an abundance of standing and down snags (ideally old-growth forest). Such habitats seem well suited to the species' gliding form of locomotion, use of cavities for nesting, and reliance on wood-borne fungi and lichens for food. The relative abundance of natural cavities in old hardwoods and their resistance to windthrow (compared to many conifers) may account for the northern flying squirrel's occupation of mixed woodland and deciduous forest just below the spruce-fir zone.

Since the northern hardwood/spruce-fir ecotone occurs at progressively higher elevations from north to south, it is not surprising that captures of G. sabrinus show a similar latitudinal trend. While individuals have frequently been found at elevations less than 800 m in New England, New York, and Pennsylvania, most West Virginia specimens have been taken at 1000-1350 m (C. Stihler, pers. comm.). In Virginia G. sabrinus generally occupies forests 1170-1630 m in altitude. In North Carolina all captures have occurred above 1540 m with the exception of two individuals, one taken in the Great Smoky Mountains at 1230 m and the other taken in the Unicoi Mountains at 1463 m (Hall, 1981; Weigl, 1968; A. Boynton, pers. comm., 1989).

Life History and Ecology

Because of their rarity, nocturnal and secretive habits, and the remoteness of th habitat, little was known of the ecology of northern flying squirrels in the southe Appalachians prior to their listing (Weigl, 1977). In the forests they occupy, the presence usually goes unnoticed. These squirrels have been located during th day by shaking or pounding on snags and dead branches, especially if these a hollow. They sometimes nest in bluebird boxes and will occasionally come to b feeders (Wells-Gosling, 1985). Their presence is often betrayed by their characteristic repeated warning calls or "chirps."

Unlike G. volans, G. sabrinus are not highly dependent on seeds and nuts for f (Weigl, 1978), and, in fact, may not be able to use conifer seeds effectively in so boreal habitats (Brink and Dean, 1966). Over much of their range they can apparently subsist on lichens and fungi (Maser et al., 1985; Weigl, 1968), but als eat seeds, buds, fruit, staminate cones, insects, and other animal material (McKeever, 1960), and have been observed ingesting tree sap (Foster and Tate, 1966; Schmidt, 1931). The year-round abundance of lichens and many species hypogeous (underground) fungi may provide a steady and, at certain seasons, almost exclusive food supply and may reduce food competition with other squirr species. A recent analysis of habitat features in Virginia revealed lichen abundan to be significantly correlated with the presence of G. sabrinus (J. Pagels, pers. comm., 1990). Periodic dependence on certain species of fungi may be a factor restricting the species to high-elevation, mesic habitats. Studies in the Pacific Northwest have indicated that northern flying squirrels play an important role in forest maintenance by dispersing nitrogen-fixing bacteria and fungal spores that form symbiotic mycorrhizal relationships with overstory species (Maser and Mase 1988).

Because of the flying squirrel's small size, the climatic severity of its habitat, and abundance of avian and mammalian predators, secure nesting sites represent a critical limiting factor. During the cooler months squirrels commonly occupy tree cavities and woodpecker holes (Baker, 1983; Jackson, 1961). Recently, they have been observed using the dense branches in the tops of spruce and fir trees as a winter refuge (P. Weigl, pers. obs.). In the summer these squirrels construct and use outside leaf nests (Cowan, 1936; Urban, 1988; Weigl and Osgood, 1974). The interior of both types of nest is lined with lichens, sedges, moss, or finely chewed bark. Squirrels have been observed entering burrows in the ground (C. Stihler, pers. comm.; Weigl, 1968; Urban, 1988), although the extent of their use is not yet known.

Northern flying squirrels are relatively gregarious and commonly share nests (Osgood, 1935; Maser et al., 1981). In West Virginia, seven adult G. sabrinus were recently observed in a single nest box and four seen in another (C. Stihler, pers. obs.); however, the spectacular winter nesting aggregations reported for G. volans (up to 50 in a nest) are unknown for this species. Northern flying squirrels apparently live in family groups of adults and juveniles, for when the species has been located, it has been possible to trap 2-8 individuals within a discrete area (P. Weigl, pers. obs.). Also, adult females have been found in nest boxes with juveniles that are clearly large enough to fend for themselves (C. Stihler, pers. obs.).

Only limited reproductive information is available for these subspecies. Investigators working with other subspecies mention two litters of 2-6 young per year and a gestation period of 37-42 days (Muul, 1969; Davis, 1963). Trapping data from the southern Appalachians provide evidence of only a single litter in spring or summer. Fourteen litters observed in nest boxes in Virginia and West Virginia over the past five years contained from one to five young, with an average of 2.9 young per litter (C. Stihler and M. Fies, pers. obs.). Two captive females from North Carolina each had litters of four young (P. Weigl, pers. obs.). Wea occurs at about two months of age. Normal longevity in the wild is unknown, b individual squirrels have been observed to reach four years of age.

Telemetry studies in the southern Appalachians have provided some data on th northern flying squirrel's activity and use of space. Animals radiotracked durin summer have a marked biphasic activity pattern with peaks between sundown a midnight and 1-3 hours before sunrise (Weigl and Osgood, 1974). During these times squirrels are extremely active in trees and on the ground and enter a num of different nests or refuges (Ferron, 1981). Studies in both West Virginia (Urba 1988) and Alaska (Mowrey and Zasada, 1982) and earlier accounts (e.g., Cover 1932) confirm this tendency to spend long periods moving along the ground. It possible that such behavior is associated with foraging on hypogeous fungi. Wh heavy fog (cloud), rain, and high winds delay the onset and decrease the intens of activity, they do not suppress it altogether (Radvanyi, 1959). At such times, flying squirrels appear to spend more time moving along the branches than glidi

Summer telemetry data also suggest individual home ranges of 2-3 hectares in North Carolina (Weigl and Osgood, 1974) and 5-7 hectares in West Virginia (Urb 1988). Radiotracking and trapping studies indicate approximate squirrel densities of one squirrel per 2-3 hectares in areas of good habitat. In Alaska, *G. s. yukonensis* have been observed moving their daily ranges within a large forested area, and using up to 34 alternate den trees (R. Mowrey, pers. comm.) Mowrey suggests that possible explanations for the long distance night-to-night movemen of *sabrinus* might include: (1) taking advantage of fungal "blooms" in remote areas, (2) a near miss by a predator forcing the squirrel into a new area, (3) ectoparasites in a previous nest, or (4) adverse weather conditions.

Recent telemetry studies in North Carolina have revealed that in winter G. sabrinus cover large areas (over 30 ha) in a short time and may move almost a kilometer in a direct line in a few minutes (Weigl and Boynton, 1990). Further data are required to determine with certainty the size and habitat characteristics of the area needed by an individual or to maintain a stable population of these squirrels.

Difficulties of Present and Future Research

All available information indicates that these two subspecies are rare over most of their range and restricted in their choice of habitat. In spite of the field studies of Linzey (1983) and extensive small-mammal trapping by numerous researchers in many parts of the Appalachians, relatively few range extensions have been reported. It is, of course, possible that the squirrel's rarity, secretive nature, and remote habitat militate against detection of populations at old or new sites. It is also conceivable that these squirrels may periodically abandon particular habitats or undergo periodic population oscillations and thus become undetectable for extended intervals. Such changes in "detectability" have been observed by Osgood (1935), J. Christian (pers. comm. to Weigl, 1970), P. Weigl (pers. obs.), and R. Mowrey (pers. comm.).

Not only are G. s. fuscus and coloratus relatively new to science, rare, and apparently localized in distribution, they are also extremely difficult to collect and study. Weigl (1968) had a capture success of one animal per 80 trap-nights in an area known to have a resident population; this record includes several week-long trapping sessions without any captures. Overall capture success in the West Virginia study at Stuart Knob, also an area with a known resident population, was one animal per 127 trap-nights (Stihler et al., 1987; Urban, 1988). These represent the minimal effort; R. Mowrey (pers. comm.) reported even lower trapping success in Alaska. Two techniques -- live-trapping and use of nest boxes -- have been used successfully to collect these squirrels. Trapping has proved the most tim effective collecting method; however, the placement and periodic checking of n boxes will likely produce more captures and demographic information per unit effort in the long run. Reports of G. sabrinus using bluebird boxes in West Vir (D. Hollingsworth, U.S. Forest Service, pers. comm., 1985) indicate the potenti attractiveness of artificial nest structures to these animals. State wildlife biolog and U.S. Forest Service personnel within the range of G. s. fuscus and colorat have undertaken extensive nest box placement programs, which are providing a wealth of additional data on squirrel locations and habits. Other potential mean detecting the presence of G. sabrinus include observation of tracks in snow or a prepared substrate, use of feeding stations, night viewing devices, etc. Whate technique is employed, working with these animals is a highly unpredictable endeavor and is likely to remain so until more is known about their ecology and behavior.

Reasons for Listing

The limited and discontinucus range of this species in the southern Appalachian makes it vulnerable to a number of both natural and human-related impacts. Ev without human intervention, small, relict populations might suffer disproportionat from genetic constraints (e.g., increased homozygosity) as well as from climatic and vegetational processes associated with post-Wisconsin changes in mountai environments. However, habitat destruction, fragmentation, or alteration associa with clearing of forests, introduced insect pests, mineral extraction, recreational o other development, pollution (heavy metals, pesticides, acid rain), and the poten for global warming outweigh any known natural threats to the species or its habi For example, in West Virginia red spruce, an important component of northern flying squirrel habitat, originally covered nearly half a million acres. Timbering operations beginning in the 1880's and ending in the 1920's removed all but 200 acres of spruce! Roughly 20% of the original forested areas have since regenerated, but not all of this acreage has yet attained the maturity characteristic of good flying squirrel habitat (Bones, 1978; Zinn and Sutton, 1976).

Introduced pests, in particular, the balsam wooly adelgid (Adelges piceae) [Ratz] [possibly also the gypsy moth (Lymantria dispar)] threaten to further reduce the extent and quality of remaining forest habitats required by a conifer-hardwood ecotone species like G. sabrinus. The balsam wooly adelgid, accidentally introduced from Europe around 1900, has spread throughout the fir forests of the eastern United States. This insect is a relatively innocuous parasite of firs in Europe, but it is extremely damaging in North America. In the eastern United States, the balsam fir (Abies balsamea) and the Fraser fir (A. fraseri) are the host species, with the latter sustaining the more serious damage and higher mortality. The death of Fraser firs occurs within 2-7 years following the initial infestation by the adelgid. Although Fraser firs were estimated to cover some 60,000 acres in the southern Appalachians (Barry and Oprean, 1979), it has been predicted that if current trends continue, the balsam wooly adelgid will eliminate mature Fraser firs within the next several decades and may eventually cause the extinction of this southern Appalachian endemic conifer (Eager IN White, 1984). The impact of this potential extinction on sabrinus is not known.

Paradoxically, the solution to one problem may be the cause of another. Lindane (gamma isomer of benzene hexachloride), the primary chemical used to control the balsam wooly adelgid, has come under scrutiny due to its toxicity to aquatic organisms (Ulmann, 1972) and its persistence in the environment. Biodegradable alternatives, such as potassium oleate, an insecticidal soap, have been used with some success on balsam wooly adelgids. However, their usefulness is limited, since they lack residual activity.

Spruce and spruce-fir declines and die-offs associated with factors other than th adelgid have become of increasing concern in the Northeast and at higher elevations in the southern Appalachians (Adams et al., 1985; Vogelmann, 1982). Although acid precipitation is believed to play some role in these declines, its ex role, as well as the contribution of heavy metal pollution, is still being investigate (Zedaker et al., 1989). High elevation sites in the spruce-fir zone of the souther Appalachian Mountains exhibit higher concentrations of heavy metals such as le copper, nickel, zinc, and manganese in forest floor material and soil than low elevation sites in the same region. Lead concentrations have been found to be much as ten times higher on the summit of Mount Mitchell, North Carolina (northern flying squirrel habitat), than in surrounding lowlands (Bruck, 1984). In some high elevation forests, lead concentrations approach those of urban areas and areas adjacent to highways (Bogle and Turner IN White, 1984). The possib of lead and copper toxicity to plants needs to be investigated with relation to the decline of conifers at high elevations in the southern Appalachians. Vogelmann (1982) suggested a possible synergistic effect of lead and acid rain, resulting in death or the sharp decline of red spruce and other plant species. Heavy metals may also have direct effects on the squirrels. For example, lichens and mycorrh fungi are known to accumulate lead (Dey IN White, 1984) and could thus pass th contamination to flying squirrels; the toxicity of lead (Eisler, 1988) and other heav metals to animals is well documented.

In addition to synergistic chemical effects, acid rain may exert deleterious effects conifers through other subtle interactions. For example, Bruck (1984) reported li successful reproduction of fir, spruce, or woody shrubs above 6,350 feet on Mou Mitchell. In this area acid rain has been found to destroy the mycorrhizae living i association with conifer roots, interfering with the regeneration and vigor of the trees. Petersen (IN White, 1984) also mentioned a potential connection between the decline of high elevation conifers and the effect of acid precipitation-caused declines in mycorrhizal symbionts. As a result, northern flying squirrels could be affected by loss or contamination of both their mycorrhizal food source and their coniferous habitat. Intensive investigations of the causes of spruce/fir decline are currently being conducted by the U.S. Forest Service and EPA, as a part of the National Acid Precipitation Assessment Program.

Modification of northern flying squirrel habitat may also have favored the spread and proliferation of competitors and pathogens. Research with captive animals indicates that G. sabrinus may be displaced by the more aggressive and agile G. volans in certain hardwood habitats where their ranges overlap (Weigl, 1978). It is unknown to what extent southern flying squirrels are expanding their range into northern flying squirrels' habitat and, if this does occur, whether sabrinus will be displaced. Evidence on the species' interactions is mixed. In two areas in North Carolina once occupied by sabrinus, only volans are now captured (P. Weigl, pers. obs.). While both species have recently been captured in close proximity in North Carolina, West Virginia, and Virginia, previous work elsewhere suggests that such sympatry is often unstable (Osgood, 1935; J. Christian, pers. comm., P. Weigl, pers. obs.). Both species have been trapped near Stuart Knob, Randolph County, West Virginia, at intervals over the past 36 years (C. Stihler, pers. comm.), but the extent and frequency of their sympatry over that time period are not clearly documented. Further studies of the species' interactions are indicated. There is also some evidence that the southern flying squirrel harbors a parasitic nematode (Strongyloides robustus) which, if transferred to the northern species, could prove lethal or debilitating (Weigl, 1975), especially in the more southern parts of the species' ranges.

Strategy for Recovery

The limited nature of existing data on the two southern subspecies of Glaucomys sabrinus and the potential vulnerability of their habitat suggest a four-part strategy for recovery. First, it is necessary to determine the distribution of the species i southern Appalachians by conducting surveys of former capture sites and new areas with apparently suitable habitat. This task is well underway. Second, are found to support this species or especially favorable habitat conditions must receive adequate protection from human-related disturbance. Fortunately, the majority of areas occupied by these endangered squirrels is in public ownershi (U.S. Forest Service, National Park Service) and these agencies are cooperatin management for the squirrels. Third, a concerted effort must be made to obtain information on flying squirrel ecology -- in particular, habitat requirements, diet, relations with G. volans. Finally, the squirrels' response to various habitat modification measures should be studied. These studies should focus on habita enhancement measures (e.g., thinning of dense stands of spruce regeneration) well as determining timber harvest methods that are compatible with protection a maintenance of squirrel populations.

Trapping and nest box captures to date have revealed clusters of capture sites, such that general areas of occupancy by these squirrels may be described. For the purposes of assessing recovery, we are defining these as "Geographic Recovery Areas" (GRAs) for each subspecies.

In keeping with the section on distribution, the following GRAs are noted for G. s. fuscus:

1. the Stuart Knob area (Randolph County, WV)
2. the Cheat Bridge area (Pocahontas and Randolph Counties, WV)
3. the Cranberry area (Greenbrier, Pocahontas, Randolph, and Webster Counties, WV)
4. the Blackwater Falls area (Tucker County, WV)
5. the Spruce Knob/Laurel Fork area (Pendleton and Randolph Counties, WV and Highland County, VA)

GRAs for G. s. coloratus are:

1. the Roan Mountain area (Mitchell County, NC and Carter County, TN)
2. the Grandfather Mountain area (Avery, Caldwell, and Watauga Counties, NC)
3. the Black Mountains (Buncombe and Yancey Counties, NC)
4. the Great Balsam Mountains (Haywood and Transylvania Counties, NC)
5. the Plott Balsams (Haywood and Jackson Counties, NC)
6. the Great Smoky Mountains (Haywood and Swain Counties, NC)
7. the Unicoi Mountains (Cherokee County, NC and Monroe County, TN)
8. the Long Hope Valley area (Ashe and Watauga Counties, NC)
9. the Whitetop-Grayson Highland area (Smyth and Grayson Counties, VA)

Additional GRAs may be defined as further survey data are accumulated.

PART II: RECOVERY

Recovery Objective:

To remove Glaucomys sabrinus fuscus and Glaucomys sabrinus coloratus from t[he] list of endangered and threatened species.

This is envisioned as a two-step process. **Down-listing** from endangered to threatened status will be possible when it can be documented that:

1. squirrel populations are stable or expanding (based on biennial sampling over a 10-year period) in a minimum of 80% of all Geographic Recovery Areas designated for the subspecies,

2. sufficient ecological data and timber management data have been accumulated to assure future protection and management, and

3. GRAs are managed in perpetuity to ensure: (a) sufficient habitat for population maintenance/expansion and (b) habitat corridors, where appropriate elevations exist, to permit migration among GRAs.

De-listing will be possible when, in addition to the above factors, it can be demonstrated that:

4. the existence of the high elevation forests on which the squirrels depend is not itself threatened by introduced pests, such as the balsam wooly adelgid or by environmental pollutants, such as acid precipitation or toxic substance contamination.

Recovery criteria for the two subspecies will be assessed independently. For example, the threat imposed by the balsam wooly adelgid to Fraser firs in the southernmost portions of the Appalachians may preclude recovery of G. s. coloratus beyond threatened status.

Narrative Outline of Recovery Tasks

1.0 Establish a recovery advisory committee to coordinate all recovery actions.

Because these squirrels occur in two FWS regions and because of the increasing interest from government and academic agencies in studying them, such a committee is necessary to ensure that recovery criteria are being met, to provide a centralized data repository, and to ensure that research efforts are not duplicated or inconsistent. The advisory committee initially will be comprised of members of the recovery team, although membership may change over time.

2.0 Determine distribution and viability of G. sabrinus populations in the southern Appalachians.

Accurate knowledge of the species' distribution is essential for protecting individual populations, understanding the relationships among populations, and monitoring long-term population changes. Great strides have been made towards completion of this task, as indicated by the establishment of GRAs. At this time, it is a matter of filling in the gaps in our information base.

2.1 Delineate occupied and potential habitat.

Historic and recent capture data provide strong evidence of a habitat preference for conifer-hardwood ecotones and mosaics, especially at higher elevations. Potential habitat may be defined areas with vegetation and elevational components similar to tha known occupied habitats. Spruce stands in much of the easter U.S. have been mapped and the data compiled in a GIS databa by the U.S. Forest Service. These data are available from the Forest Service and have been obtained by the state wildlife biologists involved with northern flying squirrel research. Information on other potential habitat types may be obtained fro aerial photos, cover type and photographic maps, and forest sta data. This task has been largely completed, although some refinement may be necessary.

2.2 Survey potential habitat to locate additional populations.

First priority areas to survey are mature spruce/fir/northern hardwood forests and ecotonal areas. Other types of habitat tha should also be surveyed, but are a lower priority, include hemlock/hardwood forests, especially in riparian areas, and northern hardwood stands.

Surveys of potential habitat via the placement and monitoring of nest boxes and/or live-trapping have proven to be highly successful. Other techniques, such as night-scope observation a feeding stations, auditory surveys, smoked aluminum and/or sno tracking and hair identification may be useful, but require addition research.

2.3 Monitor known populations.

More frequent monitoring may be at times desirable. For example, if late summer breeding is suspected, an additional check in late summer is warranted. Moreover, certain key sites may be designated for yearly monitoring. In addition to population trend data, monitoring will provide data on weights and measurements, litter size and breeding seasons, sex ratio, age structure, and social behavior; over time, information on life expectancy may be acquired as well.

As much information as possible will be obtained in the course of monitoring, including fecal samples for dietary and parasite analysis, ectoparasites, etc.

3.0 Obtain life history and ecological information for known populations of G. sabrinus of the southern Appalachians.

Such studies are necessary to determine critical factors favoring survival, growth, and reproduction.

3.1 Conduct in-depth studies of habitat requirements.

Such information is very limited for G. s. fuscus and coloratus, yet it is essential for determining whether recovery goals are being met and for making informed management decisions. For example, we must learn more about the squirrels' seasonal habitat requirements and home range to determine the size and configuration of areas that need to be protected and what manipulations, if any, are permissible.

3.11 Determine the importance of spruce and fir forest components to the survival of sabrinus.

As noted in the Habitat section, the two endangered squirrel subspecies have been captured almost exclusiv in stands containing spruce or spruce/fir. Yet in the Ur Mountains of North Carolina and Tennessee, sabrinus i apparently surviving in the complete absence of a spruc component. One capture site in West Virginia also has virtually no spruce component, although there are spruc stands nearby (C. Stihler, pers. obs.). Notably, both of these sites do contain hemlock in the overstory. An understanding of the importance of a coniferous forest component to these squirrels is crucial, for example, in cases where firs are being extirpated by the balsam woo adelgid, or where widespread spruce die-offs are occurri To gain this understanding, a comparative study of movements and habitat use of squirrels in areas with no spruce or fir, versus that of squirrels in more typical habit will be conducted. The importance of spruce-fir may also be determined via long-term monitoring of populations in non-spruce/fir habitats and in areas where spruce or fir mortality is high. Such areas will be monitored annually, a opposed to biennially, to detect any subtle changes that may occur.

3.111 Monitor ongoing studies of loss or degradation of high elevation forest resulting from insect damage and air pollution.

Since the ecology of _Glaucomys sabrinus_ in this region is intimately linked to boreal forests and ecotonal areas, widespread die-offs or reduced growth of spruce/fir or northern hardwood forests associated with insect damage or environmental contaminants could have a tremendous negative impact on these squirrels. A major wide-reaching study of spruce/fir decline is presently being conducted jointly by EPA and the U.S. Forest Service as a part of the Forest Response Program, a sub-program of the National Acid Precipitation Assessment Program (NAPAP). Two of the southern Appalachian intensive study areas in this investigation are at Mt. Rogers and Mt. Mitchell, both known to be occupied by northern flying squirrels. Squirrel researchers should be aware of this program and coordinate with spruce-fir decline investigators to understand the rates and causes of forest loss and to interpret effects of these on _sabrinus_.

3.12 _Study the relationships among population size, habitat size and habitat quality._

Answers to such questions as 'How large a population size can a given area of habitat support?' are notably elusive and relate necessarily to the quality of the habitat in question. However, such questions must be addressed if we are to manage these squirrels effectively in the long run. Answering these questions will require long-term

comparative studies of squirrel numbers and habitat us
during different seasons and in different habitat types.

3.13 <u>Study the effects of modification or loss of habitat resu</u>
<u>from timber operations or other development</u>.

Timber harvest often precedes other developmental activities such as powerline construction or recreational development. The effects of road construction and vari timber harvest and reforestation methods on <u>sabrinus</u> w be evaluated primarily on Forest Service lands. While to loss of spruce is almost certainly detrimental to <u>G</u>. <u>sabri</u> some thinning or opening of the canopy could be acceptable or even beneficial to these squirrels. Exampl of timber harvest methods to be assessed include:
(1) small block cuts, as opposed to larger clearcuts;
(2) cuts of irregular shape, to complement site-specific topographic or vegetational features; (3) shelterwood cut (approximately 40 square feet basal area per acre) for regeneration; and (4) removal of over-mature hardwoods (but not along the spruce-hardwood ecotone).

Additionally, a study or studies should be conducted specifically to determine use by <u>sabrinus</u> of clearcuts of various ages as well as their use of areas where other timber management techniques have been employed. These studies should be closely coordinated with those conducted under Task 3.11.

Certain timber management practices may favor G. sabrinus over G. volans (e.g., possibly selection against heavy mast-producing species). This possibility could be examined in conjunction with studies conducted under Task 3.3.

Some experimental timber harvesting has already been permitted in G. sabrinus habitat on the Monongahela National Forest, with an eye toward determining long-term impacts to the squirrels (W. Tolin, U.S. Fish and Wildlife Service, Elkins, WV, pers. comm., 1989). Additionally, squirrels have been located in West Virginia on a large tract with several areas that have been recently timbered. Following the response of these populations will enhance our knowledge of the long-term effects of timber harvest on these squirrels.

3.2 Study the diet of G. sabrinus.

The importance of lichens and hypogeous fungi to northern flying squirrels in the southern Appalachians requires further examination as it relates to potential competition with G. volans, the importance of G. sabrinus in forest maintenance, and the potential contamination of this food source by pesticides or heavy metals.

3.21 Examine the role of G. sabrinus in the dispersal of mycorrhizal fungi and forest maintenance.

The role that G. sabrinus plays in dispersing mycorrhizal spores, thus promoting forest regeneration (Maser et al., 1978), requires further investigation. Part of this work co be done in association with ecological or monitoring stu by collecting feces and determining whether certain fung spores are present. Further investigations may be necessary to understand fully the relationships between these flying squirrels, mycorrhizal fungi, and forest regeneration.

3.22 Investigate potential accumulation of toxics in food supp

As already stated, pesticides and heavy metals are findin their way into the northern flying squirrel's environment, b their effects are not yet known. Tests for the presence o Lindane in animals and stream water adjacent to treated areas on Mt. Mitchell have been negative (Eager IN Whit 1984). Monitoring studies were also conducted recently Roan Mountain, in conjunction with adelgid control by th U.S. Forest Service. Redback voles (*Clethrionomys gapperi*), which occupy northern flying squirrel habitat an have similar food habits, were collected for tissue analysi to determine levels of Lindane, as well as other potentiall harmful pesticides and heavy metals. Much more work is needed on biomagnification of toxins in high elevation habitats.

Although previous studies have shown that lichens concentrate toxic substances, extensive studies have not been conducted in areas occupied by northern flying squirrels. Preliminary collection and analyses of lichens from flying squirrel habitat should reveal any potential problems and determine whether additional analyses are indicated.

3.3 Study interactions with other squirrels.

3.31 Examine behavioral interactions.

The relationship between G. sabrinus and G. volans needs to be studied, to determine whether interspecific interactions impact negatively upon G. sabrinus in the long run. If this proves to be the case, habitat occupied by sabrinus will be managed to favor this species over volans. Potential competition for nesting sites between G. sabrinus and red squirrels (Tamiasciurus hudsonicus) should also be investigated.

3.32 Examine effects of Strongyloides and other parasites or diseases.

Field and laboratory studies should be conducted to elucidate more fully the pathogenicity of this parasite to G. sabrinus, and to determine whether G. volans is an effective vector. Preliminary evidence indicates that a light infestation, as has recently been found in several specimens in West Virginia and Virginia, appears to be tolerated (Pagels et al., 1990); heavier infestations may weaken the animals so that they succumb to other stresses such as pneumonia. Any other pathogens found to affect G. sabrinus specifically should be similarly studied.

4.0 Determine genetic variability within and among populations.

As our knowledge of habitat requirements expands, there will come a [t]ime when we should examine genetic variability within and among populati[ons] (= GRAs), to gain a better understanding of the origin of and interacti[ons] among populations. This information will assist in determining the appropriateness and necessity of maintaining or establishing migration corridors among the various GRAs. Additionally, genetic studies may r[eveal] any population segments with an unusual amount of diversity or rare al[leles]. Techniques used to obtain genetic material (e.g., blood sampling) shou[ld] be standardized for all researchers and designed to avoid any possibilit[y of] significant injury to the animals.

5.0 Develop management guidelines.

5.1 Develop and refine habitat management guidelines for agencies and private landowners involved in habitat-altering activities withi[n] the range of G. s. fuscus or coloratus.

Guidelines developed for private landowners and general guideli[nes] for National Forests appear in Appendix A. All national forests a[nd] parks within the subspecies' range will be encouraged to adopt similar guidelines tailored for their own needs. Guidelines will be revised as more information becomes available.

5.11 Where these guidelines specify placement of nest boxes i[n] project areas, designate a "data coordinator" for each sta[te] to keep up with results and regularly report findings to sta[te] and Federal wildlife agencies and to the recovery advisor[y] committee.

Because placement of nest boxes is becoming a primary means of gathering data on the effects of specific management activities (i.e., road building and timber sales), the number of boxes placed in project areas may become quite large. One individual per state should be assigned the task of keeping track of nest box data, live-trapping or other survey methods. State coordinators should compile a list of captures each year for the recovery advisory committee.

5.2 Develop policy and, if appropriate, methodology for translocation/reintroduction and captive rearing.

At this time, the distribution, abundance, and the genetic interchange among populations of Glaucomys sabrinus in the southern Appalachians is incompletely known. Until such information becomes available, it is not prudent to consider relocation and/or introduction an appropriate management tool. This policy may change as more is learned about life history and distribution of these squirrels.

6.0 Implement appropriate management and protection procedures.

6.1 Implement habitat management guidelines on public lands and encourage their use on private lands.

On Federal lands or where Federal permits, funding, or authorization are involved, appropriate management will be implemented through consultation under Section 7 of the Endangered Species Act. Natural resources agencies will encourage the adoption of appropriate management for those private actions affecting squirrel habitat.

6.2 Protect occupied habitat through land acquisition or other means as appropriate.

Encourage protection of unprotected occupied habitat via conservation easement, fee title acquisition, long-term lease, etc., by Federal, state, or local government agencies or by private conservation groups, in order to ensure habitat protection in perpetuity.

6.3 Protect individual squirrels and their habitat through vigorous enforcement of the Endangered Species Act and other applicable Federal and state laws.

The southern Appalachians are receiving more and more impact from mining of high elevation deposits of valuable low-sulphur coal and from recreational interests, including the development of ski resorts and vacation communities in higher elevation habitat. This mining is generally permitted by the states, in coordination with the Fish and Wildlife Service. Whenever possible, biologists should provide design input to decrease the impacts of such activities. If any Federal funds, permits, or authorization are involved, such projects would require review through Section 7 of the Endangered Species Act. Where no permits are required, developers will still be encouraged to consider potential presence of northern flying squirrels in their development plans.

7.0 Implement information/education programs.

7.1 Provide educational/management training for state and Federal foresters, game managers, and others.

Training will be provided to familiarize these individuals with the northern flying squirrel and the types of habitat it occupies.

All appropriate biologists, foresters, etc., should receive training, so that they will be sensitized to the presence of potential northern flying squirrel habitat in the course of their day-to-day activities. One such workshop was conducted in 1986 at Roan Mountain.

7.2 Prepare and distribute educational displays and informational materials.

Pamphlets, brochures, and/or displays will be used to inform the public of the differences between southern and northern flying squirrels, the importance of old-growth northern forest types to the latter species, and the adverse effects of habitat loss or modification.

7.3 Coordinate with private landowners to eliminate or minimize threats to populations.

A major threat to these squirrels is habitat alteration associated with human activities. This and other threats can be eliminated or minimized through education of individuals and public and private land-owning organizations (Tasks 7.1 and 7.2) and through land acquisition (Task 6.2).

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PART III: IMPLEMENTATION

Priorities in column one of the following implementation schedule are assigned as follows:

1. Priority 1 - All actions that are absolutely essential to prevent extinction of the species.
2. Priority 2 - All actions necessary to maintain the species' current population status.
3. Priority 3 - All other actions necessary to provide for full recovery of the species.

Appalachian Northern Flying Squirrels
Implementation Schedule

September, 1990

Priority	Task Number	Task Description	Task Duration	Responsible Agency* FWS	Responsible Agency* Other	Cost Estimates** ($000) FY1	FY2	FY3	Comments
1	2.1	Delineate occupied and potential habitat.	Completed	R4,5 / FWE	USFS, NPS, and SWA	--	--	--	
1	2.2	Survey potential habitat to locate additional populations.	Continuous	R4,5 / FWE	USFS, NPS, and SWA	45	45	45	
1	2.3	Monitor known populations.	Continuous	R4,5 / FWE	SWA	35	35	35	
1	3.13	Study effects of timber operations.	7 years	R4,5 / FWE Research	USFS and SWA	30	30	30	
1	3.32	Examine effects of _Strongyloides_ and other pathogens.	3 years	R4,5 / FWE	SWA	10	10	10	
1	5.1	Develop and refine habitat management guidelines.	1 year	R4,5 / FWE	USFS, NPS, and SWA	--	--	--	Guidelines for use on private and public lands appear in Appendix A.
1	6.1	Implement habitat management guidelines	Continuous	R4,5 / FWE	USFS, NPS, and SWA	25	25	25	
1	6.3	Protect individual squirrels and their habitat through enforcement of the ESA and other laws.	Continuous	R4,5 / FWE, LE	USFS, NPS, and SWA	1	1	1	
2	3.11	Determine importance of [illegible]	3 years	R4,5 / FWE	USFS and [illegible]	[illegible]	[illegible]	[illegible]	

Priority	Task Number	Task Description	Task Duration	Responsible Agency*		Cost Estimates** ($000)			Comments
				FWS	Other	FY1	FY2	FY3	
2	3.111	Monitor ongoing studies of effects of spruce/fir die-offs.	Continuous	R4,5 / FWE	USFS, EPA, NPS, and SWA	--	--	--	
2	3.12	Study population size/ habitat size and quality relationships.	3 years	R4,5 / FWE	USFS and SWA	30	60	60	
2	3.22	Investigate accumulation of toxics in food supply.	3 years	R4,5 / FWE (EC)	SWA	10	10	10	
2	3.31	Study behavioral interactions with other squirrels.	3 years	R4,5 / FWE	SWA	10	10	10	
2	4.0	Examine genetic variation within and among populations.	2 years	R4,5 / FWE	SWA	--	--	10	
2	5.11	Designate data coordinator in each state.	Continuous	R4,5 / FWE	SWA	3	3	3	
2	5.2	Develop policy for translocation/reintroduction.	---	R4,5 / FWE	SWA	--	--	--	Accomplished methodology will be developed if/when policy changes.
2	6.2	Protect occupied habitat through land acquisition, etc.	Continuous	R4,5 / FWE	USFS, NPS, SWA, and TNC	1	1	5	
2	7.1	Provide training for state and Federal foresters, game managers, and others.	Continuous	R4,5 / FWE	USFS and NPS	1.5	1.5	1.5	

Priority	Task Number	Task Description	Task Duration	Responsible Agency*		Cost Estimates** ($000)			Comments
				FWS	Other	FY1	FY2	FY3	
2	7.3	Coordinate with private landowners to eliminate or minimize threats to populations.	Continuous	R4,5 / FWE	SWA	2	2	2	
3	3.21	Study dispersal or mycorrhizal fungi.	2 years	R4,5 / FWE	SWA	2	2	--	
3	7.2	Prepare and distribute educational displays and informational materials.	Continuous	R4,5 / FWE	SWA	2	2	2	Results of ongoing research are needed to develop brochures and displays.

* FWS = U.S. Fish and Wildlife Service
R4,5 = Region 4, Region 5
FWE = Division of Fish and Wildlife Enhancement
LE = Division of Law Enforcement
EC = Environmental Contaminants Section

USFS = U.S. Forest Service
NPS = National Park Service
SWA = State wildlife agencies of all participating states (VA Department of Game and Inland Fisheries, WV Department of Natural Resources, NC Wildlife Resources Commission, and TE Wildlife Resources Agency).
TNC = The Nature Conservancy

** Cost estimates are for all funds: Federal, state, and private.

APPENDIX A
Suggested Guidelines for
Habitat Identification and Management

Guidelines may vary, depending on location, land use, and land ownership. Below, we present general guidelines for use on private lands, and guidelines designed for use on National Forests. These guidelines are subject to change as more data are gathered on the ecological requirements and associates of these flying squirrel subspecies in the southern Appalachians.

(1) General Guidelines for Management of G. s. fuscus and G. s. coloratus Habitat on Private Lands.

1. Potential habitat includes areas of mature spruce and/or fir stands, pure or mixed with northern hardwood and/or hemlock trees. In the southern Appalachians these forested areas are generally found at elevations above 3,300 feet, or higher further south.

2. Ideally, potential habitat, particularly old-growth areas, should be maintained intact; while limited selective cutting may be conducted, clearcutting should be avoided.

3. Any timber rotation schedules should be of a sufficient length to maintain the old-growth character of the area.

4. Nest boxes may be installed and checked regularly, to determine whether northern flying squirrels are occupying the area. Installation and checking of boxes should be coordinated with state non-game wildlife agencies. Permits are required for working with any endangered or threatened species.

(2) Suggested Standards and Guidelines for Habitat Management of the Endangered Flying Squirrels (Glaucomys sabrinus fuscus and G. s. coloratus) o[n] Public Lands.

I. Habitat Identification

A. Occupied Habitat is defined as any area where G. s. fuscus or coloratus is known to exist through positive identification, as throug[h] trapping.

B. Potentially Occupied Habitat is described as:

1. All stands containing spruce or fir [USFS Region 8 timber typ[es] 06,07,17; Region 9 types 11,13,17,87]

or

2. All stands above [3300 feet for fuscus] [4500 feet for colorat[us]] containing hemlock or northern hardwoods in any combinatio[n] [USFS Region 8 types 05,08,81; Region 9 types 81,82,85,86,89]

and

3. Stands with at least some 10 inch dbh or larger trees present and at least partial canopy closure (e.g. in mixed conifer/hardwood stands a minimum basal area of 100 squar[e] feet per acre.

II. Occupied Habitat Management

A. The size of the occupied area is defined as all area within 1/2-mile of the trapping or identification site.

B. Within occupied areas, the following options are available:

1. Redesign the project to avoid the area.

2. Consult with a wildlife biologist <u>and</u> the USFWS to determine appropriate management measures.

III. Potentially Occupied Habitat Management

A. An evaluation (based on best information and professional judgment) must be performed by a wildlife biologist to determine one of two suitability classes (high or low) (see table, p. 52).

B. If the evaluation indicates low potential suitability, the area may be treated as unoccupied.

C. If the evaluation indicates high potential suitability, the following options are available:

1. Redesign the project to avoid the area.

2. Establish reasonable evidence that the identified area is unoccupied by G. sabrinus through the use of live trapping, and/or nesting boxes. Trapping and/or use of nesting boxes must follow procedures presented in Appendix B of this plan and must be supervised by a wildlife biologist.

3. Consult with a wildlife biologist to determine appropriate management measures.

IV. Management Measures

A. Some examples of appropriate management measures that may be recommended by a wildlife biologist are:

1. Save standing snags, trees with cavities, culls and down logs.

2. Retain spruce, fir, yellow birch and beech.

3. Plant spruce or encourage natural regeneration of spruce.

4. Avoid drainages, spring seeps, and moist areas.

5. Retain a certain stocking level of residual trees of a certain diameter and/or species to accomplish a specific objective.

6. Specify size and shape of treatment areas in order to accomplish a particular objective.

Factors to score in determining habitat suitability rating (high or low):

Factor	Suitability Rating	
	Low	High
temperature	warmer	cooler
humidity	low	high
soil moisture	low	high
presence of downed logs	few	many
lichen growth	sparse	abundant
presence of moss, fern, liverwort, Lycopodium groundcover	sparse	abundant

In addition to these factors, highly favorable sabrinus habitat would have at least some large trees, with elaborate branching systems dispersed throughout. These facilitate the squirrels' movements through their home range.

APPENDIX B
Recommended Procedures for trapping, handling, and use of nest boxes for Glaucomys sabrinus

BEFORE CONDUCTING ANY FIELD WORK WITH G. SABRINUS CONTACT APPROPRIATE STATE AND FEDERAL AGENCIES CONCERNING PERMIT REQUIREMENTS

1. Conduct trapping from spring through mid-autumn. Do not trap during extremely cold, wet or windy weather. Trapping success may be decreased on clear moonlit nights.

2. Use wire mesh live-traps of size appropriate for chipmunks. Metal box traps have proven ineffective for flying squirrel capture and could cause fatality.)

3. To increase capture success, put up feeding platforms where the traps will be placed, and "pre-bait" them for several nights before trapping (time permitting).

4. Set 20 to 40 traps at a minimum spacing of 50 m. in 1 or 2 transects through areas to be trapped. The number and spacing of the traps should be tailored to the area being trapped.

5. Secure traps to the ground or attach horizontally to large, mature trees at a height of about 6 feet. Be sure to flag or otherwise visibly mark trees with traps.

6. Place moss, leaves, etc. over traps, to break the outline and to provide some cover.

7. Insert a suitable bedding material (e.g. leaves and/or cotton batting) into the traps.

8. Bait traps with a peanut butter-oat and bacon grease-fruit (apple, prune) mixture.

9. Run traps 1 week to 10 days per area. If possible, each area should be trapped during more than one season.

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THE NORTHERN FLYING SQUIRREL (*GLAUCOMYS SABRINUS*): A CONSERVATION CHALLENGE

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The northern flying squirrel (*Glaucomys sabrinus*) has an extensive range in North America, inhabiting boreal, coniferous, and mixed forests of the northern United States and Canada and the slopes of the mountains of the east and west. Most undisturbed northern populations are apparently thriving, but those in the southern mountains are considered disjunct relicts occupying declining remnants of suitable habitat. It is clear that range contraction in the past has been associated with climate and vegetation change in the Pleistocene and the large-scale timber harvests of the early 20th century and that today a significant threat comes from forest practices and development. However, the major problem in dealing with conservation of this species is understanding its complex ecological position in its regional communities and the subtle as well as obvious influences of human activities. Thus, to preserve this species over its extensive range one will have to consider its various roles as a biological opportunist, an important prey item, a disperser of mycorrhizae, a potential victim of biological warfare, and a small, secretive glider especially vulnerable to anthropogenic and possible climatic changes in the size, arrangement, and quality of its home forests.

Key words: conservation, *Glaucomys*, heterothermy, northern flying squirrel, *Strongyloides*, truffles

The ability to develop an effective conservation strategy for a vulnerable species presupposes that one knows enough about the animal's biology and the potential threats in its environment to create a meaningful protection plan. In the case of the northern flying squirrel (*Glaucomys sabrinus*), both the acquisition of adequate data and their interpretation have been a challenge. Although concern for this species over much of its range in North America has stimulated a great number of studies over the past 20 years after a long period of limited interest, the listing of some populations as endangered fueled an intense search for that ''magic'' factor or formula that might explain its biology, guarantee its survival, and eliminate its interference with the human exploitation of its home forests. We still have much to learn. As a participant in a symposium held at the annual meeting of the American Society of Mammalogists in June 2006, I was asked to address the broad problem of flying squirrel conservation. Although this topic may be approached in a number of ways, I have chosen to attempt to provide an overview—with pertinent background and examples—of 2 interacting components of this conservation issue: the particular or salient ecological factors potentially critical to species survival; and those human activities, past and present, contributing to the species' vulnerability. I am looking for common denominators—factors important to varying degrees over the wide range and diverse habitats occupied by this species as well as special, regional threats, and I wish to raise questions about current ideas and assumptions. I maintain that in the field of northern flying squirrel conservation there may be no simple solutions but instead, within some common denominator of basic biology, an array of problems and possible management strategies dictated by regional variation in squirrel ecology and in the kinds of human influences.

With some chagrin I have recently realized that I started my studies of flying squirrels as a graduate student 43 years ago. Thus, I have decided to approach the topic partially from a personal point of view, stressing my own experiences as well as findings documented in the literature and derived from discussions with other researchers. Although my studies have included many other vertebrates over the years, I have been repeatedly drawn back to flying squirrel investigations as interesting questions and concerns have arisen. Along with a few other workers, I have become a ''marked man,'' because, over the past 25 years, inquiries have poured in from federal and state agencies, conservancies, consulting firms, and various business concerns. Everyone wants definitive information on flying squirrels in order to preserve rare or endangered squirrel populations, to find a rationale to protect threats to parks and especially significant forests, or to provide justifications for logging, road building, or development in or near the species' habitat. I would argue that the predicament of the northern

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flying squirrel is often too complicated and subtle for the pat answers these people request. Thus, I hope I can be forgiven for using my own experiences in the southern Appalachians as a starting point for a broad but not a definitive discussion of the species, linking these findings to much of the other North American research.

Background

The northern flying squirrel is not uniformly threatened over its wide range across the boreal forests of North America and the montane and mixed forests of the south-trending mountains of the east and west (Wells-Gosling and Heaney 1984). Except in areas under heavy settlement and large-scale clear-cutting, this species is holding its own rather well in much of the northern part of its range. Its vulnerability is most pronounced in the mountain areas at the southern margins of its range—the southern Appalachians, Sierra Nevada, and Rocky Mountains.

It is quite clear from historical studies of climate and vegetation that the species has experienced a number of range contractions in the past (Arbogast 1999, 2007; Arbogast et al. 2005; Weigl 1968). During times of glacial advance in the Pleistocene, boreal forests repeatedly extended as broad southern peninsulas along the eastern and western mountains and even down the Mississippi Valley (Davis 1976; Delcourt and Delcourt 1981, 1987). One can assume, based on a few fossil records, that the northern flying squirrel then occupied a much larger southern range. The retreat of the glaciers starting 18,000 years ago would have confined squirrels to narrower strips of land and isolated massifs along the Appalachians and western mountains, but much of its remaining habitat was probably quite adequate. Then, in the late 19th and early 20th century the catastrophic clear-cut logging of Appalachian forests took place. Huge areas were denuded and burned over a short period of time—a process repeated in the west somewhat later (Loeb et al. 2000). From what we can surmise from species' habitat requirements this was a critical time of range contraction, disjunction, and probably population extinction in the mountains. However, it is unlikely that the public or even the biologists of the time were at all aware of the plight of the flying squirrels. Many of the subspecies considered endangered or rare today were unknown. Hall (see Hall and Kelson 1959) described *Glaucomys sabrinus lucifugus* of Utah in 1934, Miller (1936) described *G. s. fuscus* of West Virginia in 1936, and Handley (1953) described *G. s. coloratus* of North Carolina and Tennessee in 1953. Although some populations from the west were described in the 1890s, many subspecies remained undiscovered until well into the 20th century (Hall and Kelson 1959; Howell 1918).

Starting in the early 1980s the northern flying squirrel became the object of intensive research, but much of this work concentrated on the more abundant and widely distributed northwestern forms, whereas the rare, relict, often inaccessible populations of the mountain ridges to the south received only limited attention in spite of the listing of some subspecies as endangered in 1985 (Weigl et al. 1999). Fortunately, recent studies have provided much more background information, and the pace of research has accelerated. However, we still have much to learn about the peculiarities of the ecology of this species and both the obvious and subtle effects of human activity. And that is why conservation of this species is such a challenge.

Northern Flying Squirrel Ecology

General

In the simplest terms one can describe the northern flying squirrel as a small, nocturnal, nonhibernating, gliding tree squirrel that occupies boreal conifer and mixed forests and uses both tree cavities and dreys for nesting (Smith 2007; Wells-Gosling and Heaney 1984). Contrary to suggestions that this squirrel is a narrow, boreal specialist, the northern flying squirrel is best described as a behaviorally plastic opportunist, capable of adjusting its biology to wide range of conditions. For example, it is quite capable of occupying deciduous and lower-elevation woodlands of the east and west, not just the spruce, fir, and other conifer forests usually cited in the literature (B. S. Arbogast, pers. comm.; Weigl et al. 2002; Weigl and Osgood 1974). Faced with cold temperatures, turbulent weather, and short periods of food limitation, the squirrel can become heterothermic, dropping its body temperature several degrees without becoming torpid (Bowen 1992). This enables it to wait out short intervals of bad weather and make the most of its body energy reserves. Unlike most squirrels, it does not depend on seeds and nuts, even when these are available (Brink 1965; Brink and Dean 1966; Hall 1991; Mitchell 2001; Thysell et al. 1997), but, although occasionally using mast, generally subsists on fungi, lichens, buds, berries, staminate cones, and animal material, none of which it appears to store. Even its reproductive biology is rather flexible. Although the squirrel commonly produces a litter in early spring, in some areas energy availability and condition of females lead either to reproductive failure or delay, with litters being observed late into the summer and even into October or December (Raphael 1984; Weigl et al. 1999; Witt 1991, 1992). Thus, compared to the smaller southern flying squirrel (*G. volans*) and most other North American tree squirrels, *G. sabrinus* possesses some unusual ecological characteristics, in keeping with the diversity of environmental conditions it must survive.

What salient features of the ecology of the northern flying squirrel need to be considered in developing conservation measures? Our knowledge of the species is still quite fragmentary, because relatively few long-term studies have been conducted (Carey et al. 1999; Cotton and Parker 2000a, 2000b; Fryxell et al. 1998; Lehmkuhl et al. 2006; Ransome and Sullivan 2002; Smith and Person 2007; Weigl et al. 1999). Most studies have been of short duration, confined to warmer months, or limited to surveys. Long-term, year-round investigations are rare. In addition, once some populations were listed as endangered in 1985 in the Appalachians and others were deemed vulnerable because of habitat modifications in the west, researchers avidly attempted to acquire and interpret new data in a quest for unitary and perhaps overly simple strategies to preserve these squirrels. Because the extensive literature on the genetics, biogeography, and ecology have been largely reviewed by Arbogast (2007) and Smith (2007) in this issue, I will concentrate on aspects of the squirrel's biology that appear essential to conservation of the species and then raise questions about the current state of our knowledge and interpretations. Some of my comments will be based on the literature, some on personal experiences.

Habitat

In reviewing the voluminous literature on the habitats utilized by northern flying squirrels, one cannot help but be impressed by certain common features as well as some regional variations that perhaps reinforce this perceived ''common denominator'' (Waters and Zabel 1995; see Smith 2007). Northern flying squirrels generally occupy boreal or north temperate conifer, mixed conifer–hardwood, and northern hardwood forests, as found in the northern United States and Canada, at various elevations of mountain regions, and in some narrow valleys subject to cold air drainage. These habitats support old-growth forest, communities with old-growth elements, or younger woodlands usually contiguous with such forest. Such areas are usually cool and moist, have cold winters, and possess a well-developed canopy, substantial ground cover, quantities of wet, dead, and downed wood, and often organic substrates. These conditions favor an abundance of snags, cavities, witches brooms, trees festooned with lichens and moss, and a diverse array of buds, berries, seeds, and fungi. In drier sites in the west, squirrels appear to select riparian areas where these cooler and wetter conditions prevail, and where there is easy access to drinking water (Meyer et al. 2005, 2007). In fact, Carey (1989, 1995) observed differences in population densities in Washington and Oregon that might be associated with moisture conditions in various forest types. Although one can point out variations in this ''typical'' habitat description, it is clear that the northern flying squirrel is versatile enough to prosper in a wide range of forest types as long as the above conditions occur in enough favorable patches and enough habitat is left undisturbed.

Although *G. sabrinus* may be a habitat opportunist and readily uses a diversity of potentially suitable forests, habitat is a major conservation problem, exacerbated by various controversial approaches to forest management. The ongoing harvest of old-growth forest, its replacement with plantations or regenerating stands, and the increasing fragmentation of much of the remaining habitat has alarmed some biologists concerned about this and other rare animal species (see Smith 2007). When rare species are declared endangered, as in the case of the northern flying squirrel, then economic forces exert tremendous pressure on researchers to develop definitive management plans that will protect the rare organisms, but also allow a return to timber harvest and development. Such is the case in Alaska (Smith and Person 2007) where the size, quality, and connectedness of planned reserves is an issue, in the Pacific Northwest where the debate over the importance of old growth versus successional forests to rare species has raged for years (Carey 1989, 1995; Lehmkuhl et al. 2006; Waters and Zabel 1995; Witt 1992; but see Ransome and Sullivan 1997, 2002, 2004; Rosenberg and Anthony 1992), and in the Sierra Nevada where thinning, fire, and harvesting may limit the size and quality of squirrel habitat (Meyer et al. 2005; Meyer and North 2005). Another example comes from the Appalachians where the currently endangered subspecies *G. s. fuscus* of West Virginia is a candidate for delisting. In the Appalachians northern flying squirrels are commonly found in older forests of spruce (*Picea rubens*), fir (*Abies fraseri*), beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), and yellow birch (*Betula alleghaniensis*), especially in the ecotones between conifers and hardwoods. However, throughout the east from Nova Scotia, Canada (Lavers 2004), to southern North Carolina (Weigl et al. 2002) the species is known to occupy hardwood habitats without spruce and fir. An array of studies have documented the squirrel's habitat diversity (Ford et al. 2004; Menzel et al. 2006; Payne et al. 1989; Stihler et al. 1987; United States Department of the Interior, Fish and Wildlife Service 2006; Urban 1988) pointing out the importance of hardwood and mixed forest habitats. *G. sabrinus* of West Virginia is more abundant and its populations more continuous than in most parts of the east. Many of the squirrels are caught in forests in which spruce is present, and this tree species supports one of the fungal genera (*Elaphomyces*) eaten by the squirrel (Loeb et al. 2000). Therefore, the United States Fish and Wildlife Service has decided that if forests containing spruce are protected in the national forests, the flying squirrel's preservation is insured, and it can be delisted, not to the ''threatened'' level but taken off the critical list entirely. The problems with this approach are many. First, it is not clear if there is any direct causality between the presence of flying squirrels and spruce. Both animal and plant may be responding independently to the same boreal conditions. Squirrels may nest in spruces occasionally and use them as one of many food sources, but there is no proof of any obligate relationship. Second, in more than 40 years of trapping and nestbox checking in various Appalachian habitats, I almost never captured animals in extensive, pure conifer stands, although telemetry revealed that they sporadically used them. Third, such a course of action fails to sufficiently protect the northern hardwood areas often used by *G. sabrinus*. Finally, the quality and connectedness of the proposed spruce-containing reserves, now and in the future, need careful study, especially in a region where timber harvest is an important part of the local economy. My main point is that economic pressures may at times influence how ecological information is interpreted resulting in overly simplistic solutions to a conservation and political issue.

Foods

One of the especially significant aspects of northern flying squirrel ecology and conservation is the direct link between the squirrel, its diet, and the perpetuation of its forest habitats. Years ago, McKeever (1960) noted high levels of fungi in the guts of California animals, and in 1965 I discovered that North Carolina squirrels were consuming large quantities of fungi and the staminate cones of fir (Weigl 1968). Subsequently, research in the Pacific Northwest documented the dependence of northern flying squirrels on the fruiting bodies of hypogeous, mycorrhizal fungi (truffles—Carey et al. 2002; Fogel and Trappe 1978; Lehmkuhl et al. 2004; Maser and Maser 1998; Maser et al. 1978, 1985, 1986; Meyer and North 2005; North et al. 1997; Pyare and Longland 2001b). The hyphae of these underground fungi form associations with tree roots, greatly increasing their surface area for the absorption of water and minerals at a small energy cost to the tree. Many tree species grow poorly or not at all without mycorrhizae. But spore dispersal to new seedlings and older trees is a problem for an underground fungus. Based on our study of the northern flying squirrel and another truffle eater, the fox squirrel (*Sciurus niger*—Weigl et al. 1989), and the work of Zabel and Waters (1997) and Pyare and Longland (2001a), the following scenario has taken shape. The truffle produces a fruiting body that gives off a chemical signal on ripening; this causes a squirrel to avidly excavate and devour the fungus (Secrest 1990). However, although the squirrel obtains energy and certain minerals (e.g., sodium and phosphorous) from these truffles, it is unable to digest the fungal spores, which are then dropped over the landscape for days or weeks afterward (Gamroth 1988). The resulting inoculation of young trees and spread of the fungus may thus have a marked impact on the perpetuation of the forest habitat on which the squirrel depends. Although *G. sabrinus* is not the only mycophagist in its home forest, it is one of the most mobile and spends much time on the ground during foraging (Bird and McCleneghan 2005; Loeb et al. 2000; Mitchell 2001; Zabel and Waters 1997). In any case, because of these food habits and their positive effect on the trees of its habitats, conservation of this species assumes a greater dimension and significance. In fact, many of the habitat models for *G. sabrinus* are now implicitly based on recognition of this squirrel, tree, and fungus symbiosis (Ford et al. 2004; Menzel et al. 2006; Odom et al. 2001; see Smith 2007).

Given the above account of the use of hypogeous fungi, it is important to link these and other foods to certain environmental factors. Truffles are the fruiting bodies of mycorrhizal fungi and appear to be most abundant in association with larger and older living trees, especially in moist, organic soils. The time course of fungal inoculation, growth, and maturation of sporocarps may vary in different forests, but old-growth conditions may be optimal. Epigeous fungi and lichens, which also are important foods, depend on abundance of dead wood and extensive tree surface areas, respectively, and, once again, cool, wet conditions. Although lichens and animal material such as insects and carrion may help support squirrels in the winter when most other foods are unavailable, some researchers also have found evidence for winter truffle use in habitats with frozen ground. Hackett and Pagels (2003) and Smith (2007) have data on the use of underground nests, but no one has reported underground foraging in winter. The other plant materials making up the squirrels diet—staminate cones, berries, beechnuts, and some seeds—are reflective of a preference for boreal habitats and old-growth conditions but also are indicative of an opportunistic species that is not limited to truffles and that might utilize additional foods.

Demographic Considerations

In spite of the spectacular increase in northern flying squirrel studies, we have surprisingly little information on the species' life history and population biology. Most studies have been dedicated to particular questions such as home range, relative density, foods, and habitat associations. Longer-term studies (e.g., Carey et al. 1999; Fryxell et al. 1998; Smith et al. 2004, 2005; Smith and Nichols 2003; Weigl et al. 1999) have begun to fill in some gaps in our knowledge, but we know very little about most population parameters and long-term temporal and spatial trends.

Smith and Person 2007 have recently reviewed much of the demography of the species and raised questions about the distribution and stability of populations. The picture of *G. sabrinus* that is developing is of a relatively long-lived (4–7 years) species with a low reproductive rate for a small mammal. In the western part of the range of *G. sabrinus*, flying squirrels appear to be more abundant than in the east and more continuous in their distribution within the old-growth forests that they commonly occupy. However, most workers report lower densities in managed or successional stands. In the east, populations often occur in distinct patches, often kilometers away from other groups in spite of what seems to be suitable intervening habitat (Weigl et al. 1999, 2002). Also in the east, population size appears to be highly variable. In some years, squirrels will be abundant in an area; in other years the populations are low or nonexistent. Have the animals died out or moved? No answer is available, but population fluctuations have been noted by other researchers (Fryxell et al. 1998). In spite of the meager data from recaptures, it is clear that at least some of the squirrels missing in intervening sampling sessions show up again months or years later (Weigl et al. 1999).

Examination of telemetry data from throughout North America suggests that home-range size is associated with habitat quality and food resources (Smith 2007). Home ranges from 2 to 60 ha have been reported. Our own work and that of others have revealed that squirrels have relatively small core home ranges (3–15 ha) that vary somewhat with sex and season, but that many individuals will display bouts of extensive linear travel, in some cases more than a kilometer, that involve both outward movement and return (Menzel et al. 2006; Weigl et al. 1999). There is some evidence that this long-distance travel is associated with a search for foods and possibly mates (Weigl et al. 1999). Such forays may affect home-range estimates if data are taken at wide time intervals. The important question here relates to the use of space by the species. If populations in a locality can fluctuate widely in numbers, have a distinctly patchy distribution in fairly uniform forest, and consist of individuals that can cover spectacular distances, it is possible that northern flying squirrels may use and thus require much larger expanses of suitable habitat than is commonly acknowledged if they are going to survive in many parts of their range. Both habitat size and connectedness assume great significance under these conditions.

Smith and Person 2007 have recently provided an intriguing example of space use that may partially relate to the preceding discussion. Working in Alaska in undisturbed habitat, they investigated populations in prime old-growth forest and adjacent groups in a wet, mixed muskeg and forest landscape. Examination of the demographic data suggested that there was a dynamic source–sink situation governing these populations. The muskeg areas were not maintaining viable squirrel populations in a steady state, but were the beneficiary of constant migration of animals from the better forest habitats. To what extent high mobility, source–sink conditions, and metapopulation distributions of squirrels are a common phenomenon is unknown, but this may be worth investigating in areas with old-growth forest adjacent to human-modified habitats. The squirrel populations reported from cutover and regenerating areas may be more variable because they are not self-perpetuating. Certainly the status of populations in West Virginia, the Sierra Nevada, and parts of the Pacific Northwest should be evaluated with this possibility in mind.

Other Species of Animals

The fate of northern flying squirrels may be closely linked to the presence of other animal species—predators, competitors, and parasites—that are in turn often of particular concern to wildlife biologists and conservationists.

Predators.—Smith (2007), Carey et al. (1992), and Weigl et al. (1999) have described some of the potential predators of the flying squirrel, but 2 in particular may be of interest in different parts of the range. Over the past 20 years it has become clear that the northern spotted owl (*Strix occidentalis*), an endangered and much celebrated species of western forests, is especially dependent on the northern flying squirrel as a prey item (Carey et al. 1992). The owl seems to thrive in extensive old-growth forests or in habitats with old-growth elements where the squirrels are most abundant (Carey 1995; Carey et al. 1999). The size and condition of the habitat ideal for supporting both the flying squirrel and the owl have been the focus of ferocious debate (Carey et al. 1992; Ransome and Sullivan 2002; Rosenberg and Anthony 1992). Old-growth forests in the west are becoming smaller in size and increasingly fragmented, but often are viewed as the economic salvation for a timber industry that is worried about an endangered species restricting the exploitation of remaining tracts. For the squirrel the issue of habitat quality, size, and connectedness is of great importance and has been the focus of several studies. Conservation of squirrel and owl thus seems inextricably linked, but doubtless shall remain a source of intense political and economic controversy.

In the eastern United States another rare animal is periodically associated with the issue of protection of *G. sabrinus*. Every few years, wildlife biologists consider the reintroduction of the fisher (*Martes pennanti*) to the southern Appalachians; this species was known to exist in the region in the recent past. In most areas fishers can probably coexist with northern flying squirrels without problems. But in small habitat islands of the southern Appalachians with few squirrels and limited alternate prey items, a predator such as the fisher might kill off these relict populations. Although there have been no introductions of fishers in areas with isolated flying squirrel populations, this idea resurfaces frequently (R. Powell, pers. comm.) and will require the careful attention of wildlife agencies in the region.

Competitors.—Smith et al. (2004, 2007) have suggested that the biology of *G. sabrinus* in the Pacific Northwest may be different from that in Alaska and the east because of the abundance of other small mammals in western forests. This diversity of sympatric rodents might then produce a greater degree of den-site and food specialization in response to direct and diffuse completion. In reality, we have little information on resource competition between northern flying squirrels and other mammals. Although red squirrels (*Tamiasciurus hudsonicus*) and Douglas squirrels (*T. douglasii*) are often mentioned as possible competitors, there is not much evidence of any severe interaction. Flying squirrels may pilfer food from red squirrel middens and the 2 species may both use cavities for nesting sites and fungi for food, but the very different overall diets of these squirrels and their nocturnal–diurnal activity separation may minimize interactions, especially in good habitat. In many years of trapping both species, I was always surprised to find that the best years for capturing northern flying squirrels also were the best for red squirrels.

The southern flying squirrel (*G. volans*) often has been considered a major competitor (Weigl 1968, 1978). Both species are nocturnal gliders that use tree cavities for dens and both may consume fungi, insects, and plant parts. Although experimental studies suggested that *G. volans* was the more active and aggressive in interactions, especially around nests (Weigl 1978), habitat preferences, diets, and climatic tolerances of the 2 species (Bowen 1992; Bowman et al. 2005) suggest only limited competition. In fact, except in the north, the 2 species usually show limited and unstable sympatry. Thus, except for the diffuse interactions suggested by Smith et al. (2005) in the west, and a few instances of resource overlap, there is little evidence that competition per se is a significant factor in the conservation of the northern flying squirrel.

Parasites.—A particularly intricate relationship between squirrel ecology and conservation grew out of some unusual discoveries in the southern Appalachians. In the 1960s I had set out to study the interaction of *G. sabrinus* and *G. volans* in the Appalachians as a model system for evaluating aspects of competition theory (Weigl 1968). Northern flying squirrels were exceedingly rare, but after several months of trapping I eventually captured enough for the experimental parts of my study. Colonies of both species were then housed in large outdoor aviaries in North Carolina. The 1st spring saw the demise of almost all of the *G. sabrinus* except those kept in the laboratory, whereas the *G. volans* seemed to thrive in an adjacent cage. With the help of 2 veterinarians and a former zoo pathologist, I narrowed down the cause of this massive die-off to an infection by the nematode *Strongyloides robustus*. *S. robustus* has a life cycle like that of the famous hookworms (*Necator* and *Ancylostoma*): embryonated eggs released with animal feces hatch and develop into infective larvae in the substrate; these penetrate the skin of a host, are carried to the lungs where they break through to the lumen, are swallowed, and finally lodge in the intestine doing marked physical and nutritional damage (Weigl 1968; Weigl et al. 1999). The parasite is most common in warmer climates where it has been reported to cause marked pathology in wild species (Davidson 1975). Once the cause of the affliction of the captive *G. sabrinus* was determined, other wild populations of squirrels were checked. All of the captive *G. volans* in my colony were parasitized (and were probably the source of the infection in the *G. sabrinus*), but had suffered no ill effects. In fact, all populations of *G. volans* studied in subsequent years carried this parasite. On the other hand, *S. robustus* could not be found in any of the *G. sabrinus* captured on the Appalachian peaks during the remaining years of the study. In the 1980s the federal listing of the Appalachian subspecies *G. s. coloratus* prompted a new 5-year study of the northern flying squirrel over a wide area of the North Carolina and Tennessee mountains. *G. volans* now also appeared intermittently in some of the capture sites of *G. sabrinus*, although there was never any stable sympatry of the 2 species (Weigl et al. 1999). *G. sabrinus* now supported varying intensities of parasite infection, and in the summer months there appeared to be some correlation between parasite loads and the condition of the animals (Weigl et al. 1999). We eventually cultured the parasite through its life cycle in the laboratory and determined its cold sensitivity (Wetzel and Weigl 1994) and its ability to be transferred by contact with contaminated nest material or soil substrates. Based on all the data to-date and some additional studies by Pauli et al. (2004) and Sparks (2005), I would suggest the following scenario. The cold, high-elevation or northern forests occupied by *G. sabrinus* only intermittently can support *S. robustus* because of the sensitivity of the infective larvae to cold. When *G. sabrinus* moves down into the more climatically moderate forests at lower elevations or when infected *G. volans* invade the upper slopes during the summer months along paths of human-modified habitat, the 2 species come into contact, especially by using the same tree cavities or feeding areas (Hackett and Pagels 2003), and *S. robustus* is then transferred. Even if the northern flying squirrels are not killed by the parasite, its effects may be sufficiently debilitating to put the species at a disadvantage. It is interesting that only in the colder parts of the range of *G. volans*—the Great Lakes area, northern New England, Ontario, and Nova Scotia—does one get reports of some degree of sympatry of the 2 flying squirrel species (J. Bowman, pers. comm.; Lavers 2004; Pauli et al. 2004). Why then doesn't *G. volans* take over the high-elevation refuges or northern habitats of *G. sabrinus*? The answer probably lies in sensitivity to cold of *G. volans*, its dependence on stored nuts and seeds for winter survival (Bowman et al. 2005; Doby 1984), and the virtual absence of these resources in most habitats of *G. sabrinus*. In summary, *G. volans* may possess a kind of biological weapon that at least in the southern and central part of its range, can prevent the persistence and spread of *G. sabrinus* (Barbehenn 1969; Haldane 1949; Hatcher et al. 2006; Price et al. 1988; P. D. Weigl, in litt.). It has been argued recently that the loss of genetic heterogeneity in the increasingly isolated, high-elevation populations of *G. sabrinus* of the east may make the species even more susceptible to parasite and other infections (Sparks 2005). What will happen if warming climatic conditions favor invasion of higher peaks and northern habitats by *G. volans* is thus an open question in considerations of species persistence.

Genetics

In many parts of the range of the northern flying squirrel, one can reasonably argue that the species is an island inhabitant, subject to most of the constraints that afflict other such populations (Brown 1971, 1978; MacArthur and Wilson 1967). Whether occupying real islands off the coast of Alaska; widely scattered habitats of the San Jacintos, Sierra Nevada, Rocky Mountains, and perhaps the Black Hills; or the upper elevations of the southern Appalachians, the species often occurs in small, disjunct populations, relicts of broader ranges in the late Pleistocene. The genetics of these populations have received intensive study over the last 10 years (Arbogast 1999, 2007; Arbogast et al. 2005; Bidlack and Cook 2001; Browne et al. 1999; Sparks 2005; Wartell 2005; A. Wartell, in litt.). Genetic structuring, private alleles, and loss of heterozygosity have been detected in many populations, most likely as a result of reduced population size, isolation, inbreeding, bottlenecks, and other drift effects. Although inbreeding tolerance and the replacement of alleles in time by mutation (Sparks 2005) might alleviate the plight of some groups, the loss of genetic diversity is usually seen as a potential threat, especially in changing environments. The persistence of reasonably large and interconnected populations thus appears to be critical to the species survival, and that means sufficiently large habitat reserves and the maintenance of forested corridors. Such a conservation solution might work if the environmental status quo can be maintained. However, in the face of continued forest destruction, drought cycles, El Niño effects, and the still largely unknown impacts of global climate change, the reduction of available habitat and of corridors could well spell the regional demise of this species from both a loss of genetic variability and the loss of viable places to live.

The Impact of Human Activity

Habitat Size and Quality

So far I have emphasized some of the complexities of northern flying squirrel ecology and its implications for species conservation. However, it is clear that the really major threats to these squirrel's persistence come from human activities, especially in areas of small disjunct populations such as those on islands or at the southern extension of the range. Clearcutting, development, or anything that destroys extensive tracts of habitat will have obvious harmful effects. The size of the remaining forest habitat and its condition then becomes critical to survival. One has only to fly over parts of the Rocky Mountains, Sierra Nevada, and Cascades or along the Appalachians to appreciate the scope of forest destruction and roadway construction in national and privately owned forests. And landscape modification is not the only concern. Successional and regenerating communities require considerable time to develop into habitats of sufficient quality to support flying squirrels. Using demographic models, Smith and

Person (2007) have questioned the adequacy of the size of planned reserves in Alaska; Carey and others (Carey 1995; Carey et al. 1999) have provided evidence that the 2nd-growth landscapes of the Pacific Northwest do not always have the same capacity as old growth for supporting flying squirrels. In the Sierra Nevada, thinning and controlled burning may have adverse impacts on the canopy and organic material on the ground, respectively. Finally, some 2nd-growth stands may well appear to support healthy densities of squirrels, but, in reality, are population sinks for migrants from neighboring old-growth habitats and thus may not permanently maintain viable populations (Smith and Person 2007). Only long-term studies can provide the conclusive data on the suitability of these special or successional areas. The small disjunct squirrel populations of the central and southern Appalachians appear particularly vulnerable to any further modification or reduction of their habitats.

Given the above problem of loss of quality habitat, one needs to recognize 2 major forces that can aggravate this threat. One is economic and political—the demand for forest products and recreation venues, for local and regional employment, and for tax revenues and investment returns. These factors are of overwhelming significance, but are beyond the scope of this paper. The other force—climate change—is more intangible. A warming climate could cause the retreat of some tree species and communities to higher latitudes and cause the substantial reduction or elimination of boreal communities on mountains. Change in the composition and the position of communities might be especially dire in areas already modified by other human influences. Thus, the persistence of northern flying squirrels in the already-disturbed forests of West Virginia could be more tenuous than many have thought during a period of global warming. In addition to modifying community composition and distribution, climate change may have another major impact. A recent paper by Westerling et al. (2006) has documented a link between progressive climate warming and changes in the phenology, desiccation, and fire frequency in western forests. Thus, climatic warming may not only cause modifications of forest distributions, but also their complete annihilation by fire. It is likely that the desiccation observed by Westerling et al. (2006) would also have a marked impact on the moisture-requiring staple foods (fungi and lichens) of flying squirrels.

Habitat Connectedness

Along with habitat size and quality, habitat connectedness assumes an important role in species preservation. The extent of unsuitable terrain between high-quality habitat and the absence of wooded corridors could be major factors in regional survival. Frequently, the greater the reduction of contiguous forest, the wider the barriers to dispersal. Such fragmentation of flying squirrel distributions could destroy the viability of metapopulation-structured groups of squirrels, and the resulting small isolates then would be susceptible to the genetic problems mentioned earlier.

The impact of barriers on movements of flying squirrels needs further study, especially the effects of the proliferation of roadways through quality habitats. One example of barrier effects comes from the southern Appalachians. A 3-year study of an extravagant economic development scheme in the North Carolina–Tennessee mountains called the Cherohala Skyway revealed such unexpected impacts (Weigl et al. 2002). Clearly, a 2-lane scenic road removes a quantity of habitat, but, of greater significance, it also can act as a barrier to dispersal to different parts of the forest. Although *G. sabrinus* is an able glider and is known to cover distances along the ground, it is unable to cross wide, exposed roadways, especially the kind of blast-and-fill rights-of-way commonly cut into the sides of mountains. In 2 years of telemetry and trapping, no squirrel was observed to have crossed the Cherohala Skyway. The resulting range fragmentation may doom this southernmost population. In addition to barrier formation, there are 2 more-subtle impacts from a roadway. One impact was detected in the winter when snow permitted the identification of mammals moving on or along the roadway. It was obvious that various predators—bobcats, coyotes, and foxes—used the roadway as patrol routes when hunting and might easily catch any small mammals on the road. Hawks and owls also hunted over the road. Thus, one can easily see that such a right-of-way is both a physical barrier and a site of increased mortality. Another effect of roadways or similar corridors is the modification of adjacent vegetation or other habitat conditions in ways that favor the invasion of potential predators, competitors, or pathogens. In the case of *G. sabrinus*, strips of oak, cherry, and other hardwood species in disturbed areas along roadways provide foods for *G. volans* and favor its invasion of high-elevation habitats, and the transfer of *Strongyloides* to *G. sabrinus*. Thus, linear disturbances of a certain width and severity are a potential source of species fragmentation and possibly increased deleterious species interactions. The impact of roads, systems of ski trails, ridge-top wind farms, recreational vistas, and other types of habitat subdivision need careful evaluation in the future—much more then they have received to-date.

Pathogens, Pests, Pollutants, and People

Another anthropogenic factor threatening northern flying squirrels is the introduction of plant pathogens, insect pests, and industrial contaminants into squirrel habitats. In the southern Appalachians, the high-elevation conifer forests have been decimated by an adelgid insect (*Adelges piceae*) that kills Fraser fir (*Abies fraseri*), a valuable timber and Christmas tree species and a source of food and habitat for northern flying squirrels (Amman 1966; Amman and Speers 1965). The staminate cones of fir and spruce are important foods for flying squirrels in the spring when they are eaten in vast quantities. Interestingly, both field and experimental studies suggest that the essential oils from these foods suppress gut parasites such as *Strongyloides* (Weigl et al. 1999). The loss of Fraser fir then would remove a source of food (truffles, staminate cones, and possibly seed), den sites, and a possible natural medicine. In any case the adelgid killing firs, a new adelgid now destroying hemlocks, the impact of pine bark beetles in some parts of the west, and the effect of acid precipitation on vegetation and soils all represent potential threats to flying squirrels.

The last intrusion mentioned in the heading of this section of the text—people—usually goes unmentioned. One of the major effects of building of roads through prime habitat is the provision of access to lands for private and commercial development. The state or federal government builds a road, and nearby landowners demand the right to connect in order to develop their forest property. During an era of explosive interest in living in natural environments or in 2nd-home ownership, the demand for newly accessible forest land is intense and is often fueled by the economic aspirations of neighboring municipalities. A short trip on the Blue Ridge Parkway in the Appalachians reveals the result of this process. The end result is the loss and fragmentation of habitat and possibly a loss of flying squirrels. Thus, the inclusion of people as a factor along with pathogens, pests, and pollutants may indeed be appropriate.

Summary

In the past 25 years the northern flying squirrel has come under increasing scrutiny as new studies have been initiated, papers published, and various agencies alerted to its status and ecological significance. Because of physical, logistical, and economic difficulties associated with long-term research in remote and often rugged areas, our knowledge of this species is still fragmentary, especially in the southern Rocky Mountains, parts of the Sierra Nevada, the Black Hills, and the northeastern United States. Enough is known now to form a picture of the species' ecology and those aspects of its biology that may affect its preservation. In 2 cases, the northern flying squirrel makes a positive contribution to the forests it occupies. Throughout its range its use and dispersal of mycorrhizal fungi—both hypogeous and epigeous—make it an integral part of a squirrel–fungus–tree mutualism that may well help maintain the very forests needed for its survival. In the northwestern United States and western Canada, the flying squirrel is a critical food item for the endangered spotted owl. Thus, if its habitat is protected and the squirrel is permitted to flourish, the owl has a greater probability of survival.

In spite of the fact that the northern flying squirrel is something of an ecological opportunist, versatile enough to occupy several forest types, consume a number of foods, and reproduce when conditions permit, certain of its characteristics potentially increase its vulnerability. Its dependence on fungi and lichens during much of the year confine it to a certain array of old-growth, boreal forests with cool, moist climates and abundant dead wood and organic soils. The phenology of fungi, particularly the locality and timing of sporocarp production, may require the exploitation of a multitude of widely spaced, ephemeral patches and thus the use at times of extensive home ranges or reliance on long-distance travel. In short, the area needed to support these animals may be larger than our short-term telemetry studies have indicated. And although its diet and tolerance of cold conditions facilitate survival in habitats with severe climates, the low caloric density of much of its diet may be a factor in its relatively low metabolic and reproductive rates (McNab 1986).

The influence of others animals in the environment of the northern flying squirrel needs further study. In no part of its undisturbed range does it seem adversely affected by predators or competitors. Perhaps only in human-modified areas do these markedly assume importance. In the southern and central parts of the eastern United States the possibility that the nematode *S. robustus*, carried by the southern flying squirrel, harms the northern species is unresolved. However, the obvious ability of northern flying squirrels to occupy lowland, deciduous habitats in the absence of the smaller species, their confinement to high elevations when *G. volans* is present, and the instability of populations in contact zones argue for some kind of interaction. In Ontario, Nova Scotia, and northern Pennsylvania, the 2 species have been found in the same nest boxes (J. Bowman, pers. comm.; A. Lauers, pers. comm.; M. Steele, pers. comm.), but these are areas that are climatically unfavorable for the parasite. Thus, in part of the range of the northern flying squirrel a parasite-mediated interaction may be operating. Clearly more research on this topic is needed.

Although there is abundant evidence of the effect of small population size and isolation on the genetic diversity of northern flying squirrel populations, there is at present no evidence of a direct link between loss of genetic diversity and survival. The isolation of populations may occur naturally because of climatic responses of forest communities, but, more likely today, it is caused—or least aggravated—by human activity. We may never know when genetic impoverishment is a major or just a contributing factor to a population's disappearance.

All of the above ecological aspects of the biology of the northern flying squirrel may have varying effects on the perpetuation of populations in different parts of the range. When one adds the human component, the probability of survival can change spectacularly. Human influences on habitat size, quality, and connectedness are most likely the main threats to the species throughout its range. These critical factors in turn are the products not only of direct habitat destruction and modification, but indirect effects such introduced pathogens, pests, and contaminants and the slow, inexorable pressure of climate change. Survival of the species *G. sabrinus* is certainly critically dependent on an understanding of the species' ecology, but, even more important, an awareness of the impact of human activity on this ecology throughout its range.

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A Review of the Scientific Literature on Riparian Buffer Width, Extent and Vegetation

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A REVIEW OF THE SCIENTIFIC LITERATURE ON RIPARIAN BUFFER WIDTH, EXTENT AND VEGETATION

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for the

Office of Public Service & Outreach
Institute of Ecology
University of Georgia

Revised Version • March 5, 1999

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The Office of Public Service and Outreach at the Institute of Ecology provides scientific and legal expertise to the citizens of Georgia in the development of policies and practices to protect our natural heritage. The goals of the office are to:

- Develop and implement a research agenda to meet community needs;
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- Build capacity for service learning in the sciences by providing students an opportunity to apply skills learned in the traditional classroom setting to pressing community concerns and problems;
- Support high quality science education in K-12 schools by providing programs for students and instructional support and training for teachers;
- Increase awareness of the importance of addressing environmental issues proactively within the university community and the greater community.

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For more information about the Office of Public Service and Outreach at the Institute of Ecology, please contact Laurie Fowler at 706-542-3948.

EXECUTIVE SUMMARY

Many local governments in Georgia are developing riparian buffer protection plans and ordinances without the benefit of scientifically-based guidelines. To address this problem, over 140 articles and books were reviewed to establish a legally-defensible basis for determining riparian buffer width, extent and vegetation. This document presents the results of this review and proposes several simple formulae for buffer delineation that can be applied on a municipal or county-wide scale.

Sediment is the worst pollutant in many streams and rivers. Scientific research has shown that vegetative buffers are effective at trapping sediment from runoff and at reducing channel erosion. Studies have yielded a range of recommendations for buffer widths; buffers as narrow as 4.6 m (15 ft) have proven fairly effective in the short term, although wider buffers provide greater sediment control, especially on steeper slopes. Long-term studies suggest the need for much wider buffers. It appears that a 30 m (100 ft) buffer is sufficiently wide to trap sediments under most circumstances, although buffers should be extended for steeper slopes. An absolute minimum width would be 9 m (30 ft). To be most effective, buffers must extend along all streams, including intermittent and ephemeral channels. Buffers must be augmented by limits on impervious surfaces and strictly enforced on-site sediment controls. Both grassed and forested buffers are effective at trapping sediment, although forested buffers provide other benefits as well.

Buffers are short-term sinks for phosphorus, but over the long term their effectiveness is limited. In many cases phosphorus is attached to sediment or organic matter, so buffers sufficiently wide to control sediment should also provide adequate short-term phosphorus control. However, long-term management of phosphorus requires effective on-site management of its sources. Buffers can provide very good control of nitrogen, include nitrate. The widths necessary for reducing nitrate concentrations vary based on local hydrology, soil factors, slope and other variables. In most cases 30 m (100 ft) buffers should provide good control, and 15 m (50 ft) buffers should be sufficient under many conditions. It is especially important to preserve wetlands, which are sites of high denitrification activity.

To maintain aquatic habitat, the literature indicates that 10-30 m (35-100 ft) native forested riparian buffers should be preserved or restored along all streams. This will provide stream temperature control and inputs of large woody debris and other organic matter necessary for aquatic organisms. While narrow buffers offer considerable habitat benefits to many species, protecting diverse terrestrial riparian wildlife communities requires some buffers of at least 100 meters (300 feet). To provide optimal habitat, native forest vegetation should be maintained or restored in all buffers.

A review of existing models for buffer width and effectiveness showed that none are appropriate for county-level buffer protection. Models were found to be either too data-intensive to be practical or else lacked verification and calibration. Potential variables for use in a buffer width formula were considered. Buffer slope and the presence of wetlands were determined to be the most important and useful factors in determining buffer width.

Three options for buffer guidelines were proposed. All are defensible given the scientific literature. The first provides the greatest level of protection for stream corridors, including good control of sediment and other contaminants, maintenance of quality aquatic habitat, and some minimal terrestrial wildlife habitat. The second option should also provide good protection under most circumstances, although severe storms, floods, or poor management of contaminant sources could more easily overwhelm the buffer.

Option One:

- Base width: 100 ft (30.5 m) plus 2 ft (0.61 m) per 1% of slope.
- Extend to edge of floodplain.
- Include adjacent wetlands. The buffer width is extended by the width of the wetlands, which guarantees that the entire wetland and an additional buffer are protected.

- Existing impervious surfaces in the riparian zone do not count toward buffer width (i.e., the width is extended by the width of the impervious surface, just as for wetlands).
- Slopes over 25% do not count toward the width.
- The buffer applies to all perennial and intermittent streams. These may

Option Two:

The same as Option One, except:

- Base width is 50 ft (15.2 m) plus 2 ft (0.61 m) per 1% of slope.
- Entire floodplain is not necessarily included in buffer, although potential sources of severe contamination should be excluded from the floodplain.
- Ephemeral streams are not included; affected streams are those that appear on US Geological Survey 1:24,000 topographic quadrangles. Alternatively, buffer can be applied to all perennial streams plus all intermittent streams of second order or larger

Option Three:

- Fixed buffer width of 100 ft.
- The buffer applies to all streams that appear on US Geological Survey 1:24,000 topographic quadrangles or, alternatively, all perennial streams plus all intermittent streams of second order or larger (as for Option Two).

For all options, buffer vegetation should consist of native forest. Restoration should be conducted when necessary and possible.

All major sources of contamination should be excluded from the buffer. These include construction resulting in major land disturbance, impervious surfaces, logging roads, mining activities, septic tank drain fields, agricultural fields, waste disposal sites, livestock, and clear cutting of forests. Application of pesticides and fertilizer should also be prohibited, except as may be needed for buffer restoration.

All of the buffer options described above will provide some habitat for many terrestrial wildlife species. To provide habitat for forest interior species, at least some riparian tracts of at least 300 ft width should also be preserved. Identification of these areas should be part of an overall, county-wide wildlife protection plan.

For riparian buffers to be most effective, some related issues must also be addressed. These include reducing impervious surfaces, managing pollutants on-site, and minimizing buffer gaps.

Contents

I. Background and Introduction

Riparian buffers have gained wide acceptance as tools for protecting water quality, maintaining wildlife habitat and providing other benefits to people and the environment (Lowrance 1998, USEPA 1998). Today in Georgia, as in many other states, local governments are developing programs to protect riparian buffers. Laws such as the Georgia Planning Act and the Mountain and River Corridor Protection Act give counties and municipalities strong incentives to incorporate aquatic resource protection into their plans and ordinances. However, scientifically-based guidelines for local riparian buffer ordinances are not readily available. The minimum standards issued by the Department of Natural Resources' Environmental Protection Division (EPD) are not based on current scientific research and do not provide a strong level of resource protection. Many local governments are interested in developing effective, comprehensive riparian buffer regulations, but fear that without solid scientific support, such ordinances would not be legally defensible.

The purpose of this document is to provide a scientific foundation for riparian buffer ordinances established by local governments in Georgia. To achieve this goal more than 140 articles and books were reviewed with an eye toward determining the optimal width, extent (i.e., which streams are protected) and vegetation (e.g., forest or grass) of riparian buffers. This task is challenging due to the lack of research in certain geographic regions. Although a large number of riparian buffer studies have been conducted in the Georgia Coastal Plain (see Figure 1), there has been very little research specific to the physiographic provinces of North Georgia (Piedmont, Blue Ridge, Valley and Ridge) or to urban and suburban areas. Nevertheless, it is apparent in reviewing the literature that there are general trends which cut across geographic boundaries. Based on current research, it is possible to develop defensible guidelines for determining riparian buffer width, extent and vegetation that are applicable to much of Georgia and beyond. Naturally, these recommendations will not be as accurate as those supplied by data-intensive models applied on a site-by-site basis (such as the REMM model developed by Richard Lowrance and colleagues). However, these guidelines have the virtue of being simple enough to be incorporated into a county or municipal ordinance.

The guidelines proposed in this document should be viewed as a reasonable interpretation of the best available scientific research. If additional riparian buffer studies are conducted in North Georgia, urban areas, and other neglected regions, it may be possible to refine the recommendations. However, this in no way means that the current state of our knowledge is insufficient to develop good policy guidelines and implement effective buffer ordinances. As Lowrance et al noted in 1997:

> *"Research is sometimes applied to broad-scale environmental issues with inadequate knowledge or incomplete understanding. Public policies to encourage or require landscape management techniques such as riparian (streamside) management will often need to proceed with best professional judgment decisions based on incomplete understanding."*

Local officials and natural resource managers are making decisions on riparian buffers today. The scientific community would be remiss if it failed to provide these decision makers with the best available information.

To ensure that this review has covered the most relevant research and has made reasonable conclusions, other members of the scientific community were asked to review its findings. These reviewers included:

- Richard Lowrance, Ph.D., USDA-Agricultural Research Service
- David Correll, Ph.D., Smithsonian Environmental Research Center
- Cathy Pringle, Ph.D., University of Georgia
- Laurie Fowler, J.D., L.L.M., University of Georgia
- Judy Meyer, Ph.D., University of Georgia
- Ronald Bjorkland, University of Georgia
- Michael Paul, University of Georgia

Figure 1. Physiographic Provinces of Georgia.

Although there has been a signficant amount of riparian buffer research in the Georgia Coastal Plain, there has been much less research conducted in the other physiographic provinces (Keys et al 1995, as modified by J. P. Schmidt)

The corrections, additions and changes made by these reviewers, as well as the comments and suggestions other people have made on an earlier draft of this document, have been incorporated into this revised version.

Acknowledgments

This review would have been impossible without the assistance and support of numerous people, and much of the credit for this project lies with these individuals. **Laurie Fowler** suggested the idea of the project and helped to guide its progress, while **Cathy Pringle** made certain it remained focused and manageable. **Ronald Bjorkland** made extremely thorough and thoughtful comments on a draft version of this document. **Richard Lowrance** and **David Correll**, two of the leading researchers on riparian buffers, were generous enough to provide expert criticism and made important corrections. Many others have lent their expertise and have patiently taken time to answer my questions. Some of their personal comments and expert opinions are included in this review.

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- Rebecca Sharitz, University of Georgia
- Chris Skelton, Georgia Department of Natural Resources
- Brian Toth, J.P. Schmidt and Karen Payne, NARSAL, University of Georgia
- David Walters, University of Georgia
- The rest of the Pringle and Fowler Labs and my other colleagues at the Institute of Ecology

Scope of Review

There are literally hundreds of articles and dozens of books written on the subject of riparian buffer zones. The 1997 version of David Correll's riparian bibliography (Correll 1997), which is limited to works on nutrients, sediments and toxic contaminants, lists 522 citations. John Van Deventer published a bibliography in 1992 of an astounding 3252 articles that relate to riparian research and management, though most of the literature cited does not directly address *buffer* zones (Deventer 1992). Given the volume of literature available, it was apparent from the beginning that this review would have to be limited in some ways. Priority was given to:

- articles which specifically deal with the issues of riparian buffer width, extent and vegetation
- previous literature reviews
- articles focused on Georgia and the Piedmont
- seminal articles in the field

- recent articles (1990-1998) especially those not included in prior literature reviews
- articles from refereed journals (although several good government documents and other works from the “grey literature” are included)

Over 140 sources are included in this review.

Why Another Literature Review?

As of this writing there exist several excellent literature reviews on riparian buffer zones. The U.S. Army Corps of Engineers New England Division conducted a 1991 literature review for the State of Vermont called *Buffer Strips for Riparian Zone Management* that is similar in scope and purpose to this document. It differs from most other reviews in that it covers virtually all of the functions of riparian buffers, including instream and riparian wildlife habitat as well as water quality functions. It also shares this document’s focus on buffer width, although it ultimately makes no width recommendations. Another thorough and useful review is Desbonnet et al’s 1994 *Vegetated Buffers in the Coastal Zone: A Summary Review and Bibliography.* Despite its title, this work reviews research from many regions, not just the coastal zone. The review is rather weak on wildlife habitat studies, however, since it predates much of the best literature. Lowrance and a team of riparian buffer researchers collaborated on a 1997 paper that synthesizes research on sediment and nutrient retention and presents guidelines for buffers in the Chesapeake Bay watershed. Other useful reviews include Clinnick et al 1985, Muscutt et al 1993, Osborne and Kovacic 1993, Castelle et al 1994, Fennesy and Cronk 1997 and Bjorkland (unpublished).

As useful as these previous works were, a new review was necessary to include recent studies, to consider the full range of buffer functions (nutrient reduction, wildlife habitat, etc.), and to address the primary issue of concern: determining the optimal width, extent and vegetation for buffer zones in Georgia. This works relies heavily on previous reviews, although in most cases the original research articles were consulted as well.

Background on Riparian Zones

Definitions

Before proceeding it will be helpful to establish definitions for some key concepts. The word **riparian** is especially subject to confusion, and currently there appears to be no universally accepted definition of the term. One of the better definitions comes from Lowrance et al (1985): “‘Riparian ecosystems’ are the complex assemblage of organisms and their environment existing adjacent to and near flowing water.” Malanson (1991) offers an attractively simple definition: “the ecosystems adjacent to the river.” Bjorkland (unpublished) provides a thorough review of published definitions for the term. Some of these definitions even use “riparian” to refer to the edges of bodies of water other than streams and rivers. This broader usage reflects the original, legal definition of the term, which referred to land adjoining any water body (David Correll, pers. com.). In this review the term is used in two ways: (1) to refer to the “natural” riparian area (Figure 2), the zone along streams and rivers that in its undisturbed state has a floral and faunal community distinct from surrounding upland areas, and (2) in the most general sense to refer to the zone along streams and rivers which might benefit from some type of protection. Stream corridor and river corridor will sometimes be used synonymously with riparian zone.

A riparian zone that is afforded some degree of protection is a **riparian buffer zone.** The word “buffer” is used because one of the functions of the protected area is to buffer the stream from the impact of human land use activities, such as farming and construction. Numerous other terms are also used to refer to this protected zone, both in this document and in the scientific literature: riparian management zone, riparian forested buffer strip, stream buffer zone and protected stream corridor are all taken to be synonymous with riparian buffer zone for the purposes of this review. Within this document the term is also frequently shortened to buffer zone, riparian buffer or simply buffer. Note that in some fields, especially agricultural research, the term buffer is applied in a more general sense to a variety of **conservation practices.** The terms vegetated buffer strip and vegetated filter strip (VFS) are often used to refer to strips of grass or other plants installed between or below agricultural fields to reduce erosion and trap contaminants. For the sake of clarity, these terms are not used in this review.

Significance of Riparian Zones

Riparian zones are a type of ecotone, or boundary between ecosystems. Like many other ecotones, riparian buffer zones are exceptionally rich in biodiversity (Odum 1978, Gregory et al 1991, Malanson 1993, Naiman et al 1993). Naiman et al (1988) noted that ecotones can display a greater variation in characteristics than either of the systems they connect; rather than being averages of the two systems, they are something unique. For this reason alone riparian zones can be considered valuable. In addition, however, riparian zones perform a range of functions with economic and social value to people:

- Trapping /removing sediment from runoff
- Stabilizing streambanks and reducing channel erosion
- Trapping/removing phosphorus, nitrogen, and other nutrients that can lead to eutrophication of aquatic ecosystems
- Trapping/removing other contaminants, such as pesticides
- Storing flood waters, thereby decreasing damage to property
- Maintaining habitat for fish and other aquatic organisms by moderating water temperatures and providing woody debris
- Providing habitat for terrestrial organisms
- Improving the aesthetics of stream corridors (which can increase property values)
- Offering recreational and educational opportunities

(based on Schueler 1995a, Malanson 1993)

Because they maintain all of these services, riparian buffers can be thought of as a "conservation bargain": preserving a relatively narrow strip of land along streams and rivers— land that is frequently unsuitable for other uses— can help maintain good water quality, provide habitat for wildlife, protect people and buildings against flood waters, and extend the life of reservoirs. "Vegetative buffer programs, however, are rarely developed to fully consider the multiple benefits and uses that they offer to resource managers and to the general public" (Desbonnet et al 1994). Often, buffer programs are developed for a single goal, such as preventing erosion and sedimentation. However important this goal may be, programs with such a narrow focus inevitably undervalue buffers (and riparian zones in general) and may lack popular support if this goal is not met. On the other hand, programs that promote the multiple functions of buffers are likely to enjoy a wider and stronger base of support, especially when people recognize the economic benefits they can provide. It is hoped that this document will encourage the establishment of multifunctional riparian buffer protection programs.

That said, it must be acknowledged that certain buffer functions are given a higher priority than others by local governments. Water quality and aquatic habitat functions are generally considered of greatest importance. Of slightly less concern are terrestrial wildlife habitat, the floodwater storage functions of the riparian buffer, recreation and aesthetic values. The organization of this review reflects this hierarchy. The next two sections review literature on the water quality functions of riparian buffers. Section four reviews aquatic habitat functions. Section five considers the literature on buffers as terrestrial habitat, along with other functions not yet discussed. Finally, section six develops guidelines for buffer width, extent and vegetation, taking into consideration various factors and reviewing other models of buffer function. Section seven is a discussion of important related issues, such as impervious surface limits and riparian buffer crossings.

A note on measurements: Riparian buffer widths given in this review are for one side of the stream measured from the bank. Therefore, a 50 ft (15 m) buffer on a 25 ft (7.6 m) stream would actually create a corridor 125 ft (38 m) wide. Measurements are given in metric or English units, according to how they were reported in the literature, with the conversion in parentheses. Buffer recommendations are made first in English units because legislation in Georgia generally uses this system.

II. Sediment

In terms of volume, sediment is the largest pollutant of streams and rivers (Cooper 1993). In much of Georgia, sediment levels in streams have historically been high due to agricultural activities. The decline in row crop acreage and improvements in erosion control practices have led to decreased agricultural sedimentation, but in urbanizing parts of the state these gains have been offset by sedimentation from construction (Kundell and Rasmussen 1995).

Effects

Excess amounts of sediment can have numerous deleterious effects on water quality and stream biota. For a full discussion of this topic, refer to Waters 1995 and Wood and Armitage 1997. The following brief list summarizes the major sediment effects.

- Sediment in municipal water is harmful to humans and to industrial processes.
- Sediment deposited on stream beds reduces habitat for fish and for the invertebrates that many fish consume.
- Suspended sediment reduces light transmittance, decreasing algal production.
- High concentrations of fine suspended sediments cause direct mortality for many fish.
- Suspended sediments reduce the abundance of filter-feeding organisms, including mollusks and some arthropods.
- Sedimentation reduces the capacity and the useful life of reservoirs.

Sediment must be filtered from municipal water supplies at considerable cost. The greater the turbidity levels in water, the higher the price of treatment (Kundell and Rasmussen 1995). Note that both suspended sediment (sometimes approximated by turbidity measurements) and benthic sediment have detrimental biological effects, and that benthic sediment can become resuspended during high flows. Certain fish are more responsive to sediments than others. Although many species of fish found in Georgia's waters are sediment tolerant, many of the threatened and endangered species, such as darters, tend to be very sensitive to siltation (Kundell and Rasmussen 1995, Freeman and Barnes 1996, Barnes et al 1997, Burkhead et al 1997). The many endangered species of native mussels may be the most sensitive organisms of all (Morris and Corkum 1996).

Sources

Sediment in streams either comes from runoff from upland sources or from the channel itself. Upland sources include row crop agricultural

Figure 2. View of a River with an Intact Riparian Zone.

This is the Etowah River in Cherokee County, GA.

fields, exposed earth at construction sites, and logging roads, for example. Channel-derived sediment may result from the erosion of poorly stabilized banks and from scouring of the stream bed. Livestock watering in streams can contribute significantly to bank destabilization and erosion (Waters 1992). Note that much channel-derived sediment may originally have been upland sediment that is temporarily stored in the streambed or riparian zone (Trimble 1970, Wood and Armitage 1997).

Construction Sites

In urban and urbanizing areas, construction is likely to be the major source of sediment (see Figure 3). Streams draining urban areas often have higher sediment loads than those in agricultural watersheds (Crawford and Lenat 1989) and certainly have higher rates than forested areas (Wahl et al 1997). A recent report by the U.S. Geological Survey found that urban streams in Georgia are the most degraded (Frick et al 1998).

Mining

Various forms of mining can produce severe sedimentation (Waters 1992, Burkhead et al 1997). Gravel dredging can be considered a form of mining which is especially harmful because it takes place within the river itself. This has direct negative effects on stream organisms and increases downstream turbidity, as local residents and canoeists have observed (Bob James, pers. com.). In addition, dredging may release sediment-bound contaminants (Burruss Institute 1998) and contributes to stream downcutting, both at the site and upstream (Pringle 1997).

Agricultural Sources

According to Waters (1992), row-crop agriculture and livestock are the top two sources of sediment nationwide. Row crop agriculture is no longer widely practiced in much of North Georgia, and in South Georgia row-crop agriculture tends to be concentrated in upland areas (Frick et al 1998). However, cattle are raised throughout Georgia (GA Department of Agriculture 1997) and frequently are permitted direct access to streams and rivers, resulting in bank erosion (pers. obs.; see Figure 4).

Forestry

Streams in forested areas are not necessarily pristine. Improperly stabilized logging roads can yield over 350 tons of sediment per acre per year (Kundell and Rasmussen 1995). Some of the first research on riparian buffers was initiated to determine logging road setbacks (e.g., Trimble and Sartz 1957). The Georgia Forestry Commission advocates Best Management Practices (BMPs) for logging operations, but compliance is voluntary. The most recent BMPs have placed limits on logging in "streamside management zones" (buffers), which vary in width from 20-100 ft (6-30 m) depending on slope and stream type (Georgia Forestry Commission 1999).

Historic Sedimentation

Many streams and rivers in Georgia have experienced a long history of sedimentation. Throughout the 1800s and up until the 1940s, massive soil erosion from cotton farming and other forms of row crop agriculture led to severe sedimentation of streams all across the Georgia

Figure 3. Impacts of Development.

This riparian zone has been stripped of vegetation in preparation for the construction of subdivisions. A properly enforced riparian buffer ordinance could prevent this type of problem.

Piedmont (Trimble 1970, Kundell and Rasmussen 1995). Some areas, such as the upper Chattahoochee and Etowah Rivers, were also impacted by hydraulic gold mining, when "entire hillsides" were washed into streams (Glenn 1911), leading to rapid sedimentation and aggradation of rivers and floodplains (Leigh 1994). The channels of many streams were entirely filled with sediment over time. For example, the bed of the Etowah River at Canton, GA, rose 4.8 ft (1.46 m) between 1890 and 1949 (Walters unpublished). With the decline of gold mining and agriculture in the region, as well as the adoption of better soil conservation practices, sedimentation rates decreased and many Piedmont streams experienced downcutting, as channels carved deeper and wider into the loose beds of sand (Trimble 1970, Burke 1996). There is evidence, however, that as of the 1980s sedimentation is again increasing in some Piedmont rivers, perhaps as a result of construction (Burruss Institute 1998, Walters unpublished).

Figure 4. Bank Erosion from Livestock Intrusion.

Livestock intrusion into the riparian zone results in stream bank erosion and water contamination.

It appears likely that sediment now stored in stream channels continues to cause high turbidity during storms (Trimble 1970; Rhett Jackson, pers. com.). Sediment in the larger Piedmont streams and rivers may also increase as sand from tributaries migrates downstream. Riparian buffers will probably little effect on this sediment source (except as they contribute to bank stability), but they are essential in preventing additional degradation to water quality, especially in smaller tributaries.

Literature Review

Riparian buffers can reduce stream sedimentation in six ways:

1) by displacing sediment-producing activities away from flowing water (setbacks)
2) by trapping terrestrial sediments in surface runoff
3) by reducing the velocity of sediment-bearing storm flows, allowing sediments to settle out of water and be deposited on land (this includes sediments previously suspended in the river that are borne into the riparian buffer during floods)
4) by stabilizing streambanks, preventing channel erosion
5) by moderating stream flow during floods, reducing bed scour, and
6) by contributing large woody debris (snags) to streams; these can trap considerable sediment, at least temporarily

(adapted in part from US ACE 1991)

Functions one, two and three are primarily concerned with preventing terrestrial sediment from reaching the water. Functions four and five involve reducing channel erosion. This review of sediment-related literature is divided into two subsections corresponding to these two major topics. The literature on large woody debris is reviewed separately in the section regarding in-stream habitat protection.

Sediment in Surface Runoff

Numerous studies have documented the effectiveness of buffers in trapping sediment transported by surface runoff. The challenge lies in determining the necessary width of the buffer.

Width

One of the greatest challenges in trying to develop buffer width recommendations is that most studies only examined one or a few buffer widths. Fennesy and Cronk (1997) noted this problem:

> *"One problem in assessing minimum widths necessary to protect adjacent surface water is that many studies that make recommendations regarding the minimum width necessary have arrived at the figure as a byproduct of sampling design rather than deriving it experimentally."*

Nevertheless, from the research that exists it is evident that there is a positive correlation between a buffer's width and its ability to trap sediments. In their 1994 review, Desbonnet et al determined that increasing buffer width by a factor of 3.5 provides a 10% improvement in sediment removal. According to the reviewers, the most efficient width of vegetated buffers for sediment removal is 25 m (82 ft). For total suspended solids, buffer widths need to increase by a factor of 3.0 for a 10% increase in removal efficiency, and 60 m (197 ft) wide buffers provide the greatest efficiency. It is important to note that Desbonnet et al based this relationship on a composite of data from studies conducted with various methods at different location. It may not be appropriate to compare such study results. It is more illuminating to examine data from studies that compared multiple width buffers in the same location under the same study conditions. Six studies (Young et al 1980; Peterjohn and Correll 1984; Magette et al 1987, 1989; Dillaha et al 1988, 1989) have examined the effectiveness of buffers of two widths in trapping total suspended solids (TSS). In every case, buffer effectiveness

Author	Width (m)	% Slope	% Removal of TSS
Dillaha et al (1988)	4.6	11	87
Dillaha et al (1988)	4.6	16	76
Dillaha et al (1988)	9.1	11	95
Dillaha et al (1988)	9.1	16	88
Dillaha et al (1989)	4.6	11	86
Dillaha et al (1989)	4.6	16	53
Dillaha et al (1989)	9.1	11	98
Dillaha et al (1989)	9.1	16	70
Magette et al (1989)	4.6	3.5	66
Magette et al (1989)	9.1	3.5	82
Peterjohn & Correll (1984)	19	5	90
Peterjohn & Correll (1984)	60	5	94
Young et al (1980)	21.3	4	75-81
Young et al (1980)	27.4	4	66-93

Table 1. Riparian Buffer Width, Slope and TSS Removal Rates.

The ability of riparian buffers to trap suspended solids is positively correlated with width and negatively correlated with slope.

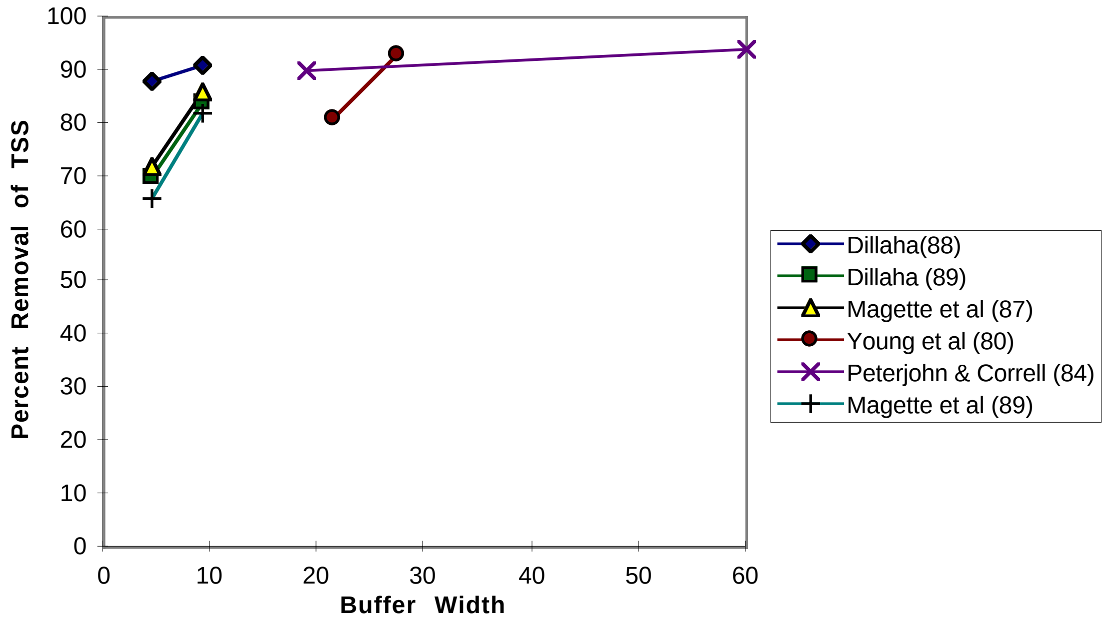


Figure 5. Removal of TSS by Buffers of Different Widths.

The ability of a riparian buffer to remove total suspended solids (TSS) from runoff is a function of the buffer's width. Note that with the exception of Peterjohn and Correll (1984) these are short-term studies; long-term studies have suggested that much wider buffers are necessary. This figure is only intended to convey the consistent relationship of buffer width and effectiveness.

increased with buffer width, although the relationship varied. Table 1 and Figure 5 summarize the results of these studies [Data from Magette et al (1987) which appear in Figure 5 were taken from Desbonnet et al (1993) because the original document was not readily available].

In a series of studies using orchardgrass buffers downslope from a simulated feedlot, Dillaha et al (1988) reported average TSS reductions of 81% for a 4.6 m (15 ft) buffer and 91% for a 9.1 m (30 ft) buffer. Dillaha et al (1989) later repeated the study using buffers of the same width and vegetation below fertilized bare cropland. This time they found average sediment reductions of 70% and 84% for buffers of 4.6 m and 9.1 m width, respectively. Magette et al (1989) conducted a similar study with grassed buffers of 4.6 m and 9.1 m downslope from plots to which they added liquid nitrogen or chicken waste. They found average sediment reductions of 66% and 82%, respectively.

Coyne et al (1994) also conducted a study of similar design, although they only used strips of 9 m (30 ft) width and conducted only one rainfall simulation rather than a series. The researchers added poultry waste to a test plot and found that the grass buffers trapped 99% of sediment. Young et al (1980) tested the efficiency of buffer strips of corn, orchard grass, oats and sorghum/sudangrass at reducing surface runoff from feedlots. They found that buffers of 21.34 m (70 ft) reduced total suspended solids by an average of 78%, while 27.43 m (90 ft) wide buffers reduced TSS by an average of 93%. Buffer slope averaged four percent.

Peterjohn and Correll (1984) found that a 50 m (164 ft) riparian buffer in an agricultural catchment in the Mid-Atlantic Coastal Plain trapped 94% of suspended sediment that entered. Ninety percent was trapped in the first 19 m (62 ft). Average slope of the buffer was about five percent.

Only a few researchers have found buffer width to be unimportant. Daniels and Gilliam (1996) found that 6 m (20 ft) wide grassed buffers and 13 m (43 ft) or 18 m (59 ft) wide combination forest/grassed buffers all reduced sediments by about 80%. However, the wider buffers included a farm vehicle access road which provided an additional source of sediment, so comparisons are not valid. Gilliam (1994) mentions that a "narrow" buffer in the Piedmont was found to trap 90% of sediment. Rabeni and Smale (1996) suggest that width of buffer may not be as important as other, qualitative characteristics, such as whether or not the topography can maintain sheet flow.

Most of the studies described above were short-term. There is significant evidence from long-term analyses that wider buffers are necessary to maintain sediment control. Lowrance et al (1986) used sediment budgets to calculate that a low-gradient riparian buffer ecosystem in the Georgia Coastal Plain trapped large amounts of sediment (35-52 Mg/ha per year) between 1880 and 1979. Later studies by Lowrance et al (1988) based on cesium-137 concentrations yielded a much higher reduction rate of 256 Mg/ha per year for the period between 1964 and 1985. The researchers found that sediments from agricultural fields were deposited throughout the riparian forest. The greatest amount (depth) of transported sediment was found 30 m (98 ft) inside the forest and the greatest cesium signal occurred 80 m (262 ft) into the forest. The results are confounded slightly by the higher affinity of cesium-137 to clay particles, which are transported farther than sand and silt (possibly leading to a higher signal deeper in the buffer), and deposition of sediment within the riparian zone by floodwaters from the stream. A similar Cs-137 study by Cooper et al (1988) in the North Carolina Coastal Plain reached similar conclusions. The riparian buffer trapped 84-90% of the sediment eroded from agricultural fields, although nearly 50% was transported more than 100 m (328 ft) into the buffer. Slopes ranged from 0-20%. These two studies suggest that although riparian zones are efficient sediment traps, the width required for long-term retention may be substantially more than is indicated by short-term experiments. Buffers of 30-100 m (98-328 ft) or more might be necessary.

Davies and Nelson (1994) found that buffers can be highly effective in reducing sedimentation to streams in logged forests, and buffer width is the determining factor. "All effects of logging were dependent on buffer strip width and were not significantly affected by [buffer] slope, soil erodibility or time (over one to five years) since logging." The authors found that a 30 m (98 ft) buffer was necessary to prevent impacts. These recommendations are in agreement with a 1985 review of the use of riparian buffers to mitigate the impacts of logging on forest streams (Clinnick 1985). One study cited in that review found that "streams with buffers of at least 30 m width exhibited similar channel stability and biological diversity to unlogged streams, whereas streams with buffers less than 30 m showed a range of effects similar to those found where no stream protection was provided" (Erman et al 1977, as cited in Clinnick 1985).

The sediment trapping efficiency of buffers can be expected to vary based on slope, soil infiltration rate, and other factors. Slope may be the best studied of these relationships. Dillaha et al (1988, 1989) found that as buffer slope increased from 11% to 16%, sediment removal efficiency declined by 7-38% (See Figure 6). The most thorough investigations of the relationship between buffer width and slope have been conducted by forestry researchers. Trimble and Sartz (1957) examined erosion of logging roads in the Hubbard Brook Experimental Forest in New Hampshire to determine how far roads should be set back from streams. They suggested a simple formula:

25 ft + (2.0 ft)(% slope).

For municipal watersheds where water quality is of very high importance, the setback should be doubled. Trimble and Sartz' formula was the basis of a Forest Service standard for many years. Swift (1986) proposed an alternative formula based on work in the Nantahala National Forest in western North Carolina. He found that when brush barriers are employed below a road, erosion is reduced dramatically. He proposed a buffer width formula of

32 ft + (0.40 ft)(% slope).

If barriers are not used the buffer width should be increased to:

43 ft + (1.39 ft)(% slope) (Swift 1986).

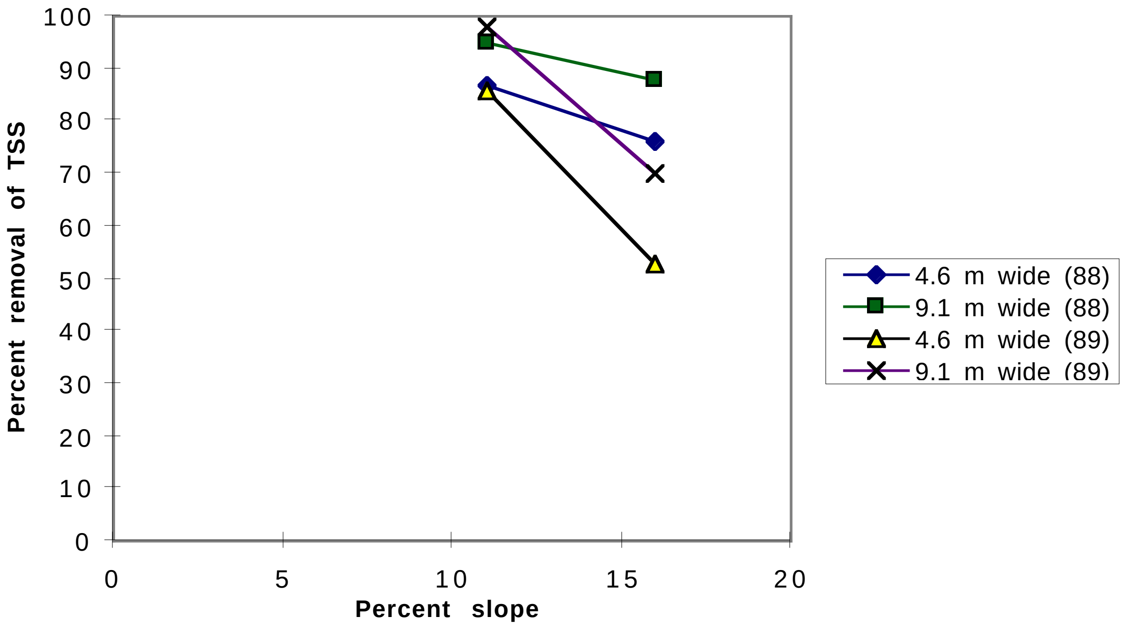


Figure 4. Removal of TSS by Buffers of Different Slopes.

Studies by Dillaha et al revealed an inverse relationship between buffer slope and reduction of total suspended solids (TSS). Slope is an important variable to use in determining riparian buffer width.

Swift only measured coarse sediment in his study, not silt and clay, which are transported much further through a buffer. This suggests that his buffer recommendations are insufficiently wide.

Lowrance et al (1997) made some generalizations about buffer effectiveness in different physiographic provinces. They noted that buffer effectiveness has been well established in the Coastal Plain, where much research has been conducted. For the Piedmont and Valley and Ridge provinces, they predicted sediment reductions of 50-90%, although they did not discuss widths necessary to achieve this reduction. The Blue Ridge province was not discussed in their review. Daniels and Gilliam (1996) suggested that the high level of runoff from Piedmont fields makes buffers valuable. They also pointed out, however, that steeper slopes and lower soil infiltration rates may make Piedmont buffers less effective in terms of trapping efficiency than buffers in the Coastal Plain.

Extent

It is very important that buffers be continuous along streams (Rabeni and Smale 1996). Gaps, crossings or other breaks in the riparian buffer allow direct access of surface flow to the stream, compromising the effectiveness of the system. The problem of buffer gaps is discussed further in Section VI.

Riparian buffers are especially important along the smaller headwater streams which make up the majority of stream miles in any basin (Osborne and Kovavic 1993, Binford and Buchenau 1993, Hubbard and Lowrance 1994, Lowrance et al 1997). These streams have the most land-water interaction and have the most opportunities to accept and transport sediment. “Protecting greenways along low-order streams may offer the greatest benefits for the stream network as a whole” (Binford and Buchenau 1993).

Ideally, therefore, a system of riparian buffers should protect all streams and rivers, regardless of size. Even ephemeral streams should be protected, since these waterways can carry appreciable flow and sediment during storms. Although such universal protection will generally not be feasible, buffer ordinances should be written to protect as many stream miles as possible— at least all perennial streams, as well as intermittent streams of second order or larger.

Vegetation

The studies reviewed above have found that for purposes of trapping sediment, both grass and forested buffers are effective. Grass buffers, although more likely to be inundated by exceptionally high levels of sediment, are useful for maintaining sheet flow and preventing rill and gully erosion. In sum, however, forested buffers have other advantages (discussed in later sections) which recommend them over grass in most cases. A combination of grass and forested buffers has been advocated by many researchers (e.g. Welsch 1991, Lowrance et al 1997) and represents a reasonable compromise.

Limitations

Buffers are most effective when uniform, sheet flow through the buffer is maintained; they are less effective in stopping sediment transported by concentrated or channelized flow (Karr and Schlosser 1977, Dillaha et al 1989, Osborne and Kovacic 1993, Daniels and Gilliam 1996). When these conditions occur, riparian buffers cannot slow the flow sufficiently to allow infiltration of water into the soil, although some sediment may still be trapped by vegetation. Clay particles are unlikely to be trapped because they form colloids in solution. Jordan et al (1993) reported that sediment *increased* across a 60 m (197 ft) wide riparian buffer in the Delmarva Peninsula because of rill erosion. Daniels and Gilliam (1996) noted that ephemeral channels in the North Carolina Piedmont were ineffective sediment traps during high-flow events. They recommended dispersing the flow from these channels through a riparian area rather than allowing them to empty directly into a perennial stream. Sheet flow can be encouraged by the use of level spreaders and other structural techniques. Welsch (1991) recommended planting a strip of grass 20 ft (6.1 m) wide at the outer edge of a riparian buffer to help convert concentrated flow to dispersed sheet flow.

It is possible for buffer vegetation to be inundated with sediments and decline in effectiveness, although under normal conditions vegetation should be able to grow through the sediment (Dillaha et al 1989). Sediment can also accumulate to the point where it forms a levee that blocks the flow of water from the slope to the stream (Dillaha et al 1989). Flow then runs parallel to this berm until it reaches a low spot, at which time it crosses into the stream in concentrated flow. Buffers on agricultural land with very high erosion may require regular maintenance to remain effective and should always be used in conjunction with other erosion control methods (Barling 1994). The importance of on-site sediment control is discussed further in a later section.

Channel Erosion

In a long-term study between 1983 and 1993, Stanley Trimble found that in San Diego Creek in suburban Los Angeles, two thirds of stream sediment resulted from channel erosion. He concluded that "stream channel erosion can be the major source of sediment in urbanizing watersheds, with deleterious downstream effects" (Trimble 1997). Clinnick's 1994 review also noted the importance of channel erosion, citing a 1990 study by Grissinger et al that suggested that "better than 80% of the total sediment yield for Goodwin Creek in northern Mississippi originates as channel and gully erosion." Likewise, Rabeni and Smale (1995), Cooper et al (1993) and Lowrance et al (1985) found that the channel can be a significant source of sediment.

One of the most important roles of protected riparian buffers is to stabilize banks. A study (Beeson and Doyle 1995) of 748 stream bends found that 67% of bends without vegetation suffered erosion during a storm, while only 14% of bends with vegetation were eroded. Non-vegetated bends were more than 30 times as likely to suffer exceptionally severe erosion as fully vegetated bends. The authors concluded, unsurprisingly, that "the denser and more complete the vegetation around a bend, generally the more effective it is in reducing erosion" (Beeson and Doyle 1995). Barling and Moore (1994) note that buffers can prevent the formation of rills and gullies in riparian areas that are otherwise highly susceptible to erosion.

Bank stabilization will not be effective if the underlying causes of channel erosion are not addressed. The major problem in urban and suburban areas is increased storm flows due to elevated surface runoff from impervious surfaces. This is discussed in more detail in Section VI. In rural areas, livestock that graze on banks and enter streams are a direct source of severe channel erosion (Figure 4). A solution is to fence the livestock out (Waters 1995) and provide alternate means of watering the animals. Use of offstream watering tanks is the preferred method, but a narrow, stabilized stream access point can also be considered as a compromise (Cohen et al 1987).

Stream channelization contributes to channel erosion by increasing stream power, leading to incision (Karr and Schlosser 1977, Malanson 1993). Formerly, stream channelization was encouraged by government agencies such as the Soil Conservation Service (now the Natural Resources Conservation Service). However, channelization is now recognized as a short term solution to drainage problems that results in long-term damage to streams and agricultural fields. In a channelized stream in Illinois, flood waters from one storm eroded as much as 1150 tons of soil from a single bank in 1982 (Roseboom and Russell 1985). In 1978 Karr and Schlosser (1978) noted that "money spent on preventing sediments from entering streams will have minimum return value in improving the quality of biota, if present channelization practices continue to destroy the habitat of stream organisms." Channelization and gravel mining can also lead to upstream impacts, resulting in headward erosion and channel downcutting (Pringle 1997).

Width

Few studies have attempted to correlate stream bank stability with riparian buffer width. Common sense suggests that relatively narrow vegetative buffers should be effective in the short term (USACE 1991). As long as banks are stabilized and damaging activities are kept away from the channel, width of the riparian buffer would not appear to be a major factor in preventing bank erosion. However, it is important to recognize that some erosion is inevitable and stream channels *will* migrate laterally, which could eventually move the stream outside the protected area. Therefore, the buffer zone should be wide enough to permit channel migration. To allow for all possible migration would require a buffer the width of the active (100-year) floodplain (Rhett Jackson, pers. com.), but a narrower buffer may still permit migration over a shorter period of time. As a general rule, buffer widths sufficient for other purposes should also be sufficient to prevent bank erosion and allow reasonable stream migration.

Extent

All channels, regardless of stream size and frequency of flow, can be subject to erosion if not properly stabilized. In their 1985 review, Clinnick et al (1985) note:

> *"During storm events it is often the ephemeral elements of the stream system that act as a source of surface flow to permanent streams (Hewlett and Hibbert 1967). The prevention of sediment accession to streams thus relies primarily on protection of these ephemeral elements."*

Daniels and Gilliam (1986) found that forested ephemeral channels were temporary sediment sinks during dry seasons but were sources of sediment during storm events. Binford and Buchenau (1993) note that such gullies and tributaries naturally have dense growth and should have excellent capacity for sediment and nutrient retention. It is essential to maintain these ephemeral channels in a vegetated condition to allow them to slow water flow, trap sediment and to prevent their serving as sediment sources (Cooper et al 1987, Binford and Buchenau 1993). Clinnick et al (1985) advocate a minimum of a 20 m wide buffer on ephemeral channels. This may not be practical in many situations, but at the least, the banks and even the bed of such channels should be vegetated and livestock intrusion should be minimized.

Vegetation

To be effective, bank vegetation should have a good, deep root structure which holds soil. Shields et al (1995) tested different configurations of vegetation and structural controls in stabilizing banks. They found that native woody species, especially willow, are best adapted to recolonizing and stabilizing banks. The authors noted that the persistent exotic vine kudzu may be the most serious barrier to vegetation restoration because it can outcompete native vegetation. Other restoration ecologists believe that kudzu and certain other exotics may still have a role in streambank restoration because they can provide good root structure (Carl Jordan, pers. com.).

Artificial methods of streambank stabilization, such as applying riprap or encasing the channel in cement, may be effective in reducing bank erosion on site but will increase erosion downstream and have negative impacts on other stream functions. Artificially stabilized banks lack the habitat benefits of forested banks and can be expensive to build and maintain. Overall, the negative consequences of artificial bank stabilization generally outweigh the benefits.

Summary and Recommendations

Riparian buffers are generally very effective at trapping sediment in surface runoff and at reducing channel erosion. Studies have yielded a range of recommendations for buffer widths; buffers as narrow as 4.6 m (15 ft) have proven fairly effective in the short term, although wider buffers provide greater sediment control, especially on steeper slopes. Long-term studies suggest the need for wider buffers. It appears that a 30 m (100 ft) buffer is sufficiently wide to trap sediments under most circumstances. This is consistent with the review of Castelle et al (1993), which found that buffers must be 30 m wide to maintain a healthy biota. This width may be extended to account for factors such as steep slopes and land uses that yield excessive erosion. It is possible to also make the case for a narrower width, although the long-term effectiveness of such a buffer would be questionable. An absolute minimum width would be 9 m (30 ft). For maximum effectiveness, buffers must extend along all streams, including intermittent and ephemeral segments. The effectiveness of a network of buffers is directly related to its extent; governments that do not apply buffers to certain classes of streams should be aware that such exemptions reduce benefits substantially. Buffers need to be augmented by limits on impervious surfaces and strictly enforced on-site sediment controls (discussed in Section VI).

Riparian buffers should be viewed as an essential component of a comprehensive, performance-based approach to sediment reduction. Periodic testing of instream turbidity should be conducted to assess the effectiveness of sediment control measures. Kundell and Rasmussen (1995) recommend a maximum instream standard of 25 NTU (nephelometric turbidity units), measured at the end of a designated segment (not below site of impact). Regular monitoring and enforcement of this standard will help ensure the effectiveness of riparian buffers and other sediment-control practices.

III. Nutrients and Other Contaminants

A. Phosphorus

Effects

Phosphorus has long been implicated in the eutrophication (overfertilization) of lakes. Eutrophication unbalances an aquatic ecosystem, leading to massive blooms of some types of algae. When these algae die off and decay, oxygen is consumed, sometimes to the point where fish and other animals cannot survive. Eutrophication can lead to other harmful effects, such as the blooms of the dinoflagellate *Pfiesteria* documented in East Coast estuaries in recent years. *Pfiesteria* has been linked to massive fish kills and releases toxins that are poisonous to humans (Burkholder 1998). In at least some Georgia lakes and reservoirs, such as Lake Allatoona, phosphorus is the most problematic nutrient and possibly the greatest pollutant overall (Burruss Institute 1998).

Sources

Potential nonpoint sources of phosphorus include:

- Fertilizers applied to agricultural fields
- Animal wastes from concentrated animal feeding operations (CAFOs) spread onto fields
- Septic drain fields
- Leaking sewer pipes
- Fertilizers applied to lawns

The relative impact of each of these sources will vary across the state. Cropland fertilization is probably not a major problem in most of north Georgia, but land-applied chicken waste from CAFOs is likely to be a significant source of pollution in some watersheds (Burruss Institute 1998, Frick et al 1998). There are hundreds of millions of chickens raised in North Georgia (Bachtel and Boatright 1996). In suburban areas septic drain fields are probably more significant, and sewer lines, especially those that run through stream valleys, can also be important phosphorus sources. The impact of lawn fertilization is unclear but potentially quite high. In 1984, the EPA estimated that Americans apply nearly a million tons of chemical fertilizers to their lawns per year. According to surveys, about 70% of lawn acreage is fertilized regularly whether or not additional nutrients are required (Barth 1995). The 1998 USGS report on the Appalachicola-Chattahoochee-Flint basin reported the highest phosphorus levels in streams draining urban, suburban and poultry-producing regions (Frick et al 1998).

Literature Review

Width

Since most phosphorus arrives in the buffer attached to sediment (Karr and Schlosser 1977, Peterjohn and Correll 1985, Osborne and Kovacic 1993) or organic matter (Miguel Cabrera, pers. com.), buffer widths sufficient to remove sediment from runoff should also trap phosphorus. In the short term researchers have found riparian buffers retain the majority of total phosphorus that enters, and retention increases with buffer width. Studies in Sweden by Vought et al (1994) determined that after 8 m (26.2 ft), grassed buffers retained 66% of phosphate in surface runoff while after 16 m (52.5 ft) 95% was retained. Mander et al (1997) in Estonia found total phosphorus trapping efficiencies of 67% and 81% for riparian buffer widths of 20 m (65.6 ft) and 28 m (91.9 ft), respectively.

A number of studies (Dillaha et al 1988 and 1989, Magette 1987 and 1989) have documented the performance of grass buffer strips in reducing total phosphorus levels (the design of these studies was briefly described in the previous section on sediment). The results are summarized in Table 2. These authors all noted that effectiveness of the buffers declined over time (the data in Table 2 represent averages of several trials), and that soluble phosphate reductions were lower than total phosphorus reductions. In one case, Dillaha et al (1988) noted that the buffer released more phosphorus than entered. Presumably this increase represented previously trapped phosphorus that was remobilized. With the exception of Dillaha et al 1988, these studies show that increasing buffer width reduces the concentration of phosphorus in runoff. Desbonnet et al also observed this correlation in their 1993 review. Based on data from a number of studies, they reported that buffer width must increase by a factor of 2.5 to achieve a 10 percent increase in phosphorus removal. Figure 7 displays the results shown in Table 2 along with results from Vought et al (1994) and Mander et al (1997).

Study	Total P Removal	
	4.6 m buffer	9.1 m buffer
Dillaha et al 1988	71.5%	57.5%
Dillaha et al 1989	61%	79%
Magette et al 1987	41%	53%
Magette et al 1989	18%	46%

Table 2. Removal of Total Phosphorus by Grass Buffers.

With one exception, studies by Dillaha et al and Magette et al found a positive correlation between the width of grass riparian buffers and the ability to trap total phosphorus in surface runoff.

Limitations

The long-term effectiveness of riparian buffers in retaining available phosphate is questionable. Whereas nitrate can be denitrified and released into the atmosphere, phosphorus is either taken up by vegetation, adsorbed onto soil or organic matter, precipitated with metals, or released into the stream or groundwater (Lowrance 1998). It is possible for a buffer to become "saturated" with phosphorus when all soil binding sites are filled; any additional phosphorus inputs will then be offset by export of soluble phosphate (Daniel and Moore 1997; Miguel Cabrera, pers. com.; Dave Correll, pers. com.). Soils become saturated at different rates, depending on factors such as cation exchange capacity and redox potential. Harvesting vegetation may be the only reasonable management technique that permanently removes phosphorus from the system. Such harvesting can destabilize the riparian area and lead to erosion, however (USACE 1991), and so should be restricted to areas well away from the stream bank. Welsch (1991) recommends 15 ft (4.6 m), although 25-50 ft (7.6 -15.2 m) would provide a greater margin of safety.

Riparian buffers are typically effective at short-term control of sediment-bound phosphorus but have low net dissolved phosphorus retention (Lowrance et al 1997). For example, Daniels and Gilliam (1986) found that riparian buffers of unspecified width reduced total phosphorus by 50%, while soluble phosphate declined by only 20%. Peterjohn and Correll (1984) found that 84% of total phosphorus and 73% of soluble phosphate were removed from surface runoff passing across a 50-m (164 ft) riparian buffer in the Maryland Coastal Plain. On the other hand, Young et al (1980) reported little difference in the reductions of soluble phosphate and total phosphorus across a 21 m (68.9 ft) wide buffer of corn. Total phosphorus declined by 67%, while soluble phosphate was reduced by 69%.

The sediment-bound phosphorus trapped by buffers may slowly be leached into the stream, especially once the buffer is saturated (Omernik et al 1981, Osborne and Kovacic 1993, Mander 1997). A number of studies have shown either no net reduction or a net increase in groundwater phosphate as it crosses the riparian buffer. Studies in which swine waste was applied to 30 m (98.4 ft) buffer strips in South Georgia showed no reduction of phosphate in shallow groundwater (Hubbard 1997). In fact, phosphate levels increased from 0.5 mg/L to 1.0 mg/L over the course of the study, although whether this represented a trend or an anomaly was unclear. Peterjohn and Correll (1984) likewise found that total phosphorus concentrations in shallow groundwater rose at their 50-m (164 ft) riparian buffer study site. In one transect, phosphorus concentrations doubled and in another they quadrupled. A study by Osborne and Kovacic (1993) found that neither a 16 m (52.5 ft) wide forested buffer nor a 39 m (128 ft) wide grass buffer reduced subsurface phosphate loads from crop land.

Note, however, that even when saturated, riparian buffers may still perform a valuable service by regulating the flow of phosphorus from the land to the stream. Sediments and organic materials that carry phosphorus in runoff during storms can be trapped by riparian vegetation. The phosphorus will still slowly leak into the water, but the stream is protected from extreme nutrient pulses (Ronald Bjorkland, pers. com.).

Vegetation

Both grass and forested buffers have been proven effective at reducing total phosphorus, and both vegetation types have also been shown to lose phosphate to the stream. Osborne and Kovacic found that forested buffers leaked phosphate to the stream faster than grassed buffers. Mander et al (1997) found that uptake in younger riparian forest stands was higher than that in more mature stands. Several researchers (Lowrance et al 1985, Groffman et al 1991, Vought et al 1994) suggest periodic harvesting of riparian vegetation to maintain higher nutrient uptake. Such harvesting is recommended in zones two and three (zones greater than 15 ft (4.6 m) from the stream) of the three-zone system promoted by the USDA (Welsch 1991). Other researchers have noted, however, that even mature forests can accumulate nutrients (USACE 1991), and Herson-Jones et al (1995) declared that "Mature forests are thought to have the greatest capacity for modulating the flow of nutrients and water throughout the ecosystem."

Similarly, phosphorus could be permanently removed before it reaches the buffer if an additional field of unfertilized crops or mowed

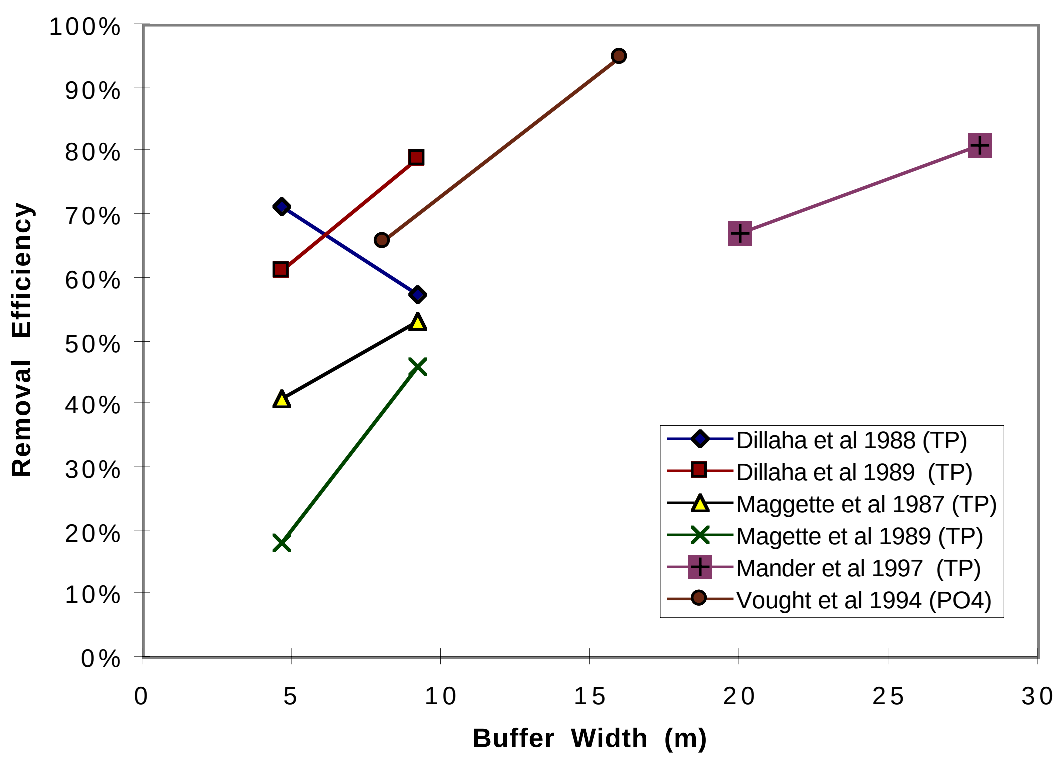


Figure 7. Phosphorus Removal from Surface Runoff by Buffers of Different Widths.

All but one study (Dillaha et al 1988) showed that the phosphorus trapping ability of a riparian buffer increases with width. In most cases, the data points shown here represent the averages of multiple runs.

hayfields were planted between the phosphorus source and the riparian buffer. Young et al (1980) found average total phosphorus reductions of 83% in 27 m (89 ft) and 21 m (69 ft) wide cropped buffers. Cropping allows for productive use of land and permanent removal of much phosphorus before runoff reaches the riparian buffer.

Extent

As for sediment control, effective nutrient control requires continuous buffers on all streams. Gilliam (1994) has noted that for purposes of nutrient reduction, "there should be a strong effort to preserve a wet, vegetated buffer next to ephemeral and intermittent channels or streams." Despite their limitations, riparian buffers are still very important because they separate phosphorus-producing activities from streams. Every unprotected stream segment or gap in the stream represents a point at which pollution can have direct access to the water. In areas without buffers, phosphorus-laden sediments and soluble phosphate can run directly into waterways with very little chance of removal. Sorrano et al (1996) predicted that in an agricultural watershed in Wisconsin, converting all riparian (within 100 m (328 ft) of a stream) agricultural land to forest would reduce phosphorus loading by 55% during a high runoff year, even assuming no net retention of phosphorus by the riparian zone.

Summary and Recommendations

Although riparian buffers can effectively trap phosphorus in runoff, they do not provide long-term storage and are not effective at filtering soluble phosphate. Phosphorus trapped in a buffer may gradually leak into the stream, especially once the buffer becomes P-saturated. Harvesting of riparian vegetation does provide a method of permanently removing some phosphorus from the system.

Riparian zones wide enough to provide sediment control (15-30 m, increasing with slope) should provide short-term control of sediment-bound phosphorus. Wider setbacks should be considered for application of animal waste, fertilization, and other activities that yield large amounts of nutrients. Buffer zones should be placed on all streams. For phosphorus removal, both forested and grassed buffers are equally useful.

Due to their limitations, riparian buffers should not be viewed as a primary tool for reducing phosphorus loading of streams. Every effort should be made to reduce phosphorus inputs at their sources. This can be accomplished through effective erosion control methods; judicious application of fertilizers; proper placement, inspection and maintenance of sewer lines; and restrictions on the land application of waste from concentrated animal feeding operations (CAFOs). If phosphorus is managed responsibly on-site, buffers can store significant amounts of the excess; but if phosphorus is uncontrolled, buffers can quickly become saturated and overwhelmed. Even with their limits, buffers still perform a valuable service by displacing phosphorus-producing activities away from streams and regulating the flow of phosphorus.

B. Nitrogen

Effects

Like phosphorus, nitrogen contributes to the eutrophication of waters. Nitrogen occurs in numerous organic and inorganic forms which are interconvertible under suitable circumstances. Nitrate (NO_3^-) has been the target of many buffer programs because it is potentially toxic to humans and animals at concentrations greater than 10 mg/L. Ammonium (NH_4^+) is another common form of nitrogen that is toxic to many aquatic organisms and is readily taken up by plants and algae. Removal of nitrate and ammonium from drinking water can be a significant water treatment expense (Welsch 1991).

Sources

Nonpoint sources of nitrogen are similar to those of phosphorus: fertilizers applied to agricultural fields; waste from concentrated animal feeding operations (CAFOs); septic drain fields; leaking sewer pipes; and fertilizers applied to lawns. The relative significance of these sources will vary from region to region.

Literature Review

In their 1994 literature review, Desbonnet et al concluded that total nitrogen removal rates for buffers are good, but nitrate reductions are variable and low. There is significant evidence that this is not a valid conclusion. A number of studies either not included in the Desbonnet et al review or published more recently show significant nitrate reductions. Fennesy and Cronk (1997) reviewed riparian buffer literature with a focus on nitrogen reduction and concluded that riparian buffers of 20-30 m (66-98 ft) can remove nearly 100% of nitrate. Gilliam (1994) declares that,

> *"Even though our understanding of the processes causing the losses of NO_3^- are incomplete, all who have worked in this research area agree that riparian zones can be tremendously effective in NO_3^- removal."*

On a landscape level, channelized tributaries with little or no riparian buffer zones may have two to three times the annual nitrate concentration of natural stream reaches with wetland or riparian buffers (Cooper et al 1994).

There are two major ways in which a riparian buffer strip can remove nitrogen passing through it, both of which can be significant:

- Uptake by vegetation
- Denitrification

Denitrification is the conversion of nitrate into nitrogen gas by anaerobic microorganisms. It represents a permanent removal of nitrogen from the riparian ecosystem and may be the dominant mechanism of nitrogen reduction in many riparian systems. Denitrification also occurs within stream channels themselves, though at rates much lower than in riparian areas, especially wetlands (Fennessy and Cronk 1997).

Unlike phosphorus, nitrate is quite soluble and readily moves into shallow groundwater (Lowrance et al 1985). In many areas, most nitrate enters the riparian zone via subsurface pathways (Lowrance et al 1984, 1985, Haycock and Pinay 1993, Muscutt et al 1993, Fennesy and Cronk 1997; but see Dillaha et al 1988). The amount of nitrogen reduction depends a great deal on the nature of these pathways: if the flow is shallow and passes through the root zone of riparian vegetation, vegetative uptake and denitrification can be significant. If the flow bypasses the riparian zone and recharges an aquifer or contributes to base flow of a stream, nitrogen loss may be much less (see Figure 9). This review first looks at the buffer width necessary to remove nitrogen from surface runoff, then considers nitrogen removal from subsurface flow. Denitrification is then examined in greater detail.

Width

Reduction of various forms of nitrogen in surface runoff is reasonably well correlated with buffer width. Dillaha et al (1988) found that 4.6 m (15 ft) and 9.1 m (30 ft) grassed filter strips were moderately effective in removing total nitrogen from surface runoff from a simulated feed lot, but ineffective in removing nitrate. Other studies of similar design by Dillaha et al (1989) and Magette et al (1987, 1989) yielded similar results. Total nitrogen removal efficiencies in all studies increased with buffer width (Table 3).

In their feedlot studies, Young et al (1980) found that 21.34 m (70 ft) buffers of cropped corn reduced total Kjeldahl nitrogen by 67% and ammonium by 71%, though nitrate increased across the buffer. Extrapolating from the data, Young et al suggested that 36 m (118 ft) wide

Study	Total N Removal	
	4.6 m buffer	9.1 m buffer
Dillaha et al 1988	67%	74%
Dillaha et al 1989	54%	73%
Magette et al 1987	17%	51%
Magette et al 1989	0%	48%

Table 3. Removal of Total Nitrogen by Grass Buffers.

Studies by Dillaha et al and Magette et al found a positive correlation between the width of grass riparian buffers and the ability to trap total nitrogen in surface runoff.

buffers are sufficient to protect water quality. Vought et al (1994) reported surface nitrate reductions of 20% after 8 m (26.2 ft) and 50% after 16 m (52.5 ft) for grass buffers in Sweden. They concluded that "a buffer strip of 10-20 m will, in most cases, retain the major part of the nitrogen and phosphorus carried by surface runoff." Jacobs and Gilliam (1985) found that in buffers downslope from Coastal Plain fields without artificial drainage, the nitrate concentration in surface runoff was reduced from 7.9 mg/l to 0.1 mg/l (99%). Though they did not report width of buffer strips used, the authors said that buffer strips of 16 m (52.5 ft) were effective.

A study by Daniels and Gilliam (1996) in the North Carolina Piedmont determined that grassed buffers of 6 m (20 ft) width and combination grass-forested buffers of 13 m (42.7 ft) and 18 m (59.1 ft) width retained 20-50% of ammonium and 50% of both total nitrogen and nitrate. Because sites had different characteristics it is not possible to determine whether width was a factor. In addition, like the Dillaha (1988, 1989) and Magette (1987, 1989) studies summarized above, Daniels and Gilliam only studied surface flow, not subsurface flow. Since in many cases most nitrate passes through buffers in the interflow, studies that ignore it may greatly underestimate (or, in some cases, overestimate) nitrate reduction.

In their studies in the Mid-Atlantic Coastal Plain, Peterjohn and Correll (1985) found that a 50 m (164 ft) buffer reduced all forms of nitrogen in surface runoff. Nitrate in shallow groundwater was reduced considerably across the buffer, but other forms of nitrogen increased in the subsurface flow. These results are summarized in Table 4.

Like Peterjohn and Correll, many other researchers have found that nitrate reduction in subsurface flow is high, although the optimal buffer width depends on factors such as the hydrologic pathway and denitrification potential. Hanson et al (1994) reported that a 31 m (102 ft) wide riparian buffer downslope from a septic tank drain field reduced shallow groundwater nitrate concentrations by 94%, from 8 mg/L to 0.5 mg/L. Jordan et al (1993) found that a 60 m (197 ft) wide riparian buffer adjacent to cropland in the Delmarva Peninsula reduced subsurface nitrate levels from 8 mg/L to less than 0.4 mg/l (95%). Most of the change occurred abruptly within the riparian forest at the edge of the floodplain, where conditions were optimal for denitrification. Mander et al (1997) found total groundwater nitrogen removal efficiencies of 81% and 80% for riparian buffer sites of 20 m (65.7 ft) and 28 m (91.9 ft) width, respectively.

Researchers at the USDA Agricultural Station in Tifton, Georgia applied swine waste to 30 m (98.4 ft) riparian buffer strips of various types. Preliminary results from 1996 showed that shallow groundwater nitrate levels were reduced from 40 mg/L at the top of the plots to 9 mg/L at the lower end of the plots, a reduction of 78% (Hubbard 1997). Previous research in the region had determined that buffers less than 15 m (49.2

		Nitrate (mg/L)	Exchangeable NH4+ (mg/L)	Particulate Org. N (mg/L)
Surface Runoff	Initial:	4.45	0.402	19.5
	Final:	0.91 (79%)	0.087 (78%)	2.67 (86%)
Subsurface Transect 1	Initial:	7.40	0.075	0.207
	Final:	0.764 (90%)	0.274 (increase)	0.267 (increase)
Subsurface Transect 2	Initial:	6.76	0.074	0.146
	Final:	0.101 (99%)	0.441 (increase)	0.243 (increase)

Table 4. Nitrogen Reductions Reported by Peterjohn and Correll (1985).

Values show initial concentration of nutrients entering the 50-m buffer and final concentrations after passing through the buffer. Values in parentheses are the percent reductions across the buffer.

ft) wide can remove significant amounts of nitrate in surface and subsurface flow (Hubbard and Lowrance 1994). Another study in the Tifton area (Lowrance 1992) had determined that a 50-60 m (~160-200 ft) wide riparian buffer reduced groundwater nitrate levels from 13.52 mg/L to 0.81 mg/L (94%) at depths of 1-2 m (3.3 - 6.6 ft). The greatest reduction occurred in the first 10 m (33 ft). Still another study using a mass balance approach (Lowrance 1984) found that a buffer of unspecified width removed 68% of total nitrogen.

Osborne and Kovacic (1993) reported that a 16 m (52.5 ft) wide forested buffer in Illinois reduced shallow groundwater nitrate levels of 10-25 mg/L to less than 1.0 mg/L (a maximum 96% reduction). A 39 m (128 ft) wide grassed buffer in the same area reduced nitrate levels of 15-44 mg/L to about 2.4 mg/L (a maximum 95% reduction).

In reviewing other studies, Vought et al (1994) concluded that nitrate reduction in subsurface flow approaches 100% between 10 m (33 ft) and 20 m (66 ft) into the buffer; increasing the riparian width beyond 20-25 m (62-82 ft) had no further effect. Pinay and Descamps, as referenced by Muscutt et al (1993), concluded that 30 m (98 ft) buffers are sufficient for removing nitrogen. Results from these studies are summarized in Table 5 and Figure 8.

Denitrification

There is an ongoing debate as to which is the dominant mechanism for nitrogen removal: denitrification or vegetative uptake. Fennessy and

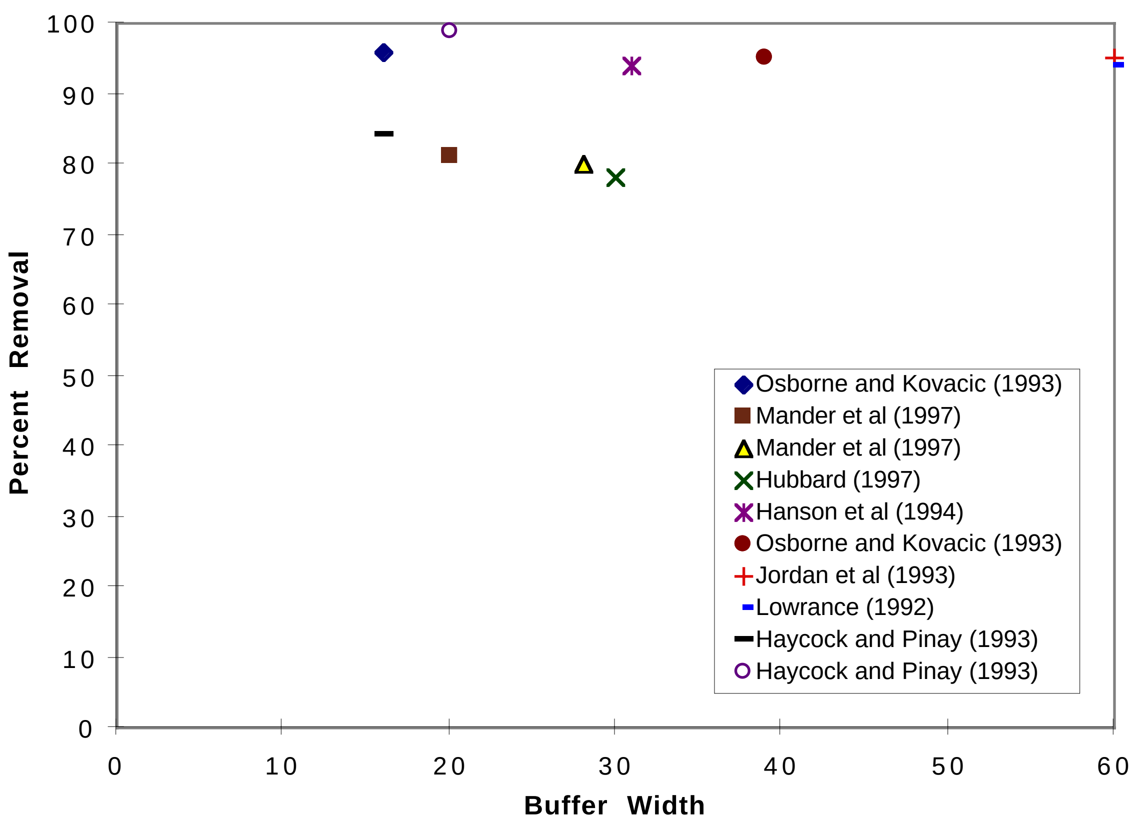


Figure 8. Subsurface Nitrate Removal in Several Studies.

Numerous studies of riparian buffers have reported high rates (>75%) of nitrate removal from shallow groundwater. There is not a clear correlation between nitrate removal rate and riparian buffer width, however.

Cronk (1997) claim that denitrification is the most significant, and there is some evidence to support this view (e.g., Jacobs and Gilliam 1985, Peterjohn and Correll 1985). Lowrance (1992) has made the case for vegetative uptake and has emphasized that, whichever method is dominant, vegetation is necessary for nitrogen removal. Lowrance (1992) and Hanson et al (1994) have reported significant nitrate reductions in shallow groundwater one to two meters deep that appear to be correlated with high denitrification rates at the surface. It appears that vegetation takes up the nitrate and transfers it to the surface layer, where it is denitrified. In the end, local conditions will likely determine which mechanism dominates (Gilliam 1994).

Denitrification is one of a number of coupled processes which are best described by thermodynamic theory (Hedin 1998). Interestingly, there is a significant inverse relationship between denitrification and phosphorus removal. Highly reducing conditions that are suitable for denitrification also favor reduction of iron oxyhydroxides, which can release bound phosphorus and increase the phosphate that is exported from the buffer (Jordan et al 1993).

Soil microorganisms have the capacity to process nitrate at much higher concentrations than they normally experience (Duff and Triska 1990, Groffman et al 1991a, b, Lowrance 1992, Hanson et al 1994, Schnabel 1997). Denitrification rates can increase quite rapidly in response to nitrate increases. In some cases, microorganisms can denitrify all of the available nitrate, but ammonium and organic N pass through the buffer because they are not processed (nitrified) quickly enough (Lowrance 1992). Many soils are also carbon limited or become carbon limited at high nitrate levels (Groffman et al 1991a, b; Hedin et al 1998). Hedin et al (1998) reported on a carbon-limited site with very high nitrate levels close to the stream; when sufficient carbon was added, denitrification levels were exceedingly high (equivalent to 6600 kg N/ha per year). To promote more carbon availability, Hedin et al recommended maintaining organically rich riparian wetlands. Duff and Triska (1990) observed denitrification in the hyporheic zone of a small headwater stream. In their study site, groundwater had a high concentration of dis-

	Width (m)	% Reduction	Final Conc. (mg/L)
Osborne and Kovacic (1993)	16	96	<1.0
Haycock and Pinay (1994)	16	84	N.R.
Haycock and Pinay (1994)	20	99	N.R.
Mander et al (1997)	20	81	N.R.
Mander et al (1997)	28	80	N.R.
Hubbard (1997)	30	78	9
Hanson et al (1994)	31	94	0.5
Osborne and Kovacic (1993)	39	95	<1.0
Jordan et al (1993)	60	95	0.4
Lowrance (1992)	60	94	0.81

Table 5. Nitrate Removal in Shallow Groundwater.

Studies have demonstrated consistently high removal rates for nitrate from shallow groundwater passing through riparian buffers. “Final Conc.” is the concentration of nitrate in groundwater leaving the riparian buffer. Concentrations over 10 mg/L (ppm) are considered potentially harmful.

solved organic carbon (DOC) which was reduced as it passed through the riparian zone. Denitrification was greatest farther from the stream where nitrate was in greatest supply. Rhodes et al (1985) reported 99% nitrate removal in riparian forests and wetlands at a high-altitude undisturbed watershed in Nevada.

Denitrification takes place under conditions of reduced oxygen availability and is correlated with soil moisture. Rates are typically very high in wetlands (Groffman et al 1991a, b; Hanson et al 1994; Collier et al 1995b). In field studies in Rhode Island Groffman et al reported that variability in nitrate reduction in subsurface flows (ranging from 14% to 97%) was almost entirely explained by soil moisture. Wetland soils had consistently high nitrate removal, while better drained upland soils had lower and more variable nitrate removal efficiencies. Since denitrification is the most permanent method for removing nitrogen, good management practices call for preservation of areas of high denitrification activity, such as wetlands (Collier et al 1995b, Hedin et al 1998). Note, however, that denitrification has also been observed under well oxygenated soil conditions, presumably indicating that there can be sites of local anoxia (Duff and Triska 1990).

Nitrate loss does not appear to be limited to warm months. In a study during the winter season in Britain, Haycock and Pinay (1993) reported that a 20 m (65.6 ft) wide poplar forested site retained 99% of the nitrate that entered, no matter how high the load level. A 16 m (52.5 ft) wide grass riparian zone retained nearly 100% of nitrate at lower concentrations but only 84% at high concentrations. All flow was subsurface. In the poplar site nitrate reduction was essentially complete after the first 5 m (16 ft) of flow. Bacterial denitrification was assumed to be the mechanism for nitrate loss, since the vegetation was dormant and uptake rates would be low. The variation between the sites may have been due to the larger amounts of carbon contributed by the poplars (Haycock and Pinay 1993). Lowrance (1992) and Osborne and Kovacic (1993) also reported nitrate removal rates that were independent of season. Lowrance noted that dormant season nitrate removal is not just due to denitrification but can result from uptake by trees as well. Groffman et al (1991b) conducted analyses of denitrification in surface soils at sites in Rhode Island. They reported total nitrogen removal efficiencies of 40% to 99% for overland flow during the summer, but rates of less than 30% in the winter. Additional field studies conducted by the same team failed to find an influence of seasonality on nitrate removal. They theorized that elevated water tables in winter may have brought denitrifying bacteria into contact with more nitrate-laden water, compensating for the lack of vegetative uptake (Groffman et al 1991b).

The characteristics of groundwater flow will determine where within the riparian buffer denitrification will occur. Lowrance (1992) found that most nitrate loss occurred at the buffer-field interface. Hedin et al (1998) and Schnabel et al (1997) studied systems in which shallow groundwater with very high nitrate concentrations entered the buffer very close to the stream. Some studies (e.g. Jordan et al 1993) have shown that denitrification rates are greatest at the edge of the floodplain. In all cases, nitrate removal occurred at locations where the water table was near the surface and both carbon and nitrate were in good supply. Determining all these factors in the field is not easy and the hydrology of many riparian areas is still poorly understood, making accurate predictions of nitrogen removal difficult (Gilliam 1994).

It appears that few researchers have documented nitrate reductions in subsurface flow in the Piedmont or Blue Ridge physiographic provinces. Lowrance et al (1997) cited data from studies by Daniels and Gilliam that show large reductions in groundwater nitrate levels in North Carolina Piedmont locations. The study sites were characterized by high water tables and shallow groundwater flow through the root zone of riparian vegetation (see Figure 9-a). When these conditions are present, as in areas of thin soils, high rates of nitrate reduction should occur (Lowrance et al 1997). However, in Piedmont soils underlain by schist/gneiss bedrock, an appreciable amount of flow may move into regional aquifers in the saprolite, bypassing the riparian root zone and contributing to the base flow of streams (Figure 9-b). A moderate level of denitrification is expected under these conditions (Lowrance et al 1997). In Piedmont soils underlain by marble bedrock, most flow may enter

Figure 9. Different Possible Groundwater Flow Paths.

Based in part of Lowrance et al (1997).

Figure 9a. Groundwater flow paths across a riparian buffer with shallow soils or an aquitard (semi-impervious layer). Flow should pass through the root zone, allowing significant removal of nutrients and contaminants.

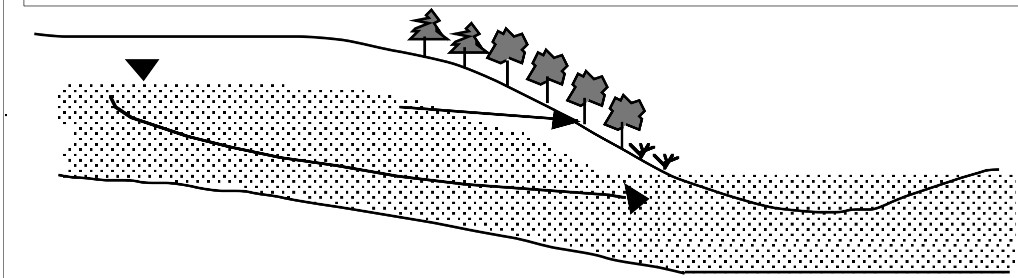


Figure 9b. Groundwater flow paths across a riparian buffer with moderately deep soils. Some flow should passes through the root zone, but some bypasses the riparian area. The area of greatest nutrient removal may be very close to the stream.

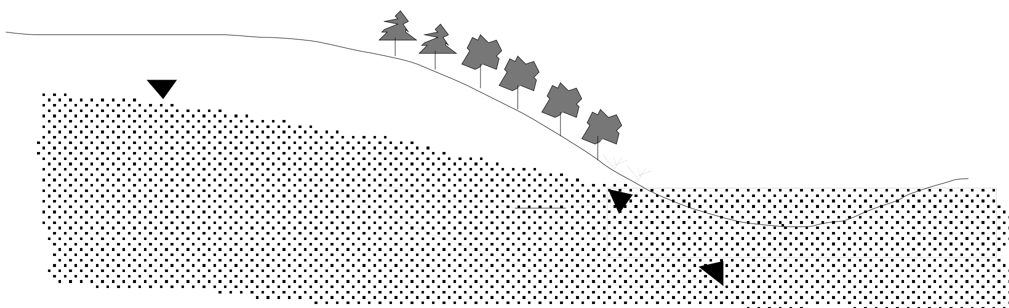


Figure 9c. Groundwater flow paths across a riparian buffer across a wide, flat floodplain with a high water table. Area of greatest nutrient removal will likely be the edge of the floodplain.

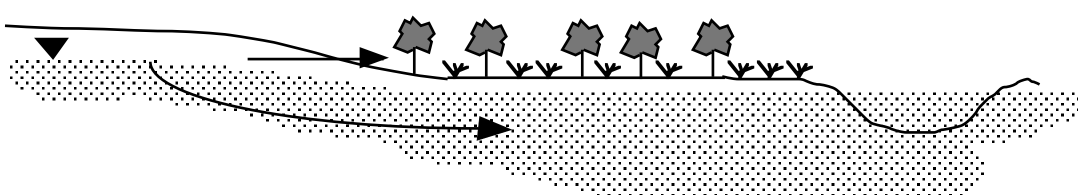
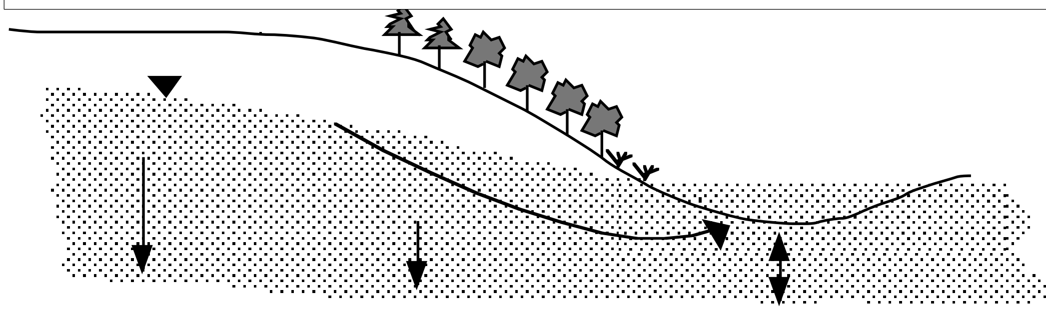


Figure 9d. Groundwater flow paths across a riparian buffer with very deep soils or fractured bedrock. Much groundwater bypasses the riparian buffer, and nutrient removal may be lower than in other systems.



regional aquifers and nitrate loss in riparian areas is probably much less significant (Lowrance et al 1997) (Figure 9-d). However, in their study of riparian sites in the Valley and Ridge physiographic province, Schnabel et al (1997) found significant denitrification rates where nitrate-rich groundwater emerged close to the stream bank. These sites had flow-restrictive layers 8-10 m (26.2 - 32.8 ft) below the surface, as would Piedmont soils with marble bedrock. The deeper flow paths changed the location of denitrification activity but did not prevent nitrate reduction. Hedin (1997) also observed a similar phenomenon at a site in Michigan with deep glacial soils.

Even within the Coastal Plain hydrology can vary significantly, affecting how nutrient reduction takes place. Human modifications also alter these patterns. Jacobs and Gilliam (1985) found that at a site in the Middle Coastal Plain of North Carolina most transport was via subsurface pathways. In a Lower Coastal Plain site, on the other hand, most movement was by surface flow. Fields in this area were drained by ditches, and a dense clay B layer prevented deeper subsurface flow.

Rates of nutrient reduction are influenced by the length of time water is retained in the buffer (Fennessy and Cronk 1997), which in turn is determined by slope, rainfall, soil characteristics, hydrologic flow path, and the width of buffer (Phillips 1989a,b). Retention time may actually be longer than would be indicated by these factors, because some researchers have shown that the flow of water is not perpendicular to the channel but oblique to the channel. Residence times in a 27 m wide buffer along a Thames, England headwater stream ranged from 12 days to over three years (Fennessy and Cronk 1997). Research on the Georgia Coastal Plain showed that it can take several seasons for nutrients and contaminants to pass through a 50-m wide riparian buffer where shallow lateral flow is the dominant pathway (Hubbard and Lowrance 1996). On the other hand, when overland flow occurs, water can pass through the buffer in a matter of minutes. Models have been developed to predict buffer effectiveness based on detention time (Phillips 1989a, b), but their accuracy is unproven.

Extent

As discussed in previous sections, protection of water quality requires preservation of buffers on as many streams as possible.

Vegetation

Both grass and forested buffers have been shown to reduce nitrogen effectively. In studies in Rhode Island, Groffman et al (1991a) found that when nitrate was added to soil cores, soils from grass plots exhibited denitrification rates an order of magnitude higher than those from forested plots. Schnabel et al (1997) also reported higher denitrification rates for grassed buffer sites. Haycock and Pinay (1993) and Osborne and Kovacic (1993), on the other hand, found higher rates of nitrate retention in forested buffers. Lowrance has concluded that overall, grass buffers are not effective at removing nutrients from shallow groundwater (Lowrance 1998). Both grass and forested buffers have proven effective in removing surface nutrients from surface runoff, although grass buffers have been more heavily studied.

Summary and Recommendations

The nitrogen removal capacity of riparian buffers has been well established. Nitrogen removal in surface runoff has been correlated with buffer width, but rates appear to be lower than for subsurface reduction. Studies have documented high levels of nitrate removal from shallow groundwater, which is the dominant mode of nitrate transport through many buffers. Nitrate may be removed by both vegetative uptake and denitrification. The buffer width necessary for nitrate reduction depends greatly on the hydrologic flow paths, and numerous studies would be required to fully characterize the pathways of shallow groundwater flow in all areas of Georgia. However, based on existing research, most areas of Georgia should generally support significant nitrogen removal in the riparian buffer.

Because the distribution of denitrification sites vary spatially, wider buffers will on average include more denitrification sites than narrower buffers. A minimal width of 15 m (50 ft) is probably necessary for most buffers to reduce nitrogen levels. Wider buffers of 30 m (100 ft) or greater would be more likely to include other areas of denitrification activity and provide more nitrogen removal. Buffers should be preserved along as many streams as possible, and it is especially important to preserve riparian wetlands, which are sites of high nitrogen removal.

C. Other Contaminants

Organic Matter and Biological Contaminants

Human and animal waste contribute to aquatic degradation in ways other than nutrient contamination. First, these wastes carry with them an array of pathogenic microorganisms. Secondly, when organic matter is broken down by aerobic bacteria in water, oxygen is consumed rapidly. High levels of organic matter with high biological oxygen demand (BOD) can use up all of the available oxygen in a stream, river or lake, killing fish and other organisms. Whereas surface waters naturally have a BOD of 0.5 to 7 mg/L, chicken wastes may have a BOD of 24,000 to 67,000 mg/L (Cooper et al 1993).

Fecal coliform is used as an indicator of pathogenic microorganisms. The levels of fecal coliform can vary temporally at a single site and can be quite high, even in areas of Georgia considered less impacted. At a monitoring station in Lake Allatoona near where the Etowah River enters the reservoir, fecal coliform levels ranging from 1.8 colonies (in May) to 24,000 colonies (in November) per 100 ml were recorded. For reference, the recommended limit for water that people directly contact (i.e., the "primary contact standard") is 200 colonies / 100 ml.

Sources of organic matter and biological contaminants include leaking sewer pipes, improperly functioning septic systems, animal waste sprayed onto fields and animal waste lagoons. Burkholder et al (1998) documented a large fish kill in deoxygenated water after a swine lagoon ruptured into the Neuse River in North Carolina.

Riparian buffers can trap waste transported in surface runoff in the same way that they trap sediments and associated nutrients. In the face of very high waste levels, however, trapping efficiency may not be adequate to reduce contaminants to a safe level. Coyne et al (1995) applied poultry manure to two test plots and measured fecal coliform reduction across 9 m (30 ft) wide grass filter strips. After artificial rain was applied, researchers found that fecal coliform concentrations were reduced by 74% and 34% in the two strips. Nevertheless, runoff exceeded the primary contact standard in every sample. A 1973 study by Young et al found that a 60 m (197 ft) long grass filter strip reduced fecal coliform by 87%, total coliform by 84% and BOD by 62% (Karr and Schlosser 1977). Cooper et al (1993) found that constructed wetlands removed 76% of BOD when coupled behind an anaerobic lagoon. There do not appear to be other studies which have addressed this issue.

Pesticides and Metals

Pesticides are intended to be toxic. When these chemicals are introduced into aquatic systems they can cause both direct mortality to organisms and various sublethal effects (Cooper et al 1993). Although a number of the most persistent and toxic pesticides have been banned in the U.S., many that are currently applied are still quite dangerous. According to Cooper et al (1993), "Some insecticides in current use not only accumulate, they can be as toxic as and often more toxic than banned organochlorines."

Besides their use in row crop agriculture, pesticides are gaining favor in forestry in the Southeast (Neary et al 1993) and are commonly used on lawns and plantings in urban areas. The EPA estimates that nearly 70 million pounds of active pesticide ingredients are applied to urban lawns each year. One survey of 500 homes found that 50 different pesticides were used (Schueler 1995a, b). Even long-banned insecticides like DDT and chlordane are commonly found in urban streams. Chlorpyrifos also appears in runoff at levels toxic to a range of wildlife including geese, songbirds and amphibians (Schueler 1995b). A recent study by the U.S. Geological Survey found that urban and suburban streams in the Atlanta region had levels of diazinon and carbaryl that exceeded aquatic life criteria (Frick et al 1998). Heavy metals are usually associated with industrial activities, and concentrations tend to be highest in streams draining urban areas (Crawford and Lenat 1989).

Buffers are very important in displacing pesticide application away from streams, preventing direct contamination and reducing the danger of drift. Many pesticides are broken down within buffer soils, while metals may bind to soil particles. Greater buffer width increases the retention time for chemicals (allowing more opportunities for contaminants to decompose) and provides more sites for binding metals. Frick et al (1998) attribute the unexpectedly low pesticide levels in agricultural Coastal Plain streams to the largely intact forested wetlands and floodplains.

The mechanisms of pesticide transport are not well understood (Muscutt et al 1993). Lowrance et al (1997) examined changes in pesticide concentrations crossing a 50-m (164 ft) wide buffer in the Georgia Coastal Plain. Atrazine and Alachlor were reduced from 34 μg/L and 9.1 μg/L, respectively, to less than 1 μg/L. The chemicals took three years to enter groundwater, and it appeared that they first moved laterally across the buffer before infiltrating deeply. Hatfield et al (1995) found that grassed filter strips of 40 ft (12.2 m) and 60 ft (24.4 m) removed 10-40% of the atrazine, cyanazine and metolachlor passing across them. Arora et al (1996) found that 20.12 m (66 ft) wide riparian buffers of 3% slope retained 8-100% of the herbicides (atrazine, metolachlor and cyanazine) that entered during storm events. The variation was related to the amount of runoff that occurred during the storms. None of these studies examined the long-term fate of pesticides or their degradation products.

Neary et al (1993) reviewed recent studies in the Southeast on the use of buffers in reducing pesticide contamination of water. They found that cases of high concentrations of pesticides in water only occurred when no buffer was used or when they were applied within the buffer (i.e., the buffer was violated). Regular use of buffer strips kept pesticide residue concentrations within water-quality standards. Neary concluded that "Generally speaking, buffer strips of 15 m (49 ft) or larger are effective in minimizing pesticide residue contamination of stream flow."

Herson-Jones et al (1995) concluded that urban buffers have shown a moderate to high ability to remove or retain hydrocarbons and metals from surface runoff. They cited data from a 1992 study by the Metropolitan Seattle Water Pollution Control Department which found removal rates exceeding 40% for lead, 60% for copper, zinc and iron, and 70% for oil and grease. Studies in Rhode Island (Groffman et al 1991b) also found high metal retention rates. The authors reported that riparian buffers retained all the copper that was added to them. This retention depends on cation exchange capacity (CEC) of the soil, however, and it is possible for buffer CEC to become saturated, just as it can under high phosphorus loads.

Summary and Recommendations

Based on the limited studies available, riparian buffers are useful for reducing levels of biological contaminants and organic matter, but by themselves may not be sufficient to protect water quality. Buffers at least 9 m (~30 ft) wide and probably much wider are needed; width should be extended for steeper slopes that would reduce buffer contact time. Every effort should be made to reduce these contaminants at their source, and it is wisest to prohibit sources within the floodplain, regardless of buffer width.

Buffers can remove pesticides and heavy metals, but the width necessary is unclear from the existing research. Neary's (1993) recommendation of 15 m (49 ft) should be viewed as a minimum width, since the studies by Hatfield et al (1995) and Lowrance et al (1997) suggest that significantly wider buffers may be required.

IV. Other Factors Influencing Aquatic Habitat

Aquatic habitat quality is very important in the Southeast, which has a high level of fish and mussel diversity. Probably the most important factor affecting the habitat of aquatic organisms is sediment, discussed in detail in Section II. This section will discuss other factors that influence the habitat of stream organisms and the characteristics of buffers required to support high quality habitat.

Woody Debris and Litter Inputs

Large woody debris (LWD) deposited into the stream from the riparian zone provides essential habitat for many fish. According to May et al (1996), LWD is the most important factor in determining habitat for salmonids (salmon, trout and related fish). Leaf litter and other organic matter from riparian forests, including terrestrial invertebrates that drop into the water, are an important source of food and energy to stream systems.

In studies in Alaska, researchers found that during the winter, salmonid survival depended upon the amount of debris in streams (Murphy et al 1986). Stream reaches that were protected by 15-130 m (49-427 ft) wide riparian buffers were found to be similar in habitat quality to old growth reaches. Clear cutting led to short-term increases in summer salmonid populations because overall stream production increased, but in the winter there was insufficient debris to provide shelter for fish. Forested stream corridors are necessary to provide regular inputs of LWD and removal of riparian forest can have long-term negative effects. Gregory and Ashkenas note that "of all the ecological functions of riparian areas, the process of woody debris loading into channels, lakes and floodplains requires the longest time for recovery after harvest" (Gregory and Ashkenas 1990). Collier et al (1995a) recommend a buffer width of at least one tree height to maintain inputs of LWD, although for stability purposes (i.e., to prevent windthrow) they suggest that a width equal to three tree heights may be necessary.

The type and amount of riparian vegetation is correlated with different fish communities (Baltz and Moyle 1984), and studies indicate that native vegetation is important for proper stream functioning (Abelho and Graça 1996, Karr and Schlosser 1978). Stream organisms may not be adapted to the leaf fall patterns or the chemical characteristics of leaves from nonnative trees, suggesting that management schemes should include the maintenance and restoration of native vegetation (Abelho and Graça 1996).

Removal of riparian forests can cause a fundamental shift in stream energy dynamics, moving the system from heterotrophy (where production is based on inputs of leaves and other terrestrial matter) to autotrophy (where production is based on algae) (Allan 1995). This shift also alters the seasonal dynamics of the stream (Schlosser and Karr 1981). Streams with riparian vegetation experience a peak in organic matter in the fall, but streams without riparian vegetation experience peaks in the summer.

Aquatic invertebrates are important components of the stream system, so much so that they are commonly used as indicators of stream health.

Aquatic invertebrates are the major food source for many, if not most freshwater fish. Riparian vegetation, in turn, provides leaves and other forms of litter that feed these invertebrates. Additionally, most aquatic invertebrates emerge from the stream as adults and use the riparian zone for reproduction (Erman 1984). Riparian vegetation also influences the amount and type of *terrestrial* invertebrates that fall into streams. Some fish, such as brown trout, may rely on terrestrial invertebrates for most of their food (Dahl 1998). A study in New Zealand found that pasture streams had a much lower biomass (1/6th to 1/12th) of terrestrial invertebrates than either ungrazed grassland or forest streams (Edwards and Huryn 1996). Other factors, such as altitude and stream width, were not found to be significant in the study. Of course, the importance of terrestrial organisms as a food source is most important in headwater streams and less significant in larger streams and rivers that have higher algal production (Vannote et al 1980).

Temperature and Light Control

In Georgia, like most of the U.S., the native vegetation in riparian zones is hardwood forest. These forests keep headwater streams cool by providing shade for the surface water and reducing the temperature of the shallow groundwater that feeds the stream. Removing these riparian forests will increase stream temperatures, and even minor changes in temperature can cause major changes in the fish community (Baltz and Moyle 1984).

Although increases in temperature and light can generate increased aquatic production in some cases, many aquatic organisms can only survive within a relative narrow temperature range (Allen 1995). Trout are a well-known and commercially important example of a fish that cannot tolerate high temperatures. Thermal fluctuations can have a range of direct effects on mussels, including reproductive problems and death (Morris and Corkum 1996). Higher water temperatures also decreases oxygen solubility, which harms many organisms and also reduces the water's capacity to assimilate organic materials and increases the rate at which nutrients solubilize and become readily available *(Karr and Schlosser 1978)*.

Factors other than shading affect stream temperature, however. Dams can cause profound changes to the stream thermal regime that override the influence of riparian forests. Impoundments that release water from the top increase downstream water temperature, while bottom-release dams decrease downstream water temperature. Additionally, discharges of cooling water from power plants can greatly increase water temperature.

On small streams, however, shading is likely to be the most important factor. A study by Barton et al (1985) found that most of the variation in the maximum water temperature was related to the fraction of forested bank within 2.5 km upstream of the study site, while maximum weekly temperature was correlated with buffer length and width. This regression accounted for 90% of temperature variability. The authors reported that water temperature was the only important factor determining the presence of trout. For these fish to be present, 80% of banks within 2.5 km upstream had to have forests of at least 10 m (33 ft) wide, or sufficient to shade the stream (Barton et al 1985). In Georgia, Gregory et al (in press) found that mean water temperatures in some Coastal Plain streams with no riparian cover approached 37° C in the summer, while nearby streams with forested riparian zones were 15° cooler. Temperature in the streams varied by as much as 20° C in the winter and spring. The authors suggested that the thermal variability was likely to be a factor in the variability in aquatic invertebrate communities they found in the streams.

A study of mussels in Ontario found dramatic differences between mussel communities in forested and agricultural catchments (Morris and Corkum 1996). Agricultural streams were dominated by one species of tolerant mussel (*Pyganodon grandis*), which represented 62.5% of individuals in those rivers (and only 1% in forested basins). The authors identified temperature as an important variable influencing the shift, although nutrients may also have been a factor.

In a review of several articles on the subject, Osborne and Kovacic (1993) concluded that buffer widths of 10-30 m (33-98 ft) can effectively maintain stream temperatures. Shading has the greatest impact on smaller streams. Collier et al (1995b) note that "generally, protecting or planting small headwater streams achieves the greatest temperature reduction per unit length of riparian shade." This again indicates the need to establish buffers on even the smallest streams when possible.

Summary and Recommendations

Removal of riparian forests has a profoundly negative effect on stream biota. Davies and Nelson (1994) summarized the range of effects clearcutting can have on stream communities: "Logging significantly increased riffle sediment, length of open stream, periphytic algal cover, water temperature and snag volume. Logging also significantly decreased riffle macroinvertebrate abundance, particularly of stoneflies and leptophlebiid mayflies, and brown trout abundance." The researchers recommended a 30 m (98 ft) buffer to mitigate these effects. At a minimum, a 50 ft (15 m) buffer appears necessary to provide woody debris inputs to the stream. No tree harvesting should occur within 25 ft (12 m) of the stream (50 ft/15 m is preferable), and harvesting in the remainder of the buffer should leave some mature and senescent trees. Native vegetation should be preserved whenever possible. To maintain stream temperatures, riparian buffers must be at least 10 m (30 ft) wide, forested, and be continuous along all stream channels to maintain proper stream temperatures. It is important to note that while some other riparian functions (e.g., sediment and nutrient retention) can be performed adequately by grassed buffers, forested buffers of native vegetation are vital to the health of stream biota.

Article	Widths Studied (m)	Min. Width Recommendation (m)
Hodges and Krementz (1996)	36-2088	100
Keller et al (1993)	25-800	100
Kilgo et al (1998)	25-500	both narrow and wide
Kinley & Newhouse (1997)	14-70	70
Smith & Schaefer (1992)	20-150	no recommendation
Spackman and Hughes (1995)	25-200	150-175
Thurmond et al (1995)	15-50	15
Triquet et al (1990)	15-23	no recommendation

Table 6. Riparian Buffer Recommendations from Avian Studies.

The recommendations of the literature on riparian corridor widths for birds are summarized here. The second column shows the range of buffer widths studied by the authors. The third column shows the authors' recommendations for the minimum corridor widths necessary to support bird populations.

V. Terrestrial Wildlife Habitat

Riparian corridors support an exceptional level of biodiversity, due to natural disturbance regimes, a diversity of habitats and small-scale climatic variation (Naiman et al 1993). Gregory and Ashkenas (1990) found that riparian forests in the Willamette National Forest support approximately twice the number of species than are found in upland forests. Riparian zones also support many rare species (Naiman et al 1993). Riparian areas are a declining habitat, however. Malanson (1993) estimates that 70% of natural riparian communities have been lost; in some areas losses may be as high as 98%. Naiman et al (1993) put the loss at an average 80% for North America and Europe.

Literature Review

Gregory and Ashkenas (1990) have noted that riparian buffers established for water quality and fisheries needs may not meet the habitat requirements of terrestrial wildlife. The ability of a stream corridor to support wildlife is usually directly related to its width (Schaefer and Brown 1992). Narrow buffers may support a limited number of species, but wide buffers will be required to maintain populations of riparian-dependent interior species. Generally, most researchers advocate preserving as wide a buffer as possible. Schaefer and Brown (1992) have suggested that a protected river corridor should cover the floodplain and an additional upland area on at least on one side. Other researchers have attempted to quantify the necessary width according to the needs of various riparian-dependent species.

Birds

Over the last decade there has been an abundance of research on the use of riparian corridors by birds. The recommendations of many of these studies are summarized in Table 6.

Triquet et al (1990) examined bird populations in mature forest, clearcut forest, and clearcuts with a 15-23 m (49-76 ft) wide riparian buffer. They found that retaining the buffer provides habitat for some species of mature-forest and edge-dwelling songbirds that otherwise would be absent. Birds associated with mature forests virtually disappeared from the clearcut site, though at the buffer strip site the decline was much less (Triquet et al 1990).

Keller et al (1993) assessed bird species at 117 riparian corridors of 25 m (82.0 ft) to 800 m (2624 ft) width in Maryland and Delaware. They found that the total number of neotropical migrant species increased with forest width, and ten species increased in abundance as width increased. Keller et al recommended preserving riparian corridors at least 100 m (328 ft) wide, and even wider corridors when possible. Where intact riparian areas exist, they suggested giving priority to the widest corridors available. However, they said efforts to create or increase riparian forest width should focus first on streams with no vegetation and then on narrow (<50 m / 164 ft) forests: “The presence of even a narrow riparian forest dramatically enhances an area’s ability to support songbirds compared to a stream surrounded only by agricultural fields or herbaceous riparian habitats” (Keller et al 1993).

In surveys in Vermont forests, Spackman and Hughes (1995) found that 90% of bird species are included within 150-175 m (492-574 ft) buffers along most streams. At two streams the distance was less. For most sites, 90% of plant species are represented within 15 m (49 ft) of the stream. Because Spackman and Hughes studied riparian areas that were part of intact mature forests, however, their findings are not completely relevant to riparian buffers bounded by open fields or urban development.

Kilgo et al (1998) studied bird richness and abundance in bottomland hardwood forests in Southern South Carolina. Widths of forests ranged from less than 50 m (164 ft) to over 1000 m (3280 ft), measured on both sides of the stream (except on the Savannah River, where forest on only one side was measured because the river was a flight barrier). Uphill land cover was either closed-canopy pine or field-scrub. The authors reported that species richness showed a strong positive correlation with bottomland forest width, even though the adjacent habitat was also forested. They did not find a significant buffering effect of the pine habitat. That is, sites with field-scrub habitat uphill rather than pine forest showed no decrease in bird abundance and richness. This may be due to the great width of most of the buffers the researchers studied. Kilgo et al (1998) recommend protecting both narrow and wide riparian buffers, although they point out the importance of preserving the few remaining areas of wide bottomland hardwood forest.

In a study in the Altamaha basin of the Georgia Coastal Plain, Hodges and Krementz (1996) measured densities of neotropical migrant birds in narrow (36-330 m/118-1082 ft), medium (440660 m/1443-2165 ft) and wide (1520 - 2088 m/~0.95-1.3 mi) riparian corridors. They found a significant increase in bird densities for several species between 50 m (164 ft) and 100 m (328 ft). Beyond 100 m, there was little increase associated with wider corridors. The researchers suggested that "forest corridors of about 100 m should be sufficient to maintain functional assemblages of the six most common species of breeding neotropical migratory birds."

Thurmond et al (1995) examined bird populations in narrower riparian corridors in the Ogeechee River basin in the Upper Coastal Plain of Georgia. Riparian buffers of 50 ft (15.2 m), 100 ft (31 m) and 164 ft (50 m) adjoining pine plantations of less than five years age were compared to mature riparian areas. The authors found that breeding forest interior species were virtually absent from all buffer strips, although overall abundance and densities in these strips were higher than in the adjacent pine plantations. They concluded that narrow protected stream corridors are important in maintaining greater bird diversity even though they are insuffîcent for protecting interior species.

Smith and Schaefer (1992) found small differences between bird populations in narrow (20-60 m/ 66-197 ft) and wide (75-150 m/246-492 ft) naturally vegetated buffers in an urbanized North Florida watershed. Area-sensitive species such as Acadian Flycatchers and Hooded Warblers were not found in the narrow buffers. Summer Tanagers were not recorded anywhere in the urbanized area, but they were found in a nearby undisturbed riparian forest. The researchers found that during spring, bird species diversity and evenness were less in Hogtown Creek, but average density was greater. During winter, bird density and richness were greater in Hogtown Creek.

Kinley and Newhouse (1997) studied breeding bird populations in riparian buffers of 14 m (46.0 ft), 37 m (121 ft) and 70 m (230 ft) in British Columbia. They found that densities of all birds increased as buffer width increased, and they concluded that "narrower riparian reserve zones are of less value than wider reserve zones."

Researchers have frequently reported bird densities and richness that are equal or greater in narrow buffers or clearcut areas. After clearcutting, bird diversity and abundance may increase because of the influx of open-habitat and edge-habitat birds (e.g., Triquet et al 1990). This is an example of the edge effect: boundaries like forest edges (and riparian zones) tend to be especially rich in biodiversity. It is a management problem in some ecosystems to maximize both edge habitat and interior habitat. In addition, many species require more than one ecological system in which to complete their life cycles (Naiman et al 1988). However, generally speaking, animals that exploit impacted areas and edges are more likely to be habitat generalists that are less in need of protection. Measurements of species richness and population density are less useful than indices of similarity between developed and undeveloped sites. Management on the local scale for maximum richness and density will almost certainly result in the loss of habitat specialists.

Mammals

Few studies have explicitly addressed the issue of how wide riparian buffers need to be to support mammal populations. Cross (1985) found that riparian zones in mixed conifer forest sites in southwest Oregon supported a higher diversity and density of small mammal species than upland habitat. Diversity and species composition in a 67 m (220 ft) wide riparian buffer bordered by a clearcut were found to be comparable to undisturbed sites.

Large mammals, as well as large reptiles such as alligators, are also important for the role they play in determining the structure of streams and riparian zones (Naiman and Rogers 1997). For example, beavers create wetlands in areas where they would otherwise not exist, increasing the overall diversity of the aquatic community in those regions (Snodgrass and Meffe 1998). Removal of large animals leads to simplification of the ecosystem and loss of diversity (Naiman and Rogers 1997).

Reptiles and Amphibians

Riparian zones are often rich in both diversity and abundance of reptiles and amphibians. In some mountainous areas the number and biomass of salamanders can exceed that of birds and mammals (Brode and Bury 1984).

Reptiles and amphibians vary in their dependence upon riparian areas. Many amphibians spend their entire lives within the stream and riparian zone, while other species use it for breeding or as part of a larger range (Brode and Bury 1984). In Western Oregon reptiles and amphibians that are dependent upon riparian areas may require buffers of 75-100 m (246-328 ft) (Gomez and Anthony 1996). The authors noted that many species may also require preservation of large areas of old growth and upland habitat as well. Likewise, in a study of Carolina Bays, Burke and Gibbons (1995) found that a 275 m (902 ft) upland buffer is required to protect all nest and hibernation sites for certain freshwater turtles. Beyond a certain width, however, habitat heterogeneity is probably more important than habitat width. Burbrink et al (1998) found that 100 m (328 ft) naturally vegetated riparian zones supported reptile and amphibian diversities that were as high as 1 km (0.62 mi) wide naturally vegetated riparian zones.

Vegetation

Relatively few riparian studies have focused on the needs of native terrestrial vegetation. Gregory et al (1992) observed that "riparian zones are commonly recognized as corridors for movement of animals within drainages, but they also play an important role within landscapes as corridors for dispersal of plants." Riparian zones provide areas of habitat heterogeneity and can support high plant diversity. In the Vermont Appalachians, Spackman and Hughes (1995) found that 90% of plant species surveyed were represented within 15 m (49 ft) of the stream.

Many floodplain plants require regular cycles of flooding for seed dispersal and germination. Dam regulation of the Savannah River has desynchronized the conditions necessary for germination of tupelo and cypress seeds (Schneider et al 1989; Sharitz et al 1990). As a result, these once-dominant species are no longer reproducing effectively, which may ultimately lead to a shift in the forest composition. Similarly, dam-altered flow regimes have prevented regeneration of cottonwood trees in areas of the western United States (Poff et al 1997).

In terms of vegetation required for terrestrial wildlife, it is almost axiomatic that native plants are necessary to support healthy populations of native species. Studies have shown that pine plantations and other monoculture or nonnative vegetation tend to support a lower abundance and diversity of wildlife (e.g., Dickson 1978). Native riparian vegetation should always be protected and restored when necessary. Preserving the natural hydrology of the stream system will also help preserve native plants.

Riparian Buffers as Movement Corridors

One of the incidental benefits frequently ascribed to riparian buffers is their use as movement corridors for terrestrial wildlife. Riparian corridors may be more suitable in this role than other types of corridors because they tend to be environmentally diverse (Cross 1985). However, there has been considerable debate concerning whether animals actually use corridors and whether corridors should be a conservation priority. Reed Noss (1983, 1987) has been a strong advocate of movement corridors for connecting preserves and maintaining genetic exchange between animal populations. Simberloff and Cox (1987) and Simberloff et al (1992) have pointed out that corridors have some potential negative consequences and are not always the wisest use of conservation funds. A lack of empirical research on both sides of the issue has prevented resolution of the debate.

Harrison (1992) suggested minimum corridor widths for migration of large mammals, but the scale of his recommendations (0.6 to 22 km wide) is not appropriate for most riparian corridors. Machtans et al (1996) examined songbird abundance in 100 m (328 ft) wide buffer strips adjacent to clearcut forest to determine whether birds used them as movement corridors. They found that juveniles do use the corridors for dispersal, but that the adults in the buffer are probably residents. Given the lack of consensus and research on the use of riparian buffers as movement corridors, it is more defensible to base buffer width on habitat requirements of terrestrial organisms. Because there is general agreement that riparian buffers offer important high-quality habitat, there is little need to debate their merits as movement corridors at this time.

Summary and Recommendations

While narrow buffers offer considerable habitat benefits to many species, protecting diverse terrestrial riparian wildlife communities requires some buffers of at least 100 m (~300 ft). Bird abundance and diversity may be high in impacted areas, but sensitive interior-dwelling species will be lost unless some wide riparian tracts are preserved. To provide optimal habitat, native forest vegetation should be maintained or restored in all buffers. Riparian buffers may also serve as movement corridors, but considering the contentiousness of this issue it is most defensible to base buffer width on habitat requirements.

However desirable they might be, however, 300 ft wide buffers are not practical on all streams in most areas. Therefore, minimum riparian buffer width should be based on water quality and aquatic habitat functions. This should result in an abundance of narrow riparian corridors that will offer good habitat for many terrestrial species. In addition, at least a few wide (300-1000 ft/~90300 m) riparian corridors and large blocks of upland forest should be identified and targeted for preservation. This will provide habitat for those species that rely on areas of interior forest. Protection of these wide riparian corridors for terrestrial wildlife should be a part of an overall habitat-protection plan for the county or region.

Flood Control and Other Riparian Buffer Functions

Flooding is a natural feature of aquatic and riparian ecosystems. The frequency, duration and magnitude of floods helps to determine both the physical and biological characteristics of the riparian zone (Junk et al 1989). As discussed above, many riparian plants rely on cycles of flooding for seed dispersal and recruitment, while many fish species use riparian zones as nurseries, spawning grounds or feeding areas during high flows. A healthy riparian zone and a healthy stream system requires the maintenance of the natural flow regime (Poff et al 1997).

Of course, while floods are good for the stream and the riparian zone, they can be very damaging to human structures and activities. Removal of riparian vegetation, drainage of wetlands and development of floodplains leads to larger magnitude floods that cause greater damage to property (Poff et al 1997, FIFMTF 1996). Michener et al (1998) reported that flooding in South Georgia in 1994 and 1997 was greatly ameliorated by the largely intact natural riparian areas. Riparian wetlands are especially valuable for flood water storage.

Other factors can exacerbate flooding and need to be considered. Channelization, although in many cases conducted for flood control purposes, can actually increase the magnitude of flooding downstream (Roseboom and Russell 1985, Poff et al 1997). The Federal Interagency Floodplain Management Task Force now discourages such structural controls on flooding and promotes the preservation of floodplains in a natural state (FIFMTF 1996). Impervious surfaces also greatly increase stream storm flows, as discussed in Section VID.

To provide maximum protection from floods and maximum storage of flood waters, a buffer should include the entire floodplain. Short of this, the buffer should be as wide as possible and include all adjacent wetlands.

As outlined in the introduction, riparian buffers perform a number of other important functions, such as providing recreational and aesthetic benefits. These are beyond the scope of this document, although some of the economic benefits of buffers are discussed in a separate paper.

VI. Development of Riparian Buffer Guidelines

From the literature discussed above, it is relatively easy to recommend guidelines for buffer extent and vegetation:

Extent: Buffers should be placed on all perennial, intermittent and ephemeral streams to the maximum extent feasible. The overall effectiveness of the buffer is a function of how many stream miles are included. A good practical goal is to protect all perennial streams as well as all intermittent streams of second order and higher.

Vegetation: Buffers should consist of native forest along the stream to maintain aquatic habitat. Further from the stream (at least 25-50 ft), some harvesting of trees may be permissible and an outer belt of mowed grass can be useful for retaining nutrients and dissipating the energy of runoff.

Width, however, is somewhat more problematic. Although some buffer functions do not demand great width, others (especially removal of sediment) require significant width under some conditions. Current regulations in Georgia mandate fixed width buffers, regardless of topographic conditions or other factors. However, it is evident from the discussion so far that numerous variables influence buffer function. The question is, which of these variables are the most important? Is it possible to incorporate the most significant factors into a variable-width formula? To help answer these questions, several previously developed models and formulae for describing buffer function or determining buffer width are reviewed.

A. Review of Models to Determine Buffer Width and Effectiveness

Phillips (1989a, 1989b) derived two equations to describe buffer performance, both of which compare a given buffer with a reference buffer. The first (the Hydraulic Model) focuses exclusively on overland flow of sediments and sediment-bound contaminants:

$$B_b/B_r = (K_b/K_r)(L_b/L_r)0.4(s_b/s_r)\text{-}1.3(n_b/n_r)0.6$$

Where B=buffer effectiveness, K=saturated hydraulic conductivity, L=width of buffer, s=slope, and n=Manning's roughness coefficient. Subscripts b and r denote the buffer in question and the reference buffer, respectively.

The second formula, the Detention Model, considers both overland flow and subsurface flow.

$$B_b/B_r = (n_b/n_r)0.6(L_b/L_r)2(K_b/K_r)0.4(s_b/s_r)\text{-}0.7(C_b/C_r)$$

Where C= soil moisture storage capacity and the other variables are the same as in the above equation.

As noted by Muscutt et al (1993), Phillips did not verify his models experimentally, nor were they field-tested or calibrated. It is also important to note that the equations are only as good as the reference buffer selected for comparison. The parameters for reference buffers in Phillips' studies were not based on real reference sites, but rather were based on typical recorded values from the regions under study. Phillips admits his choices are "somewhat arbitrary" (Phillips 1989b). Although his second model is designed to address nitrogen removal, many factors influencing denitrification and vegetative uptake of nutrients (the two major mechanisms for nitrogen reduction) are ignored. Thus, according to the model, wetland areas are poor riparian buffers, a prediction that runs counter to both scientific research and common sense. Nevertheless, Phillips' model may represent a good starting point and could prove somewhat useful if experimentally verified and calibrated.

Despite the limitations of Phillips' model, Xiang (1993, 1996) and Xiang and Stratton (1996) used Phillips' detention model as a basis for a series of studies using a Geographic Information System (GIS) to delineate buffers in North Carolina. Although these studies provide an excellent example of the utility of GIS for large-scale delineation and study of buffers, they are still based on an untested formula.

The Riparian Ecosystem Management Model simulates the daily processing of water, sediment, carbon, and nutrients in a three-zone buffer system (Lowrance 1998). It is a computer simulation that allows buffer managers to determine the water quality impacts of buffer systems of different widths, slopes, soils, and vegetation. So far it has been tested and calibrated at sites near Tifton, Georgia, in the Coastal Plain, and additional verification is planned for other sites around the country. Initial testing revealed that REMM is accurate at predicting buffer function under many conditions, but at times appreciable error was observed (Lowrance 1998).

REMM is probably the most detailed and realistic model of riparian buffer function developed so far. Once it has been tested and calibrated for different regions, it should be a very useful tool for determining buffer characteristics on a site-by-site basis. Sensitivity analyses may also reveal which environmental factors are the most significant in different areas, and this information could be used to develop a simpler model that could be applied county- or municipality-wide. At this point, however, REMM is too data-intensive to be useful for policy purposes.

Williams and Nicks (1988) used the Chemical, Runoff, and Erosion from Agricultural Management Systems (CREAMS) model to evaluate grass filter strips of widths of 3-15 m, slopes of 2.4-10% and various roughness coefficients on a 1.6 ha wheat site. They concluded that CREAMS "can be a useful tool for evaluating filter strip effectiveness in reducing sediment yield." The authors only conducted one experimental verification, however, which showed moderately close correlation (erosion was overestimated by 38%). Flannagan et al (1989) found that under ideal conditions, CREAMS can effectively predict sediment deposition in grass filter strips. The authors developed a simplified version of the model with extremely good correlation (r^2= 0.99) to the original. In a later study, Williams and Nicks (1993) used CREAMS and WEPP (Water Erosion Prediction Project) to evaluate the effectiveness of riparian buffers of 20-30 m at 200 Conservation Reserve Program sites in Central and Eastern U.S. Only selected results were reported. Predictions from WEPP and CREAMS varied, sometimes greatly: in one case, WEPP predicted an 85% reduction in soil loss while CREAMS predicted a 10% reduction. No attempt was made to verify predictions with field observations.

Mander (1997) also developed a model for buffer width:

$$P=(tqfi^{1/2})/(mK_i n)$$

Where P= buffer width, t= a conversion constant (0.00069), q= the mean intensity of overland flow, f= either the distance between stream and watershed boundary or the ratio of catchment area to stream segment length, i= slope, m= roughness coefficient (not Manning's), K_i= water infiltration rate and n= soil adsorption capacity.

At this time, there does not appear to be any published verification of Mander's model.

Nieswand et al (1990) determined that slope and width were the main factors influencing the effectiveness of buffers in trapping sediment and associated pollutants. They developed a simple formula for determining width based on a modified Manning's equation:

$$W= k(s^{1/2})$$

Where W= width of buffer in feet

k = 50 ft (constant)

s = percent slope expressed as a whole number (e.g., 5% slope=5)

The constant "50 ft" is somewhat arbitrary; it was chosen based on common buffer recommendations, with the assumption that a 50 ft buffer at one percent slope provides adequate protection to streams. The authors also recommend that slopes greater than 15% and impervious surfaces are ineffective and should not be credited in buffer width calculations (Nieswand et al 1990).

An unusual system of determining buffer width was developed by Budd et al (1988) for a county east of Seattle, Washington. While not a formula or model as such, it is worth mentioning because it purports to consider various stream corridor variables. The method for width determination involves a subjective evaluation of stream buffer characteristics, such as erosion potential, wildlife habitat quality, etc., along with the threats to the stream segment. Based on these factors, the assessor recommends a width for protected buffers, much as a doctor recommends a prescription for a patient. It is unclear, however, how the assessor actually determines the width. No rules or guidelines are supplied. When Budd et al (1988) applied this method to Bear-Evans Creek in Washington, they almost invariably recommended a width of 50 feet, regardless of local conditions. Clearly the results of such a survey will reflect the biases of the assessor, and without guidelines such a protocol is of little practical use.

It is evident that none of these models are appropriate for delineating riparian buffers at the county scale. Some are too data-intensive to be easily applied on large scale, some have not been properly field-tested or calibrated, some do not account for factors influencing significant processes, and some yield inconsistent results with one another. A new, simple formula is needed. The next section considers what variables should be incorporated into this formula.

B. Factors Influencing Buffer Width

It is evident from the preceding sections that there are a range of variables that influence the effectiveness of buffers. These include:

- slope of banks and areas contributing flow to the stream segment
- rainfall
- soil infiltration rate (saturated hydraulic conductivity) and other soil factors (redox potential, pH, temperature)
- soil moisture content
- floodplain width
- catchment size
- land use
- impervious surfaces
- vegetation, including litter and other surface cover characteristics (often quantified as a roughness coefficient such as Manning's N)

Clinnick (1985) identified soil type, slope and cover factors as important variables. Binford and Buchenau (1993) thought the most important factors were catchment size, slope, and land use. Fennessy and Cronk (1997) identified detention time as the most important variable; this is actually an aggregate of several variables, including slope, soil factors, surface cover characteristics, hydrological factors, and others. Osborne and Kovacic (1993) suggested that factors influencing nutrient removal efficiency of buffers include sedimentation rates, drainage characteristics, soil characteristics (i.e. redox potential), organic matter content and type, temperature, successional status and nutrient loading rates.

The following is a discussion of each of these factors and considerations on the practicality of incorporating them into a variable-width buffer formula that can readily be applied county-wide. As will be seen below, in many cases it is clear that some factors are important, but practical considerations make it difficult to incorporate them on a large scale. The purpose of this paper is to develop guidelines that are not only scientifically defensible and reasonably accurate, but that can also be readily applied to any property with minimal effort and data collection. When it is possible to conduct a detailed analysis of on-site conditions to determine the optimal buffer for a specific tract of land, additional variables should be considered. In that case, a more accurate model, such as REMM, should be used.

Slope

The slope of the land on either side of the stream may be the most significant variable in determining effectiveness of the buffer in trapping sediment and retaining nutrients. The steeper the slope, the higher the velocity of overland flow and the less time it takes nutrients and other contaminants to pass through the buffer, whether attached to sediments or moving in subsurface flow. Slope is a variable in virtually all models of buffer effectiveness and should definitely be included in a formula for buffer width.

Although Nieswand et al (1990) make a case for a width that varies exponentially with slope, research by Trimble and Sartz (1957) and Swift (1986) found a linear relationship in their field studies. Trimble and Sartz suggested that width should increase by either two or four feet for each percent increase in slope; Swift suggested that width should increase either 0.40 or 1.39 feet for each percent increase in slope. Since Swift ignored small silt and clay particles, his variables are apt to be low. Therefore, this review follows Trimble and Sartz' recommendation of increasing the buffer by 2 ft per 1% increase in slope.

Many researchers have noted that very steep slopes cannot effectively remove contaminants, though there is debate over what constitutes a steep slope. Among the recommendations are:

- 40% slope (Cohen et al 1987)
- 25% slope (Schueler 1995a)
- 15% slope (Nieswand et al 1990)
- 10% slope (Herson-Jones et al 1995)

Georgia's minimum standards for Mountain Slope Protection apply to certain slopes over 25%. Soil surveys typically do not recommend agriculture on slopes over 10% because of the erosion hazard. On the other hand, Swift found that riparian buffers on logging roads were able to trap sediment even on extremely steep (80%) slopes, though again small particles are not considered. There appear to be no other studies which evaluate buffer effectiveness at greater than moderate slopes. Any cutoff will be somewhat arbitrary, but 25% appears to be reasonable given the range shown above, until further research can clarify the issue. Therefore, the buffer width should increase by two feet for each slope percent up to 25%. Slopes steeper than this are not credited toward the buffer width.

Rainfall

The pattern and intensity of rainfall are important factors in determining the effectiveness of buffers. Daniels and Gilliam (1996) found that most of the sediment that passed through a riparian buffer did so during a single storm. One study cited by Karr and Schlosser (1976) found that 75% of agricultural erosion occurs during four storms a year, but another study they cite found that smaller rain events caused at least 50% of erosion and there was significant regional variability. It would be expected that in regions where rainfall is uniform and light, narrower buffers may effectively manage most of the sediment and nutrients that enter them. In areas that experience seasonal storms of high intensity, wider buffers may be necessary. However, there do not appear to be any studies that quantify the relationship between rainfall and buffer effectiveness. Several studies (e.g. Dillaha et al 1988, 1989; Magette et al 1987, 1989) described in this review simulated heavy rainfall conditions on test plots, but the studies were short-term and rainfall intensity comparisons were not made. Magette (1987) reported that buffer effectiveness declined as rainfall events increased. Others (Cooper et al 1987, Lowrance et al 1988) have examined the effectiveness of buffers over a sufficiently long time frame to include large storms. These long-term studies indicated the need for wider buffers than were recommended by most short-term studies. Groffman et al (1991b) suggested that denitrification rates are lower during storms because buffer residence times are decreased, but no empirical evidence was available. Hanson et al (1994) reported increases in denitrification rates in response to storms. Precipitation is incorporated into the "R" factor of the Universal Soil Loss Equation, which provides a rough estimate of erosion from agricultural fields or other plots. It may be possible to use this factor in a formula for buffer width; this warrants further investigation.

Buffers should be designed to effectively handle runoff and subsurface flow-rates from a one-year storm event. Just as for other stormwater best management practices, allowances should be made for exceptional (10-year or 25-year) events. In the absence of hard data, however, it is not possible to draw a valid relationship between rainfall patterns and buffer width. When possible, buffer effectiveness should be assessed through stream water quality measurements during or after storms. When buffers are found to be ineffective they should be widened or additional on-site controls should be implemented.

Catchment Size/Hydraulic Loading

It is logical that an increase in catchment size will demand an increase in buffer width. That is, a stream segment that drains five acres will collect more pollutants than one draining two acres, and a larger buffer will be required. The relationship may not be so simple, however. Sorrano et al (1996) argue that sediment/nutrient loading models that directly correlate loading rate with catchment area ignore the fact that as distance from the stream increases, sediments and nutrients are less likely to actually enter surface water. In other words, the areas closest to stream channels are far more important than more distal areas, and an increase in contributing area does not necessarily correspond to an increase in contaminant loading. Hatfield et al (1995) reported that catchment size had little influence on pesticide removal by buffer strips.

Additionally, denitrification rates usually increase to accommodate increased nitrate loading rates, as long as carbon does not become limiting. Haycock and Pinay (1993) reported 99% nitrate reductions in 20 m (66 ft) wide wooded riparian buffers regardless of the level of nitrate loading. Mander et al (1997) found that nutrient retention bears a strong log-log relationship with nutrient load; i.e., as load increases, retention increases (They also make a rather unconvincing case that retention efficiency declines slightly with higher loads). It appears that increased nutrient and/or hydraulic load does not necessarily require a wider buffer. In any case, the relationship is sufficiently complex that catchment size is not a reasonable variable to include in a simple buffer model.

Soil Factors

Soil characteristics determine in large part whether or not overland flow occurs, how fast water and contaminants move to the stream, and other factors relevant to the effectiveness of the riparian buffer. Denitrification rates are strongly influenced by soil moisture and soil pH (Groffman et al 1991a,b).

However, determining soil characteristics on a county-wide scale is somewhat problematic. According to Steve Lawrence of the Natural Resources Conservation Service, soil survey maps may not be sufficiently accurate for application in a model for buffer width (pers. com.). The minimum mapping unit is 3-4 acres, and inevitably some “inclusions” occur: these are small areas of different soil type lumped in with the dominant soil type. Mapping accuracy is even lower for soils that have been disturbed by construction or other activities (Elizabeth Kramer, pers. com.). Therefore, without detailed and potentially expensive on-site soil analyses, it is unlikely that including soil factors as variables would add greatly to the accuracy of a model. Of course, it may be reasonable to consider very general soil characteristics, such whether a soil is hydric (frequently flooded) and the overall hydrology of the area. In cases where on-site analysis is possible, it may be reasonable to adjust buffer width accordingly.

Soil Moisture & Wetlands

Denitrification rates show a positive correlation with soil moisture content (e.g. Groffman et al 1991a, b, Hanson et al 1994, Schnabel et al 1997). Wetlands, those soils with the highest moisture levels, have long been recognized for their value in trapping sediment and nutrients. They are also recognized as important animal habitat and are valuable in reducing flood impacts.

Riparian wetlands are significant enough to merit automatic inclusion in a buffer system. The width of the buffer should be extended by the width of all adjacent wetlands. For example, if a site that would otherwise have a 75 ft (22.9 m) wide buffer is found to include part of a 50 ft (15.2 m) wide area of riparian wetlands, total buffer width should be extended to 125 ft (38 m). Constructed wetlands are becoming more common as a component in human and animal waste treatment systems. However, natural wetlands require buffers of their own and should never be used to process untreated waste (Lowrance 1997b, Hubbard 1997).

Floodplain

The floodplain represents the region of material interchange between land and stream, as well as the limits of stream channel migration. Studies reviewed above have shown that the entire floodplain can be a site of significant contaminant removal. For this reason, it makes sense to extend the buffer to the edge of the floodplain whenever possible. In their buffer guidelines for Willamette National Forest, Gregory and Ashkenas (1990) declare that “the riparian management zone should include the entire [100 year] floodplain. Failure to do so will seriously jeopardize the riparian management objectives during major floods.” Schueler (1995) also recommends including the floodplain.

Including the entire floodplain is, naturally, also the best way to minimize damage from floods.

Therefore, whenever feasible, the riparian buffer should be extended to the edge of the 100-year floodplain. Even when this is not possible, certain activities and structures should be excluded from the floodplain because of the risk they pose to the stream. These include animal waste lagoons, animal waste spray fields, hazardous and municipal waste disposal facilities, and other potential sources of severe contamination.

Land Use

Urban and agricultural watersheds experience greater sedimentation and eutrophication than forested watersheds (Crawford and Lenat 1989). A study in coastal South Carolina found that a stream draining an 11 ha urbanized watershed had a 66% greater sediment load than a stream draining a 37 ha forested watershed, despite its smaller catchment area (Wahl et al 1997). Studies by Crawford and Lenat (1989) clearly showed that for all indicators (sediment, nutrients, metals, fish, invertebrates), urban streams are more heavily impacted than either forested or agricultural streams. Agricultural watersheds also display serious impacts, although they can still retain a healthy (if altered) biota (Crawford and Lenat 1989). A recent study by the U.S. Geological Survey found that Piedmont, Georgia streams draining watersheds that are mostly forested maintain the healthiest fish communities. Agricultural and suburban streams are worse, and urban streams tend to be the most degraded.

Incorporating land use into a formula for riparian buffer width presents some practical difficulties, however, because the relationship is quite complex. For example, even though urban streams tend to suffer greater impacts than other streams, urban buffers also tend to be less effective because storm drains deliver a large proportion of runoff directly to the channel. Therefore, widening a buffer in an urban area may have less of an effect on water quality than widening a buffer in an agricultural area. In a similar vein, does a pristine stream running through a forest need a smaller or larger buffer than an agricultural stream? Although the stream may not appear to be threatened, the absence of a sufficiently wide buffer might allow a logging road to be built too close, damaging the stream with sediment. Furthermore, land uses of the same general category (e.g., farming) could have very different effects on a stream. For example, one property zoned agricultural might be planted to cotton and produce massive sediment loads, while another might be planted to unmanaged pine trees. Administration of such a program would be difficult and subject to frequent challenges.

A more practical approach is to establish riparian buffer widths that are sufficiently wide to mitigate the great majority of land use impacts. Specific activities that are especially damaging should be subject to additional setbacks. In addition, pollution should be managed on-site, impervious surfaces should be limited and riparian buffer bypasses should be minimized (see below). These controls may do more to improve stream water quality and habitat than additional increases in the riparian buffer width.

Impervious surfaces

Because impervious surface area is so closely correlated with stream water quality, it may be considered as a variable for determining buffer width. It is far more effective, however, to treat impervious surface as a controllable variable and implement impervious surface limits and controls. This is discussed in more detail in a later section. In any case, however, preexisting impervious surfaces near the stream will not effectively perform buffer functions and should not count toward buffer width. For example, if a 30 ft wide road parallels a stream, the riparian buffer should be increased by 30 ft on the road side.

Vegetation

Vegetation characteristics may influence buffer effectiveness and therefore necessary width. However, in this report vegetation is considered a factor under management and not a width variable.

C. Buffer Guidelines for Water Quality Protection

In the previous section it was established that buffer width should vary based on slope and should include wetlands. One final task remains before buffer guidelines are presented: to determine the minimum width of the buffer. Without considering terrestrial habitat, most recommendations for minimum buffer widths range from 15 m (~50 ft) to 30 m (~100 ft). It might be possible to determine the correct width from within this range by conducting additional research in the region of interest. In the absence of this, however, the choice of minimum width amounts to a choice regarding margin of safety or, conversely, acceptable risk. The greater the minimum buffer width, the greater the margin of safety in terms of water quality and habitat preservation. Accordingly, several options are proposed: The first two are variable-width options, one with a 100 ft base width, and one with a 50 ft base width. The first can be considered the "conservative" option: it meets or exceeds many buffer width recommendations, and therefore should ensure high water quality and support good habitat for native aquatic organisms. The second is the "riskier" option: it should, under most conditions, provide good protection to the stream and good habitat preservation, although heavy rain, floods, or poor management of contaminant sources could more easily overwhelm the buffer. All of these options are defensible given the literature reviewed here. As a third option, a 100 ft fixed-width riparian buffer is recommended for local governments that find it impractical to administer a variable-width buffer.

Option One:

- Base width: 100 ft (30.5 m) plus 2 ft (0.61 m) per 1% of slope.
- Extend to edge of floodplain.
- Include adjacent wetlands. The buffer width is extended by the width of the wetlands, which guarantees that the entire wetland and an additional buffer are protected.
- Existing impervious surfaces in the riparian zone do not count toward buffer width (i.e., the width is extended by the width of the impervious surface, just as for wetlands).
- Slopes over 25% do not count toward the width.
- The buffer applies to all perennial and intermittent streams. These may be defined on the basis of USDA soil survey maps, USGS topographic maps,or other method which most accurately represents true conditions.

Option Two:

The same as Option One, except:

- Base width is 50 ft (15.2 m) plus 2 ft (0.61 m) per 1% of slope.
- Entire floodplain is not necessarily included in buffer, although potential sources of severe contamination be excluded from the floodplain.
- The buffer applies to all perennial and intermittent streams. These may be defined on the basis of USDA soil survey maps, USGS topographic maps,or other method which most accurately represents true conditions.

Figure 9 shows an example of how Option Two can be applied to a theoretical riparian landscape.

Option Three:

- Fixed buffer width of 100 ft.
- The buffer applies to all perennial and intermittent streams. These may be defined on the basis of USDA soil survey maps, USGS topographic maps,or other method which most accurately represents true conditions.

All of the buffer options described will provide habitat for many terrestrial wildlife species. However, significantly wider buffers are necessary to provide habitat for forest interior species, many of which are species of special concern. The most common recommendation in the literature on wildlife (most of which focuses on birds) is for a 100 m (300 ft) riparian buffer. Although this is not practical in many cases, local governments should preserve at least some riparian tracts of 300 foot width or greater. Identification of these areas should be part of an overall, county-wide wildlife protection plan.

Activities Prohibited in the Buffer

As a general rule, all sources of contamination should be excluded from the buffer. These include:

- land disturbing activities
- impervious surfaces
- logging roads
- mining
- septic tank drain fields
- agricultural fields
- waste disposal sites
- application of pesticides and fertilizer (except as necessary for buffer restoration)
- livestock

One exemption to this list that local governments may wish to consider is construction of a single family home. Minimum standards for river corridor protection issued by the Environmental Protection Division cannot by law prohibit the building of a single-family dwelling within the buffer for protected River Corridors (OCGA 12-2-8). Local governments that develop ordinances more stringent than the minimum standards may also wish to make this exemption.

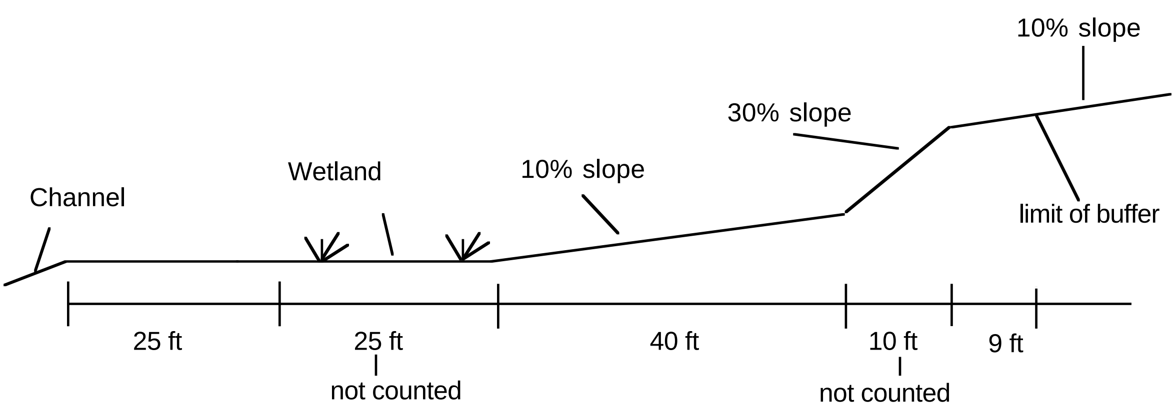
The Three-Zone Buffer System

A three-zone riparian buffer system has been suggested for agricultural areas to allow some limited use of riparian land while preserving buffer functionality (Welsch 1991). Zone one, which extends from the bank to 15 ft (4.6 m) within the buffer, is undisturbed forest. Zone two is a managed forest, beginning 15 ft (4.6 m) from the bank and extending to 75 ft (22.9 m). Periodic harvesting and some disturbance is acceptable within this zone. Zone 3 is a grassed strip, beginning 75 ft (22.9 m) from the bank and extending to the buffer's edge at 95 ft (29.0 m). Controlled grazing and mowing may be permitted in this zone.

While the three-zone system represents a good compromise for buffers on agricultural land, it introduces an added level of complexity to a buffer ordinance that may not be warranted, especially if a variable-width system is used. Local governments may want to encourage the three-zone system as a voluntary practice on

Figure 9. Applying a Flexible Width Buffer.

This diagram illustrates how Buffer Option 2 is applied to a hypothetical landscape. The average slope of the stream valley here is 12%, which means the buffer should be 50+24= 74 ft wide. The width of the wetland and the steep slope are added to the total width, so the buffer actually covers some 109 ft. If an impervious surface were present, its width would also be added to the total.



agricultural lands. Additional information is available from the Natural Resources Conservation Service.

Is This Possible?

An ordinance that establishes 100 ft or wider buffers on all perennial streams may sound unrealistic or too heavy-handed for most local governments. But such an ordinance is not as draconian as it first sounds. It is important to bear in mind that in most areas, such land use laws must of necessity exempt existing land uses: no local government is going to tell a small property owner that he must move his house or convert his lawn to forest (although he could be actively encouraged to do the latter). The people who are most affected are developers, who must now incorporate buffers into their designs. This will not necessarily have a negative economic impact. Several studies have shown that people will pay a premium to live or work near greenways or other protected areas, and this allows the developer to recoup at least some of the costs of not developing up to the stream bank. Finally, any buffer ordinance should always include clear, fair rules for variances, which will insure that anyone who is unfairly impacted by the law can obtain relief. More information on how local governments can develop and implement riparian buffer ordinances is included in a separate "Guidebook for Developing Local Riparian Buffer Ordinances," available from the University of Georgia Institute of Ecology Office of Public Service and Outreach. The document discusses various tools for protecting buffers, case studies of existing buffer protection programs, important issues of concerns such as "takings," and includes model riparian buffer ordinances.

D. Other Considerations

Establishing a system of protected riparian buffers is an important step in preserving healthy streams. However, a number of other steps must be taken if buffers are to be truly effective.

Reducing Impervious Surfaces

In a natural forested watershed, overland flow is quite rare, occurring only during the most severe rainstorms. Impervious surfaces, on the other hand, transfer most precipitation into runoff, leading to increased surface erosion, higher and faster storm flows in streams, and increased channel erosion. As a consequence, urban streams characteristically have greatly elevated sediment levels (Wahl et al 1997, Frick et al 1998). Flow from impervious surfaces also carries pollutants directly to streams, bypassing the natural filtration that would occur by passage through soil. Impervious surfaces are so closely correlated with urban water pollution that they are commonly used as an indicator of overall stream quality (Arnold and Gibbons 1996). May et al (1997) note that impervious surfaces are the "major contributor to changes in watershed hydrology that drive many of the physical changes affecting urban streams." Trimble (1997) ascribed the cause of large-scale channel erosion in San Diego Creek to increased impervious surfaces in the watershed. Impervious surfaces also decrease groundwater recharge and stream base flow levels (Ferguson and Suckling 1990). In a study of Peachtree Creek in Atlanta, Ferguson and Suckling (1990) also linked impervious surfaces to an increase in evapotranspiration; water evaporates quickly from impervious surfaces, creating a warm microclimate which increases transpiration rates in trees and plants. This further reduces stream flows, except during rainstorms. In short, impervious surfaces cause "flashy" streams with low base flows and very high storm flows.

Riparian buffers cannot protect a stream from channel erosion if the stream is constantly scoured by high storm flows caused by runoff from impervious surfaces. All municipalities and counties experiencing urban and suburban growth should consider enacting impervious surface controls in addition to riparian buffer ordinances. These limits can range from 10-12%, the point at which streams are considered impacted, to 30%, the point at which streams can be considered degraded (Klein 1979). If existing technologies were vigorously applied, impervious surfaces could be nearly eliminated (Bruce Ferguson, pers. com.). Further information on reducing impervious surfaces is available in the publication *Land Development Provisions to Protect Georgia Water Quality* (UGASED 1997) and in a recent publication of the Etowah Initiative (Miller and Sutherland 1999).

On-Site Management of Pollutants

Riparian buffers alone are not enough to mitigate the effects of otherwise uncontrolled upland activities (Binford and Buchenau 1993). As Barling (1994) notes, “buffer strips should only be considered as a secondary conservation practice after controlling the generation of pollutants at their source.” In many cases it may be easier, cheaper and preferable to prevent pollutants from moving off site in the first place.

Sediment

In the case of agricultural regions, erosion reduction efforts should focus on keeping soil in fields, where it is usable, rather than trapping it in the riparian zone, where it is much more difficult to salvage. The Natural Resources Conservation Service, the Georgia Soil and Water Conservation Commission, extension agencies and other governmental and non-governmental organizations can provide detailed information on effective best management practices to reduce erosion. It is essential to follow these BMPs *in addition* to protecting functioning riparian buffer strips. Local governments need to take a coordinating role in ensuring that the various agencies and the agricultural community cooperate to reach water quality goals for the basin.

Likewise, BMPs must be faithfully implemented and enforced in construction projects. A review by Brown and Caraco (1997) found that in many cases, half of all practices specified in erosion and sediment control (ESC) plans were not implemented correctly and were not working. Contractors habitually saved money by cutting ESC installation and maintenance. Surveys also found that ESC practices rated as “most effective” by experts were seldom applied while those rated “ineffective” are still widely used. The authors also report that a field assessment of silt fences found that 42% were improperly installed and 66% were inadequately maintained. They conclude that while a substantial amount of money is now spent on ESC practices, “much of this money is not being well spent— practices are poorly or inappropriately installed, and very little is spent on maintaining them” (Brown and Caraco 1997). Kundell and Rasmussen (1995) have noted the importance of inspections and enforcement of BMPs in Georgia.

Nutrients

Because riparian zones can become saturated with phosphorus, it is very important to manage sources of this nutrient. Septic drain fields and sewer pipes can leak soluble phosphorus and should be located as far from streams as possible. Frequently, however, sewer pipes are routed through stream corridors, creating an extreme hazard if they should leak. Although a review of setback recommendations for septic tank drain fields and sewer pipes is beyond the scope of this document, a minimum distance of 100 ft (~30m) appears prudent considering the magnitude of the risk. Sewer pipes should only cross streams when absolutely necessary.

Concentrated Animal Feeding Operations (CAFOs) typically dispose of large amounts of nutrient-rich waste by holding it in waste lagoons and applying it to fields either as fertilizer or simply as a disposal method (Linville 1997). Large CAFOs may be considered point source polluters and can be required to obtain a federal permit (issued in Georgia by the EPD), but nationally only 12% of CAFOs are actually permitted (Linville 1997). Waste lagoon spills can be devastating to rivers (Burkholder 1997, Linville 1997), and placement of such lagoons should be carefully regulated. It is probably best to determine placement on a site-by-site basis to ensure that if a spill occurs, its effect on the stream system will be minimized. At the least, lagoons should be located outside of riparian buffers and the 100-year floodplain.

Because riparian buffers are generally effective at removing nitrogen from animal waste (Groffman et al 1991a), manure application strategies should be based on phosphorus, especially in watersheds where it is a limiting nutrient (Daniel and Moore 1997, Miguel Cabrera, pers. com.). Studies by Miguel Cabrera have shown that the concentration of phosphorus in runoff is proportional to the amount applied to the field, and that current application rates are many times higher than they should be to maintain phosphorus at acceptable levels (pers. com.). New approaches to phosphorus management are gaining acceptance and hold some promise for reducing the amount of phosphorus that can reach the stream. However, even with the best available controls, manure application rates likely will need to be reduced substantially to prevent phosphorus pollution of streams (Miguel Cabrera, pers. com.).

Riparian Buffer Crossings/Bypasses

Road crossings and other breaks in the riparian buffer effectively reduce buffer width to zero and allow sediment and other contaminants to pass directly into the stream (Swift 1986). Buffer crossings, or even just narrow points in the buffer, may be the locations of the majority of contaminant transport to the stream (Weller et al 1998). All buffer crossings should be minimized, but when they are necessary, Schueler (1995) suggests the following guidelines:

- Crossing width should be minimized
- Direct (90 degree) crossing angles are preferable to oblique crossing angles
- Construction should be capable of surviving 100-year floods
- Free-span bridges are preferable to culvertizing or piping the stream

Special care must be taken to stabilize banks around the buffer crossing. Crossings should be regularly monitored, especially after severe storms and floods, to determine if excessive sedimentation is occurring. Sewer lines which cross streams should also be inspected to ensure they are not leaking or damaged in any way.

It is also essential to minimize practices which cause water flow to bypass the riparian zone. Drain tiles used to improve drainage from agricultural fields discharge flow directly into the stream (Fennessy and Cronk 1997, Osborne and Kovacic 1993, Vought et al 1994). Jacobs and Gilliam (1985) compared fields drained by a riparian buffer with fields drained by ditches and drain tile. They observed high nitrate reduction in the riparian buffer, but much lower nitrate loss in drainage ditches and very little nitrate removal for fields drained by tile. Nitrate levels in tile drains in Georgia agricultural fields have been found to be several times higher than the levels in the shallow aquifer (Frick et al 1998). Constructing riparian wetlands at the outfall of the drain tile would help to slow the transport of pollutants into the stream and permit nutrient uptake and removal (Osborne and Kovacic 1993).

Similarly, in urban areas, storm drains carry contaminant-laden water from impervious surfaces directly into streams. This practice should be discontinued. Ideally, runoff should be allowed to infiltrate into the soil as close as possible to the source. If some drainage is required, outflow should either be directed in the form of sheet flow across a suitably wide riparian buffer or into a storm water detention ponds or constructed wetlands. When necessary, constructed wetlands may be incorporated into the riparian buffer if they are properly located and do not harm existing wetlands or other critical riparian features (Schueler 1995a).

For More Information

For additional information on how local governments can develop riparian buffer ordinances, a "Guidebook for Developing Local Riparian Buffer Ordinances," is available from the University of Georgia Institute of Ecology Office of Public Service and Outreach (phone: 706-542-3948; email: lfowler@uga.cc.uga.edu). For additional scientific information on riparian buffers, all of the sources cited in this review are listed in the References section which follows.

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The Environmental Consequences of Forest Roads and Achieving a Sustainable Road System

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Table of Contents

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Introduction

The Forest Service faces many challenges with its vastly oversized, under-maintained, and unaffordable transportation system. With 370,643 miles of system roads and 137,409 miles of system trails (USDA Forest Service 2019), the network extends broadly across every national forest and grassland and through a variety of habitats, ecosystems and terrains. An impressive body of scientific literature addresses the various effects of roads on the physical, biological and cultural environment. Numerous studies demonstrate the harmful environmental consequences to water, fish, wildlife, and ecosystems.

In recent years, the scientific literature has expanded to address the effects of roads on climate change adaptation and conversely the effects of climate change on roads, as well as the multiple benefits of road removal on the physical, biological and cultural environments.

The first section of this paper provides a literature review summarizing the most recent science related to the environmental impacts of forest roads and motorized trails. The second section focuses on climate change effects and strategies to address the growing ecological consequences to forest resources. The third section provides background and specific direction for the Forest Service to provide for an ecologically and economically sustainable road system, including recommendations for future action.

I. Impacts of Transportation Infrastructure and Access to the Ecological Integrity of Terrestrial and Aquatic Ecosystems and Watersheds

It is well understood that transportation infrastructure provides access to national forests and grasslands and also harms aquatic and terrestrial environments at multiple scales. In general, the more roads and motorized trails the greater the impacts. Since its emergence, the field of road ecology and the resulting research has proven the magnitude and breadth of ecological issues related to roads; entire books have been written on the topic (e.g., Forman et al. 2003, van der Ree et al. 2015), and research centers continue to expand their case studies, including the Western Transportation Institute at Montana State University and the Road Ecology Center at the University of California - Davis.[1]

Below, we provide a summary of the current understanding of the impacts of roads and motorized access on terrestrial and aquatic ecosystems, supplementing long-established, peer-reviewed literature reviews on the topic, including Gucinski et al. (2000), Trombulak and Frissell (2000), Coffin (2007), and Robinson et al. (2010). More targeted reviews have been published on the effects of roads on insects (Munoz et al. 2015), vertebrates (da Rosa 2013), and animal abundance (Fahrig and Rytwinski 2009, Benítez-López et al. 2010). Literature reviews on the ecological and social impacts of motorized recreation include Gaines et al. (2003), Davenport and Switalski (2006), Ouren

[1] See http://www.westerntransportationinstitute.org/programs/road-ecology and http://roadecology.ucdavis.edu/

et al. (2007), Switalski and Jones (2012), and, more recently, Switalski (2017). In addition to the physical and environmental impacts of roads, increased visitation has resulted in intentional and unintentional damage to many cultural and historic sites (Spangler and Yentsch 2008, Sampson 2009, Hedquist et al. 2014).

A. Impacts on geomorphology and hydrology

The construction and presence of forest roads can dramatically change the hydrology and geomorphology of a forest system leading to reductions in the quantity and quality of aquatic habitat (Al-Chokhachy et al. 2016). While there are several mechanisms that cause these impacts (Wemple et al. 2001, Figure 1), most fundamentally, compacted roadbeds reduce rainfall infiltration, intercepting and concentrating water, and providing a ready source of sediment for transport (Wemple et al. 2001). In fact, roads contribute more sediment to streams than any other land management activities on Forest Service lands (Gucinski et al. 2000). Surface erosion rates from roads can be up to three orders of magnitude greater than erosion rates from undisturbed forest soils (Endicott 2008).

Erosion and sediment produced from roads occur both chronically and catastrophically. Every time it rains, sediment from the road surface and from cut-and fill-slopes is picked up by rainwater that flows into and on roads (fluvial erosion). The sediment that is entrained in surface flows are often concentrated into road ditches and culverts and directed into streams. The degree of fluvial erosion varies by geology and geography, and increases with increased motorized use (Robichaud et al. 2010). Closed roads produce significantly less sediment than open drivable roads (Sosa Pérez and Macdonald 2017, Foltz et al. 2009).

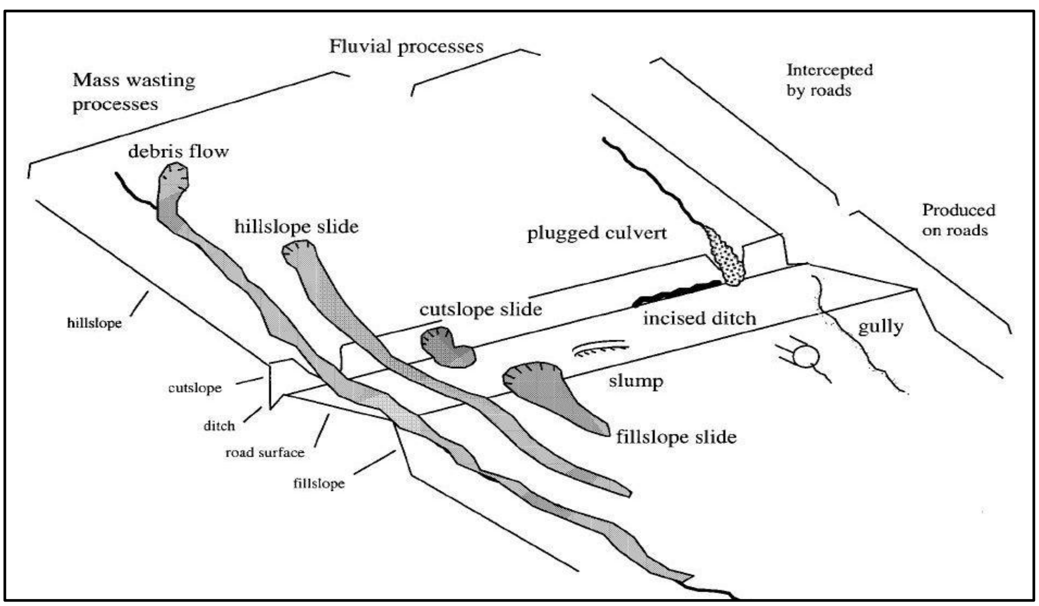

Figure 1: Typology of erosional and depositional features produced by mass-wasting and fluvial processes associated with forest roads (reprinted from Wemple et al. 2001).

Roads also precipitate catastrophic failures of road beds and fills (mass wasting) during large storm events leading to massive slugs of sediment moving into waterways (Gucinski et al. 2000, Endicott 2008). This typically occurs when culverts are undersized and cannot handle the volume of water funneled through them, or they simply become plugged with debris and sediment. The saturated roadbed can fail entirely and result in a landslide, or the blocked stream crossing can erode the entire fill down to the original stream channel.

The erosion of road- and trail-related sediment and its subsequent movement into stream systems affects the geomorphology of the drainage system in a number of ways. It directly alters channel morphology by embedding larger gravels as well as filling pools. It can also have the opposite effect of increasing peak discharges and scouring channels, which can lead to disconnection of the channel and floodplain, and lowered base flows (Gucinski et al. 2000). The width/depth ratio of the stream changes can trigger changes in water temperature, sinuosity and other geomorphic factors important for aquatic species survival (Trombulak and Frissell 2000).

B. Impacts on aquatic habitat and fish

Roads can have dramatic and lasting impacts on fish and aquatic habitat. Increased sedimentation in stream beds has been linked to decreased fry emergence, decreased juvenile densities, loss of winter carrying capacity, increased predation of fish, and reductions in macro-invertebrate populations that are a food source to many fish species (Gucinski et al. 2000, Endicott 2008). Roads close to streams reduce the number of trees available for large wood recruitment, and reduce stream-side shade (Meredith et al. 2014.) On a landscape scale, these effects add up to: changes in the frequency, timing and magnitude of disturbance to aquatic habitat and changes to aquatic habitat structures (e.g., pools, riffles, spawning gravels and in-channel debris), and conditions (food sources, refugia, and water temperature; Gucinski et al. 2000).

River fragmentation

Roads also act as barriers to migration and fragment habitat of aquatic species (Gucinski et al. 2000). Where roads cross streams, road engineers usually place culverts or bridges. Undersized culverts interfere with sediment transport and channel processes such that the road/stream crossing becomes a barrier for fish and aquatic species movement up and down stream (Erikinaro et al. 2017). For instance, a culvert may scour on the downstream side of the crossing, actually forming a waterfall up which fish cannot move. Undersized culverts can infringe upon the channel or floodplain and trap sediment causing the stream to become too shallow and/or warm such that fish will not migrate past the structure. Or, the water can move through the culvert at too high a gradient or velocity to allow fish passage (Endicott 2008).

River fragmentation is problematic for many aquatic species but especially for anadromous species that must migrate upstream to spawn. Well-known native aquatic species affected by roads include salmon such as coho (*Oncorhynchus kisutch*), Chinook (*O. tshawytscha*), and chum (*O. keta*); steelhead

(*O. mykiss*), a variety of trout species including bull trout (*Salvelinus confluentus*) and cutthroat trout (*O. clarki*), as well as other native fish and amphibians (Endicott 2008). The restoration and mitigation of impassable road culverts has been found to restore connectivity and increase available aquatic habitat (Erikinaro et al. 2017), and the quality of aquatic habitat (McCaffery et al. 2007).

C. Impacts on terrestrial habitat and wildlife

Roads and trails impact wildlife through a number of mechanisms including: direct mortality (poaching, hunting/trapping), changes in movement and habitat-use patterns (disturbance/avoidance), as well as indirect impacts including altering adjacent habitat and interference with predator/prey relationships (Coffin 2007, Fahrig and Rytwinski 2009, Robinson et al. 2010, da Rosa and Bager 2013). Some of these impacts result from the road itself, and some result from the uses on and around the roads (access). Ultimately, numerous studies show that roads reduce the abundance, diversity, and distribution of several forest species (Fayrig and Ritwinski 2009, Benítez-López et al. 2010, Munoz et al. 2015).

Abundance and distribution

The extensive research on roads and wildlife establish clear trends of wildlife population declines. Fahrig and Rytwinski (2009) reviewed the empirical literature on the effects of roads and traffic on animal abundance and distribution looking at 79 studies that addressed 131 species. They found that the number of documented negative effects of roads on animal abundance outnumbered the number of positive effects by a factor of 5. Amphibians, reptiles, and most birds tended to show negative effects. Small mammals generally showed either positive effects or no effect, mid-sized mammals showed either negative effects or no effect, and large mammals showed predominantly negative effects. Benítez-López et al. (2010) conducted a meta-analysis on the effects of roads and infrastructure proximity on mammal and bird populations. They found a significant pattern of avoidance and a reduction in bird and mammal populations in the vicinity of infrastructure. Muñoz et al. (2015) found that many insect populations have declined as well.

Direct mortality, disturbance, and habitat modification

Road and motorized trail use affect many different types of species. For example, trapping, poaching, collisions, negative human interactions, disturbance and displacement significantly impact wide ranging carnivores (Gaines et al. 2003, Table 1). Hunted game species such as elk (Cervus canadensis), become more vulnerable from access allowed by roads and motorized trails resulting in a reduction in effective habitat among other impacts (Rowland et al. 2005). Slow-moving migratory animals such as amphibians, and reptiles who use roads to regulate temperature, are also vulnerable (Gucinski et al. 2000, Brehme et al. 2013). Roads and motorized trails also affect ecosystems and habitats because they are major vectors of non-native plant and animal species (Gelbard and Harrison 2003). This can have significant ecological and economic impacts when aggressive invading species overwhelm or significantly alter native species and systems.

Table 1: Road- and recreation trail-associated factors for wide-ranging carnivores (Reprinted from Gaines et al. (2003)[2]

Focal species	Road-associated factors	Motorized trail-associated factors	Nonmotorized trail-associated factors
Grizzly bear	Poaching Collisions Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance	Poaching Negative human interactions Displacement or avoidance
Lynx	Down log reduction Trapping Collisions Disturbance at a specific site	Disturbance at a specific site Trapping	Disturbance at a specific site
Gray wolf	Trapping Poaching Collisions Negative human interactions Disturbance at a specific site Displacement or avoidance	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site
Wolverine	Down log reduction Trapping Disturbance at a specific site Collisions	Trapping Disturbance at a specific site	Trapping Disturbance at a specific site

Habitat fragmentation

At the landscape scale, roads fragment habitat blocks into smaller patches that may not be able to support interior forest species. Smaller habitat patches result in diminished genetic variability, increased inbreeding, and at times local extinctions (Gucinski et al. 2000; Trombulak and Frissell 2000). For example, a narrow forest road with little traffic was a barrier in Arizona to the Mt. Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*; Chen and Koprowski 2013). Fragmentation intensifies concerns about grizzly bear population viability, especially since roads increase human/bear interactions exacerbating the problem of excessive mortality (Proctor et al, 2012)

Roads also change the composition and structure of ecosystems along buffer zones, called edge-affected zones. The width of edge-affected zones varies by what metric is being discussed; however, researchers have documented road-avoidance zones a kilometer or more away from a road (Robinson et al.2010; Table 2). In heavily roaded landscapes, edge-affected acres can be a significant percentage of total acres. For example, in a landscape where the road density is 3 mi/mi^2 and where the edge-affected zone is estimated to be 500 ft from the center of the road to each side, the edge-affected zone is 56% of the total acreage.

[2] For a list of citations see Gaines et al. (2003).

Table 2: A summary of some documented road-avoidance zones for various species (adapted from Robinson et al. 2010).

Species	Avoidance zone m (ft)	Type of disturbance	Reference
Snakes	650 (2133)	Forestry roads	Bowles (1997)
Salamander	35 (115)	Narrow forestry road, light traffic	Semlitsch (2003)
Woodland birds	150 (492)	Unpaved roads	Ortega and Capen (2002)
Spotted owl	400 (1312)	Forestry roads, light traffic	Wasser et al. (1997)
Marten	<100 (<328)	Any forest opening	Hargis et al. (1999)
Elk	500–1000 (1640-3281)	Logging roads, light traffic	Edge and Marcum (1985)
Grizzly bear	3000 (9840)	Fall	Mattson et al. (1996)
	500 (1640)	Spring and summer	
	1122 (3681)	Open road	Kasworm and Manley (1990)
	665 (2182)	Closed road	
Black bear	274 (899)	Spring, unpaved roads	Kasworm and Manley (1990)
	914 (2999)	Fall, unpaved roads	

Migration disruption

Roads disrupt migration of large ungulates, such as elk, impeding travel at multiple scales, including seasonal home range use and migration to winter range (Buchanan et al. 2014, Prokopenko et al. 2017). For example, a recent study found migrating elk changed their behavior and stopover use on migration routes that were roaded (Paton et al. 2017). The authors suggest this disturbance may lead to decreased foraging, displacement of high-quality habitat, and affect the permeability of the migration route. In addition, roads disrupt grizzly bear movements influencing dispersal away from the maternal home range and ultimately influencing population-level fragmentation." (Proctor et al. 2018).

Oil and gas development (and associated roads) reduced the effectiveness of both mule deer and pronghorn migration corridors in western Wyoming. (Sawyer et al. 2005). Multiple studies found that mule deer increased their rate of travel during migrations, reducing stop over time and their use of important foraging habitats (Sawyer et al. 2012, Lendrum et al. 2012; Ledrum et al. 2013;). A study in Colorado found that female mule deer changed their migration timing which may change alignment with vegetative phenology and potentially result in energetic and demographic costs (Lendrum et al. 2013).

D. Road density thresholds for fish and wildlife[3]

It is well documented that, beyond specific road density thresholds, certain species will be negatively affected, and some risk being extirpated (Robinson et al. 2000, Table 3). Most studies that look into the relationship between road density and wildlife focus on the impacts to large endangered carnivores or hunted game species, although high road densities certainly affect other species. Grizzly bears have been found to have a higher mortality risk as road density increases (Boulanger and Stenhouse 2014). Gray wolves (*Canis lupus*) in the Great Lakes region and elk in Montana and Idaho also face increased mortality risk, and have undergone the most long-term and in-depth analysis. Forman and Hersperger (1996) found that in order to maintain a naturally functioning landscape with sustained populations of large mammals, road density must be below 0.6 km/km^2 (1.0 mi/mi^2).

A number of studies show that higher road densities also impact aquatic habitats and fish (Table 3). Carnefix and Frissell (2009) provide a concise review of studies that correlate cold water fish abundance and road density, and from the cited evidence concluded that:

> 1) no truly "safe" threshold road density exists, but rather negative impacts begin to accrue and be expressed with incursion of the very first road segment; and 2) highly significant impacts (e.g., threat of extirpation of sensitive species) are already apparent at road densities on the order of 0.6 km/km^2 (1.0 mi/mi^2) or less, (Carnefix and Frissell (2009), p. 1).

Cold water salmonids such as threatened bull trout, are particularly sensitive to the impacts of forest roads. The U.S. Fish and Wildlife Service's Final Rule listing bull trout as threatened (USDI Fish and Wildlife Service 1999) addressed road density stating:

> … assessment of the interior Columbia Basin ecosystem revealed that increasing road densities were associated with declines in four non-anadromous salmonid species (bull trout, Yellowstone cutthroat trout, westslope cutthroat trout, and redband trout) within the Columbia River Basin, likely through a variety of factors associated with roads (Quigley & Arbelbide 1997). Bull trout were less likely to use highly roaded basins for spawning and rearing, and if present, were likely to be at lower population levels (Quigley and Arbelbide 1997). Quigley et al. (1996) demonstrated that when average road densities were between 0.4 to 1.1 km/km^2 (0.7 and 1.7 mi/mi^2) on USFS lands, the proportion of subwatersheds supporting "strong" populations of key salmonids dropped substantially. Higher road densities were associated with further declines (USDI Fish and Wildlife Service (1999), p. 58922).

Anderson et al. (2012) showed that watershed conditions tend to be best in areas protected from road construction and development. Using the U.S. Forest Service's Watershed Condition Framework assessment data, they showed that National Forest lands protected under the Wilderness Act tend to have

[3] We intend for the term "road density" to refer to the density of all roads within national forests, including system roads, closed roads, non-system roads, temporary roads and motorized trails, and roads administered by other jurisdictions (private, county, state).

the healthiest watersheds. In support of this conclusion, McCaffery et al. (2005) found that streams in roadless watersheds had less fine sediment and higher quality habitat than roaded watersheds. Miller et al. (2017) showed that in 20 years of monitoring forests managed by the Northwest Forest Plan there were measurable improvements in watershed conditions as a result of road decommissioning, finding "...the decommissioning of roads in riparian areas has multiple benefits, including improving the riparian scores directly and typically the sedimentation scores."

Table 3: A summary of some road-density thresholds and correlations for terrestrial and aquatic species and ecosystems (reprinted from Robinson et al. 2010).

Species (Location)	**Road density (mean, guideline, threshold, correlation)**	**Reference**
Wolf (Minnesota)	0.36 km/km2 (mean road density in primary range); 0.54 km/km^2 (mean road density in peripheral range)	Mech et al. (1988)
Wolf	>0.6 km/km^2 (absent at this density)	Jalkotzy et al. (1997)
Wolf (Northern Great Lakes re- gion)	>0.45 km/km^2 (few packs exist above this threshold); >1.0 km/km^2 (no pack exist above this threshold)	Mladenoff et al. (1995)
Wolf (Wisconsin)	0.63 km/km^2 (increasing due to greater human tolerance	Wydeven et al. (2001)
Wolf, mountain lion (Minne- sota, Wisconsin, Michigan)	0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained popula- tions)	Thiel (1985); van Dyke et al. (1986); Jensen et al. (1986); Mech et al. (1988); Mech (1989)
Elk (Idaho)	1.9 km/km^2 (density standard for habitat effectiveness)	Woodley 2000 cited in Beazley et al. 2004
Elk (Northern US)	1.24 km/km^2 (habitat effectiveness decline by at least 50%)	Lyon (1983)
Elk, bear, wolverine, lynx, and others	0.63 km/km^2 (reduced habitat security and increased mortality)	Wisdom et al. (2000)
Moose (Ontario)	0.2-0.4 km/km2 (threshold for pronounced response)	Beyer et al. (2013)
Grizzly bear (Montana)	>0.6 km/km^2	Mace et al. (1996); Matt- son et al. (1996)
Black bear (North Carolina)	>1.25 km/km^2 (open roads); >0.5 km/km2 (logging roads); (interference with use of habitat)	Brody and Pelton (1989)
Black bear	0.25 km/km^2 (road density should not exceed)	Jalkotzy et al. (1997)
Bobcat (Wisconsin)	1.5 km/km^2 (density of all road types in home range)	Jalkotzy et al. (1997)
Large mammals	>0.6 km/km^2 (apparent threshold value for a naturally functioning landscape containing sustained popula- tions)	Forman and Hersperger (1996)
Bull trout (Montana)	Inverse relationship of population and road density	Rieman et al. (1997); Baxter et al. (1999)

Fish populations (Medicine Bow National Forest)	(1) Positive correlation of numbers of culverts and stream crossings and amount of fine sediment in stream channels (2) Negative correlation of fish density and numbers of culverts	Eaglin and Hubert (1993) cited in Gucinski et al. (2001)
Macroinvertebrates	Species richness negatively correlated with an index of road density	McGurk and Fong (1995)
Non-anadromous salmonids (Upper Columbia River basin)	(1) Negative correlation likelihood of spawning and rearing and road density (2) Negative correlation of fish density and road density	Lee et al. (1997)

E. Roads and Fires

Wildland forest fire plays an essential role in many forest ecosystems, and with climate change, fire will increasingly shape National Forest lands. Humans have made fire more common on the landscape, and studies have found that forest roads can affect fire regimes and localized fuel regimes. Changes in the timing and location of fire can alter the natural fire regime and has negative, cascading effects in ecological communities. For example, a change in timing and frequency of fire can result in habitat loss and fragmentation, shift forest composition, and affect predator-prey interactions (DellaSalla et al. 2004). Following a fire, exposed bare ground on roads can result in chronic erosion, catastrophic culvert failures, and noxious weed invasion.

Forest roads can increase the occurrence of human-caused fires, whether by accident or arson, and road access has been correlated with the number of fire ignitions (Syphard et al. 2007, Yang et al., 2007, Narayanaraj and Wimberly 2012, Nagy et al. 2018). A recent study found that humans ignited four times as many fires as lightning. This represented 92% of the fires in the eastern United States and 65% of the fire ignitions in the western U.S. (Nagy et al. 2018). Another study that reviewed 1.5 million fire records over 20 years found human-caused fires were responsible for 84% of wildfires and 44% of the total area burned (Balch et al. 2017).

In addition to changes in frequency, human-caused fires change the timing of fire occurring when fuel moisture is significantly higher than lightning-started fires (Nagy et al. 2018.). Forest roads may also limit fire growth acting as a fire break and providing access for suppression (Narayanaraj and Wimberly 2011, Robbinne et al. 2016). The result is a spatial and temporal distribution of fire that differs from historical fire regimes.

Roaded areas create a distinct fire fuels profile which may influence ignition risk and burn severity (Narayanaraj and Wimberly 2013). Forest roads create linear gaps with reduced canopy cover, and increased solar radiation, temperature, and wind speed. Invasive weeds and grasses common along roadsides also create fine fuels that are highly combustible. These edge effects can change microclimates far into the forest (Narayanaraj and Wimberly 2012, Ricotta et al. 2018). While there is little definitive research on roads and burn severity, an increase in the prevalence of lightning-caused fires in roaded areas may be due to roadside edge effects (Arienti et al 2009, Narayanaraj and Wimberly 2012). Furthermore, watersheds that have been heavily roaded have typically received intensive management in the past leaving forests in a condition of high fire vulnerability (Hessburg and Agee 2003).

Roadless areas are remote and secure from many human impacts such as unintentional fire starts or arson. A forest fire is almost twice as likely to occur in a roaded area than a roadless area (USDA Forest Service 2000). In fact, human-ignited wildfire is almost five times more likely to occur in a roaded area than in a roadless area. (USDA Forest Service 2000). Higher road density correlates with an increased probability of human-caused ignitions. (Syphard et al. 2007).

After a forest fire, roads that were previously well vegetated often burn or are bladed for fire suppression access or firebreaks leaving them highly susceptible to erosion and weed invasion. Roads are a source of chronic erosion following a fire, and pulses of hillslope sediment and large woody debris can result in culvert failures (Bisson et al. 2003). Fine sediment is frequently delivered to streams and reduces the quality of aquatic habitat. Noxious weeds are established on many forest roads, and post-fire weed invasion can be facilitated by creating a disturbance, reducing competition, and increasing resource availability (Birdsaw et al. 2012).

II. Climate Change and Transportation Infrastructure

Before the Trump administration took office, the Forest Service recognized the importance of considering and adapting to changing climate conditions. The USDA Strategic Plan for Fiscal Years 2014-2018 set a goal to: "Ensure our national forests and private working lands are conserved, restored, and made more resilient to climate change, while enhancing our water resources." (USDA 2014, p 3). As climate change impacts grow more profound, forest managers must consider the impacts *on* the transportation system as well as *from* the transportation system. In terms of the former, changes in precipitation and hydrologic patterns will strain infrastructure, resulting in damage to streams, fish habitat, and water quality as well as threats to public safety and loss of access. As to the latter, the fragmenting effect of roads on habitat will impede the movement of species which is a fundamental element of adaptation. Through planning, forest managers can proactively address threats to infrastructure, and can actually enhance forest resilience by removing unneeded roads to create larger patches of connected habitat.

A. Climate change, forest roads, and fragmented habitat

It is expected that climate change will be responsible for more extreme weather events, leading to increasing flood severity, more frequent landslides, changing hydrographs, and changes in erosion and sedimentation rates and delivery processes (Schwartz et al. 2014, USDA FS 2018). The Forest

Service Office of Sustainability and Climate has compiled climate change vulnerability assessments for several regions of the Forest Service discussing near-term consequences for managers to consider. (Halofsky et al. 2017, 2018a, 2018b, 2019, with additional vulnerabilities displayed below in Table 4).

> Warmer locations will experience more runoff in winter months and early spring, whereas colder locations will experience more runoff in late spring and early summer. In both cases, future peakflows will be higher and more frequent, (Halofsky et al. 2018b at ii).
>
> The frequency and extent of midwinter flooding are expected to increase. Flood magnitudes are also expected to increase because rain-on-snow-driven peak flows will become more common," (*Id.* at 83).
>
> Roads and other infrastructure that are near or beyond their design life are at considerable risk to damage from flooding and geomorphic disturbance (e.g., debris slides). If road damage increases as expected, it will have a profound impact on access to Federal lands and on repair costs, (*Id.* at viii).

Magnifying these consequences is the fact that roads, culverts and trails in national forests were designed for storms and water flows typical of past decades, and may not be designed for the storms in future decades. Hence, climate driven changes may cause transportation infrastructure to malfunction or fail (USDA Forest Service 2010, ASHTO 2012). The likelihood is higher for facilities in high-risk settings—such as rain-on-snow zones, coastal areas, and landscapes with unstable geology. The following consequences may occur (USDA Forest Service 2010):

- access to national forests will be interrupted temporarily or permanently as roads wash-out due to landslides or blown-out culverts during events of heavier precipitation or flooding;
- public safety will be compromised as roads, trails and bridges become unstable due to landslides, undercut slopes, or erosion of water-logged slopes due to heavy rainfall; and
- infrastructure may be compromised or abandoned along coastal areas or low-lying estuaries when inundated during high tides and coastal storms as sea-levels rise.

Forests fragmented by roads will likely demonstrate less resistance and resilience to stressors, like those associated with climate change (Noss 2001, see also Table 4. below). First, the more a forest is fragmented (and therefore the higher the edge/interior ratio), the more the forest loses its inertia characteristic, and becomes less resilient and resistant to climate change. Second, the more a forest is fragmented, characterized by isolated patches, the more likely the fragmentation will interfere with the ability of species to track shifting climatic conditions over time and space.

> Hence, roads may impede the movement of many species in response to climate change. Closing unnecessary roads and providing wildlife crossings on roads with heavy traffic might mitigate some of these effects (Noss 1993; Clevenger & Waltho 2000), (Noss (2001) p. 584).

Watershed types within national forests may change which will impact hydrology and when high streamflows occur (Halofsky et. al. 2011). A study in Washington's Mt. Baker-Snoqualmie National

Forest (MBSNF) shows that currently 27% of the roads are in watersheds classified as rain-dominated but that will increase to 75% by 2080 - increasing risk of damage to infrastructure (Strauch 2014). By 2040, 300 miles of forest roads in this forest will be located in watersheds that are projected to see a 50% increase in 100-year floods. Landslide risk will be higher during the winter and spring and decline during summer and autumn. These changes reinforce the importance of transportation analysis that incorporates the impacts of climate change.

Earlier snowmelt may open previously snow-closed roaded areas for a greater portion of the year. While this may appear to benefit visitors that wish to access trails and camps early in the spring, this may also put them in harm's way with melting snow-bridges, avalanche chutes and flooding events (Strauch 2015). Wildlife historically protected by snow-closed roads would be more vulnerable.

B. Modifying infrastructure to increase resilience

To prevent or reduce road-triggered landslides and culvert failures, and other associated hazards, forest managers will need to take a series of actions. In December 2012, the USDA Forest Service published a report entitled, *Assessing the Vulnerability of Watersheds to Climate Change* (Furniss et al., 2013) which reinforces that forest managers need to be proactive in reducing erosion potential from roads:

> Road improvements were identified as a key action to improve condition and resilience of watersheds on all the pilot forests. In addition to treatments that reduce erosion, road improvements can reduce the delivery of runoff from road segments to channels, prevent diversion of flow during large events, and restore aquatic habitat connectivity by providing for passage of aquatic organisms. As stated previously, watershed sensitivity is determined by both inherent and management-related factors. Managers have no control over the inherent factors, so to improve resilience, efforts must be directed at anthropogenic influences such as instream flows, roads, rangeland, and vegetation management…. [Watershed Vulnerability Analysis (WVA)] results can also help guide implementation of travel management planning by informing priority setting for decommissioning roads and road reconstruction/maintenance. As with the Ouachita NF example, disconnecting roads from the stream network is a key objective of such work. Similarly, WVA analysis could also help prioritize aquatic organism passage projects at road-stream crossings to allow migration by aquatic residents to suitable habitat as streamflow and temperatures change, (Furniss et al., 2013, p. 22-23).

Other Forest Service reports support road-related actions to increase climate resilience including replacing undersized culverts with larger ones, prioritizing maintenance and upgrades, and restoring roads to a natural state when they are no longer needed and pose erosion hazards (USDA Forest Service 2010, USDA Forest Service 2011a, Furniss et al., 2013, USDA FS 2018, Halofsky et al. 2018a).

The Forest Service has developed several resources to identify and mitigate climate change impacts on forests and infrastructure. The aforementioned climate change vulnerability assessments for each region focus on causes, consequences, and options to address them. For example, Halofsky et al. (2018a) reviews the effects and adaptation options for Region 1 (Northern Region) of the Forest Service, and identifies the increased magnitude of peak streamflows as a primary impact to road infrastructure. Adaptation strategies identified in the report include:

> ...increasing the resilience of stream crossings, culverts, and bridges to higher peakflows and facilitating response to higher peakflows by reducing the road system and disconnecting roads from streams. Tactics include completing geospatial databases of infrastructure (and drainage) components, installing higher capacity culverts, and decommissioning roads or converting them to alternative uses. (Halofsky et al. 2018a)

U.S. Forest Service Transportation Resiliency Guidebook provides a review of the impacts of climate change on Forest Service infrastructure, and a process to assess and address climate change impacts at local and regional levels (USDA FS 2018; Table 4). Included in the guidebook is a step-by-step guide for identifying vulnerabilities and preparedness planning within their transportation network (USDA FS 2018). In addition, the guidebook recommends using the forest plan revision process as "an opportunity to analyze baseline conditions and climate change vulnerabilities and to develop climate resilient strategies for the future." (USDA FS 2018). The Forest Service should use the transportation resilience guidebook to inform forest plan revision analysis and plan components to address climate change in the context of the forest's transportation system.

Table 4. Role of adaptation strategies in reducing climate change impacts of Forest Service lands (reprinted from USDA FS 2018).

	Impacts on Transportation	**Example Strategies to Reduce Impacts**
Heavy Precipitation / Flooding	Flooded roadways interrupting service Damage/destruction of roads and bridges Pavement buckling Erosion comprising soil stability and transportation assets Slope failures Landslides damaging and disrupting routes Plugged or blown out culverts	Retrofit facilities Relocate facilities Upgrade culverts and drainage facilities Build new facilities to climate ready standards Protect existing infrastructure Divest in assets
Wildfires	Additional woody debris that plug culverts Reduced slope stability causing increased landslides Increased heavy vehicle traffic wear and tear on FS roadways	Sustain forest ecology Protect forests from severe fire and wind disturbance
Tree Mortality	Fallen trees disrupt access along transportation routes Increased need for clearing hazard trees along roadways Provide forest fuel for wildfire	Facilitate Forest community adjustments through species transitions

Individual forests have also drafted climate mitigation strategies. The Olympic National Forest in Washington, has developed documents oriented at protecting watershed health and species in the face of climate change, including a 2003 travel management strategy and a report entitled, *Adapting to*

Climate Change in Olympic National Park and National Forest (USDA FS 2011a). The report calls for road decommissioning, relocation of roads away from streams, enlarging culverts as well as replacing culverts with fish-friendly crossings (Table 5). In the travel management strategy, Olympic National Forest recommended that one third of its road system be decommissioned and obliterated. In addition, the plan called for addressing fish migration barriers in a prioritized and strategic way – most of these are associated with roads.

Table 5: Current and expected sensitivities of fish to climate change and associated adaptation strategies and action for fisheries and fish habitat management and relevant to transportation management at Olympic National Forest and Olympic National Park (reprinted from USDA Forest Service 2011a).

Current and expected sensitivities	**Adaptation strategies and actions**
Changes in habitat quantity and quality	Implement habitat restoration projects that focus on re-creating watershed processes and functions and that create diverse, resilient habitat.
Increase in culvert failures, fill-slope failures, stream adjacent road failures, and encroachment from stream-adjacent road segments	Decommission unneeded roads.
	Remove sidecast, improve drainage, and increase culvert sizing on remaining roads.
	Relocate stream-adjacent roads.
Greater difficulty disconnecting roads from stream channels	Design more resilient stream crossing structures.
Major changes in quantity and timing of streamflow in transitional watersheds	Make road and culvert designs more conservative in transitional watersheds to accommodate expected changes.
Decrease in area of headwater streams	Continue to correct culvert fish passage barriers.
	Consider re-prioritizing culvert fish barrier correction projects.
Decrease in habitat quantity and connectivity for species that use headwater streams	Restore habitat in degraded headwater streams that are expected to retain adequate summer streamflow (ONF).

C. Reducing fragmentation to enhance aquatic and terrestrial species adaptation

Reconnecting fragmented forests has been shown to benefit native species (e.g., Damschen et al. 2019). Decommissioning and upgrading roads can reduce fragmentation of both aquatic and terrestrial systems. For example, reducing the amount of road-generated fine sediment deposited on salmonid nests can increase the likelihood of egg survival and spawning success (Switalski et al. 2004, McCaffery et al. 2007). Strategically removing or mitigating barriers such as culverts has been shown to restore aquatic connectivity and expand habitat (Erkinaro et al. 2017). Decommissioning roads in riparian areas may provide further benefits to salmon and other aquatic organisms by permitting reestablishment of streamside vegetation, which provides shade and maintains a cooler, more moderated microclimate over the stream (Battin et al. 2007, Meridith et al. 2014). Coordinating the repair of an aging road system with the mitigation of aquatic organism passage may allow for restoring connectivity while improving infrastructure (Nesson et al. 2018).

One of the most well documented impacts of climate change on wildlife is a shift in the ranges of species (Parmesan 2006). As animals migrate, landscape connectivity will be increasingly important (Holman et al. 2005), and restoring and mitigating migration routes in key wildlife corridors will increase wildlife resiliency. Access management in important elk migration sites would reduce disturbance and improve connectivity (Parton et al. 2017). Similarly, a recent study found grizzly bear population density increased 50 percent following the restriction of motorized recreation (Lamb et al. 2018). Decommissioning roads in key wildlife corridors will also reduce the many road-related stressors. Road decommissioning restores wildlife habitat by providing security and food such as grasses, forbs, and fruiting shrubs (Switalski and Nelson 2011, Tarvainen and Tolvanen 2016).

Forests fragmented by roads and motorized trail networks will likely demonstrate less resistance and resilience to stressors, such as weeds. As a forest is fragmented and there is more edge habitat, Noss (2001) predicts that weedy species with effective dispersal mechanisms will increasingly benefit at the expense of native species. However, decommissioned roads when seeded with native species can reduce the spread of invasive species (Grant et al. 2011), and help restore fragmented forestlands. Off-road vehicles with large knobby tires and large undercarriages are also a key vector for weed spread (e.g., Rooney 2006). Strategically closing and decommissioning motorized routes, especially in roadless areas, will reduce the spread of weeds on forestlands (Gelbard and Harrison 2003).

D. Transportation infrastructure and carbon sequestration

The relationship of road restoration and carbon has only recently been explored. There is the potential for large amounts of carbon (C) to be sequestered by restoring roads to a more natural state. When roads are decompacted during reclamation, vegetation and soils can develop more rapidly and sequester large amounts of carbon. Research on the Clearwater National Forest in Idaho estimated total soil C storage increased 6-fold compared to untreated abandoned roads (Lloyd et al. 2013). Another study concluded that reclaiming 425 km (264 miles) of logging roads over the last 30 years in Redwood National Park in Northern California resulted in net carbon savings of 49,000 Megagrams (54,013 tons) of carbon to date (Madej et al. 2013, Table 5). A further analysis found that recontouring roads had higher soil organic carbon than ripping (decompacting) the roads (Seney and Madej 2015). Finally, a recent study in Colorado found that adding mulch or biochar to decommissioned roads can increase the amount of carbon stored in soil (Ramlow et al. 2018).

Kerekvliet et al. (2008) used Forest Service estimates of the fraction of road miles that are unneeded, and calculated that restoring 126,000 miles of roads (i.e. 30% of the road system) to a natural state would be equivalent to revegetating an area larger than Rhode Island. In addition, they calculate that the net economic benefit of road treatments are always positive and range from US $0.925-1.444 billion.

Table 6. Carbon budget implications in road decommissioning projects (reprinted from Madej et al. 2013).

Road Decommissioning Activities and Processes	Carbon Cost	Carbon Savings
Transportation of staff to restoration sites (fuel emissions)	X	
Use of heavy equipment in excavations (fuel emissions)	X	
Cutting trees along road alignment during hillslope recontouring	X	
Excavation of road fill from stream crossings		X
Removal of road fill from unstable locations		X
Reduces risk of mass movement		X
Post-restoration channel erosion at excavation sites	X	
Natural revegetation following road decompaction		X
Replanting trees		X
Soil development following decompaction		X

E. The importance of Roadless Areas and intact mature forests

Undeveloped natural lands provide numerous ecological benefits. They contribute to biodiversity, enhance ecosystem representation, and facilitate connectivity and provide high quality or undisturbed water, soil and air (Strittholt and Dellasala 2001, DeVelice and Martin 2001, Crist and Wilmer 2002, Loucks et al. 2003, Dellasalla et al. 2011, Anderson et al. 2012, Selva et al. 2015). They can also serve as ecological baselines to help us better understand our impacts to other landscapes, and contribute to landscape resilience in the face of climate change.

Forest Service roadless lands, in particular, are heralded for the conservation values they provide. The benefits are described at length in the preamble of the Roadless Area Conservation Rule (RACR)[4] as well as in the Final Environmental Impact Statement (FEIS) for the RACR[5], and include: high quality or undisturbed soil, water, and air; sources of public drinking water; diversity of plant and animal communities; habitat for threatened, endangered, proposed, candidate, and sensitive species and for those species dependent on large, undisturbed areas of land; primitive, semi-primitive non- motorized, and semi-primitive motorized classes of dispersed recreation; reference landscapes; natural appearing landscapes with high scenic quality; traditional cultural properties and sacred sites; and other locally identified unique characteristics (e.g., include uncommon geological formations, unique wetland complexes, exceptional hunting and fishing opportunities).

The Forest Service, National Park Service, and the U.S. Fish and Wildlife Service recognize that protecting and connecting roadless or lightly roaded areas is an important action agencies can take to

[4] Federal Register, Vol. 66, No. 9. January 12, 2001. Pages 3245-3247.
[5] Final Environmental Impact Statement, Vol. 1, 3–3 to 3–7

enhance climate change adaptation. For example, the *Forest Service National Roadmap for Responding to Climate Change* (USDA Forest Service 2011b) establishes that increasing connectivity and reducing fragmentation are short- and long-term actions the Forest Service should take to facilitate adaptation to climate change. The National Park Service also identifies connectivity as a key factor for climate change adaptation along with establishing "blocks of natural landscapes large enough to be resilient to large-scale disturbances and long-term changes," and other factors. The agency states that: "The success of adaptation strategies will be enhanced by taking a broad approach that identifies connections and barriers across the landscape. Networks of protected areas within a larger mixed landscape can provide the highest level of resilience to climate change."[6] Similarly, the *National Fish, Wildlife and Plants Climate Adaptation Partnership's Adaptation Strategy* (2012) calls for creating an ecologically-connected network of conservation areas.[7]

Crist and Wilmer (2002) looked at the ecological value of roadless lands in the Northern Rockies and found that protection of national forest roadless areas, when added to existing federal conservation lands in the study area, would 1) increase the representation of virtually all land cover types on conservation lands at both the regional and ecosystem scales, some by more than 100%; 2) help protect rare, species-rich, and often-declining vegetation communities; and 3) connect conservation units to create bigger and more cohesive habitat "patches."

Roadless lands also are responsible for higher quality water and watersheds. Anderson et al. (2012) assessed the relationship of watershed condition and land management status and found a strong spatial association between watershed health and protective designations. Dellasalla et al. (2011) found that undeveloped and roadless watersheds are important for supplying downstream users with high-quality drinking water, and developing these watersheds comes at significant costs associated with declining water quality and availability. The authors recommend a light-touch ecological footprint to sustain the many values that derive from roadless areas including healthy watersheds.

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[6] National Park Service. Climate Change Response Program Brief. http://www.nature.nps.gov/climatechange/adaptationplanning.cfm. Also see: National Park Service, 2010. Climate Change Response Strategy. http://www.nature.nps.gov/climatechange/docs/NPS_CCRS.pdf. Objective 6.3 is to "Collaborate to develop cross-jurisdictional conservation plans to protect and restore connectivity and other landscape-scale components of resilience."

[7] See http://www.wildlifeadaptationstrategy.gov/pdf/NFWPCAS-Chapter-3.pdf. Pages 55- 59. The first goal and related strategies are:

> Goal 1: Conserve habitat to support healthy fish, wildlife, and plant populations and ecosystem functions in a changing climate.
>
> Strategy 1.1: identify areas for an ecologically-connected network of terrestrial, freshwater, coastal, and marine conservation areas that are likely to be resilient to climate change and to support a broad range of fish, wildlife, and plants under changed conditions.
>
> Strategy 1.2: Secure appropriate conservation status on areas identified in Strategy 1.1 to complete an ecologically-connected network of public and private conservation areas that will be resilient to climate change and support a broad range of species under changed conditions.
>
> Strategy 1.4: Conserve, restore, and as appropriate and practicable, establish new ecological connections among conservation areas to facilitate fish, wildlife, and plant migration, range shifts, and other transitions caused by climate change.

Allowing roadless and other intact forested areas to reach their full ecological potential is an effective and crucial strategy for atmospheric carbon dioxide removal. Moomaw et al (2019) termed this approach as "proforestation" and explained,

> [f]ar from plateauing in terms of carbon sequestration (or added wood) at a relatively young age as was long believed, older forests (e.g., >200 years of age without intervention) contain a variety of habitats, typically continue to sequester additional carbon for many decades or even centuries, and sequester significantly more carbon than younger and managed stands, (Luyssaert et al., 2008; Askins, 2014; McGarvey et al., 2015; Keeton, 2018).

The authors recommend "scaling up" proforestation, which includes both protecting and expanding designations of intact forested areas, as a cost-effective means to increase atmospheric carbon sequestration.

III. Achieving a Sustainable Minimum Road System on National Forest Lands

A. Background

For two decades, the Travel Management Rule, 36 C.F.R. Part 212, has guided Forest Service road management and use by motorized vehicles. It is divided into three parts: Subpart A, the administration of the forest transportation system; Subpart B, designation of roads, trails, and areas for motor vehicle use; and Subpart C, use by over-snow vehicles. *See* 36 C.F.R. Part 212.

Table 7. Travel Management Rule Subparts – Objectives, Requirements & Products

36 C.F.R. §212	Objective:	Requires:	Product(s):
Subpart A; Roads Rule 2001	To achieve a sustainable national forest road system.	Use a science-based analysis to identify the minimum road system and roads for decommissioning	- Travel Analysis Report - Map with roads identified as "likely needed" and "likely unneeded"
Subpart B; Travel Management Rule 2005	To protect forests from unmanaged off-road vehicle use by ending cross-country travel and ensuring the agency minimizes the harmful effects from motorized recreation.	Designating a system of roads, trails and areas available for off-road vehicle use according to general and specific criteria.	- Motor Vehicle Use Maps that indicate what roads/trails are open for motorized travel
Subpart C; Travel Management Rule	To protect forests from unmanaged over-snow vehicle use in a manner that minimizes their harmful effects.	Designating specific roads, trails and/or areas for oversnow vehicle use according to the criteria per	- Oversnow vehicle maps designating trails and areas for winter motorized recreation

		Subpart B.	

This broad-based national rule is needed because at over 370,000 miles, the Forest Service road system is long enough to circle the earth over 14 times and it is over twice the size of the National Highway System.[8] It is also indisputably unsustainable from ecological, economic and management perspectives. The majority of the roads were constructed decades ago when design and management techniques did not meet current standards (Gucinski et al. 2000, Endicott 2008), making them more vulnerable to erosion and decay. Further, current design standards and best management practices have not been updated to address climate change realities. Exacerbating the problem are massive Forest Service road maintenance backlogs that forces the agency to forego actions necessary to ensure proper watershed function, such as preventing sediment pollution and sustaining aquatic organism passages. Nationally, the total deferred maintenance backlog reached $5.5 billion in FY 2019 of which $3.1 billion is associated with roads.[9] As a result, the road network is not only a massive economic liability, it is also actively harming National Forest System lands, waters, fish and wildlife.

Over the past two decades the Forest Service - largely due to the Travel Management Rule - has made some limited efforts to identify and implement a sustainable transportation system. Yet, overall the agency has yet to meet the requirements of Subpart A. The challenge for forest managers is figuring out what is a sustainable road system and how to achieve it – a challenge exacerbated by climate change. It is reasonable to define a sustainable transportation system as one where all the roads and trails are located, constructed, and maintained in a manner that minimizes harmful environmental consequences while providing social benefits and within budget constraints. This could potentially be achieved through the use of effective best management practices. However, the reality is that even the best transportation networks can be problematic simply because they exist and usher in land uses that, without the access, would not occur (Trombulak and Frissell 2000, Carnefix and Frissell 2009, USDA Forest Service 1996), and when they are not maintained to the designed level they result in environmental problems (Endicott 2008; Gucinski et al. 2000). Moreover, what was sustainable yesterday may no longer be sustainable under climate change realities since roads designed to meet older climate criteria may no longer hold up under new scenarios (USDA Forest Service 2010, USDA Forest Service 2011b, AASHTO 2012, Furniss et al., 2013, Schwartz et al. 2014, USDA FS 2018, Halofsky et al. 2018a, 2018b).

Given consistent budget shortfalls and increasing risks from climate change vulnerabilities, it is clear the agency has an urgent need to both identify and implement a minimum road system, one that will ensure the protection of all Forest Service system lands. However, without specific direction from the Forest Service's Washington D.C. office or Congress, it is reasonable to expect the agency will

8 USDOT Federal Highway Administration, Office of Highway Policy Information. https://www.fhwa.dot.gov/policyinformation/pubs/hf/pl11028/chapter1.cfm
9 USDA Forest Service. 2019. FY2020 Budget Justification. p.83.

continue to rely on piecemeal, project-level analyses to identify the minimum road system. Such an approach is inefficient, and insufficient to achieve a sustainable road system forestwide.

Further, where the Forest Service does act to comply with Subpart A, it typically fails to consider shortcoming in its previous travel analysis processes. In fact, an independent review of 38 Travel Analysis Processes and corresponding reports conducted in 2016 by the U.S. Department of Transportation John A. Volpe National Transportation Systems Center found three overarching concerns:

- A lack of clarity regarding the process;
- Failure to follow 36 CFR 212.5(b) direction and Washington Office guidance; and
- Omission of required documents, referenced appendices, or key supporting materials.

Compounding these concerns is the fact that not only do project-level NEPA analyses fail to account for the TAP shortcomings, they also fail to consider real road/motorized densities when identifying the minimum road system. Moreover, these analyses erroneously assume best management practices and project-specific design features will be effective when the Forest Service authorizes actions to achieve a sustainable road system. Finally, if the project-level decision includes actual road decommissioning, the analysis typically fails to consider or specify treatments, resulting in a legacy of ghost-roads persisting on the landscape. The following sections expand on these shortcomings, which the Forest Service must consider in all project-level analyses, and when revising its land and travel management plans.

B. Using Real Road and Motorized Trail Densities to Identify a Minimum Road System

As the Forest Service works to comply with Subpart A, it is crucial that the agency incorporate the true road and motorized trail densities in both its travel analysis process and NEPA-level analyses. Further, the agency must establish standards in land management plan revisions and amendments to ensure each forest achieves an ecologically sustainable minimum road system. Road density analyses should include closed roads, non-system roads, temporary roads, and motorized trails. Typically, the Forest Service calculates road density by looking only at open system road density. From an ecological standpoint, this is a flawed approach since it leaves out the density calculations of a significant percent of roads and motorized trails on the landscape. These additional roads and motorized trails impact fish, wildlife, and water quality, and in some cases, have more of an impact than open system roads. In this section, we provide justification for why a road density analyses should include more than just open road density whenever the Forest Service evaluates the ecological health of an area during NEPA-level analysis or other processes such as for watershed assessments, forest plan revisions or during travel analysis.

Impacts of closed roads

It is crucial to distinguish the density of roads physically present on the landscape, whether closed to vehicle use or not, from "open-road density." An open-road density of 1.5 mi/mi² has been established as a standard in some national forests as protective of some terrestrial wildlife species. However, many areas with an open road density of 1.5 mi/mi² often have more miles of closed roads which are still hydrologically connected and negatively affecting aquatic and wildlife habitat. This higher density occurs because many road "closures" may block vehicle access, but do nothing to mitigate the hydrologic alterations the road causes. The problem is often further compounded by the existence of "ghost" roads that are not captured in agency inventories, but that are nevertheless physically present and causing hydrologic alteration (Pacific Watershed Associates 2005).

Closing a road to public motorized use can mitigate the impacts on water, wildlife, and soils only if proper closure and storage techniques are followed. Flow diversions, sediment runoff, and illegal incursions will continue unabated if the road is not hydrologically stabilized and adequately blocked from motorized traffic. The Forest Service's National Best Management Practices for non-point source pollution recommends the following management techniques for minimizing the aquatic impacts from closed system roads: eliminate flow diversion onto the road surface, reshape the channel and streambanks at the crossing-site to pass expected flows without scouring or ponding, maintain continuation of channel dimensions and longitudinal profile through the crossing site, and remove culverts, fill material, and other structures that present a risk of failure or diversion (USDA Forest Service 2012).

As noted above, many species benefit when roads are closed to motorized use. However, the fact remains that closed system roads are often breached resulting in impacts to fish and wildlife. A significant portion of gates and closure devices are ineffective at preventing motorized use (Griffin 2004, USFWS 2007). For example, in a legal decision from the Utah District Court, *Sierra Club v. USFS*, Case No. 1:09-cv-131 CW (D. Utah March 7, 2012), the court found that, as part of analyzing alternatives in a proposed travel management plan, the Forest Service failed to examine the impact of continued illegal use. In part, the court based its decision on the Forest Service's acknowledgement that illegal motorized use is a significant problem and that the mere presence of roads is likely to result in illegal use.

In addition to the disturbance to wildlife from motorized use, incursions and the accompanying human access can also result in illegal hunting and trapping of animals. The Tongass National Forest refers to this in its EIS to amend the Land and Resources Management Plan. Specifically, the Forest Service notes in the EIS that Alexander Archipelago wolf mortality due to legal and illegal hunting and trapping is related not only to roads open to motorized access, but to all roads, and that *total road densities* of 0.7-1.0 mi/mi² or less may be necessary (USDA Forest Service 2008).

Impacts of unauthorized (non-system) roads

As of 1998, there were approximately 130,000 miles of non-system roads in national forests (USDA Forest Service, 1998). However, the creation of unauthorized roads continues to be a problem as the Forest Service struggles to properly enforce travel management plans protecting areas from motorized travel. No requirements are in place directing the agency to track or inventory unauthorized roads, therefore currently their precise number is unknown. These roads contribute significantly to the environmental impacts of the transportation system on forest resources, just as forest system roads do. Because the purpose of a road density analysis is to measure the impacts of roads at a landscape level, the only way to do this is for the Forest Service to include all roads, including non-system roads, when measuring impacts. An all-inclusive analysis will provide a more accurate representation of the environmental impacts of the road network within the analysis area.

Impacts of temporary roads

Temporary roads are not considered system roads. Most often they are constructed in conjunction with timber sales. Temporary roads have the same types of environmental impacts as system roads, although at times the impacts can be worse if the road persists on the landscape because they are not built to last. It is important to note that although they are termed temporary roads, their impacts are not temporary. According to Forest Service Manual (FSM) 7703.1, the agency is required to "Reestablish vegetative cover on any unnecessary roadway or area disturbed by road construction on National Forest System lands within 10 years after the termination of the activity that required its use and construction."

Regardless of the FSM 10-year direction, temporary roads often remain for much longer because timber sale contracts typically last 3-5 years or more. If the timber purchaser builds a temporary road in the first year of a five-year contract, its intended use may not end until the full project is complete, which can include post-harvest actions such as prescribed burning. Even though the contract often requires the purchaser to close, obliterate and seed the roadbed with native vegetation, this work typically occurs after a few years of treatment activities. The temporary road, therefore, could remain open for 7-8 years or longer before the FSM ten-year clock starts ticking. Therefore, temporary roads can legally remain on the ground for up to 20 years or more, yet they are constructed with fewer environmental safeguards than modern system roads. Exacerbating the problem is the rise of landscape-scale projects that last between 10-20 years. Unless there is explicit direction requiring temporary road removal within a certain time after treatment activities, it is likely these roads could persist for decades.

Impacts of motorized trails

Motorized use on trails has serious harmful effects similar to roads, and it is crucial for the Forest Service to include motorized trails in its density calculations. As we note several times in Section I above, scientific research and agency publications find similar impacts between motorized trails and roads. Off-road vehicle (ORV) use on trails impact multiple resources, resulting in soil compaction and erosion, trampling of vegetation, as well as wildlife habitat loss, disturbance, and direct mortality. Many of these impacts increase on trails not planned or designed for vehicles, as is often the case when the Forest Service designates ORVs on trails built for hiking or equestrian uses. In many instances the agency designates motorized use on unauthorized trails created through illegal use or from a legacy of unmanaged cross-country travel, further exacerbating the related harmful effects. For a full review of the environmental and cultural impacts on forest lands see Switalski and Jones (2012), and for a review of impacts in arid environments see Switalski (2018).

C. Using Best Management Practices to Achieve a Sustainable Road System

Numerous Best Management Practices (BMPs) were developed to help create a more sustainable transportation system and identify restoration opportunities. BMPs provide science-based criteria and direction that land managers follow in making and implementing decisions about human uses and projects that affect natural resources. Several states have developed BMPs for road construction, maintenance, and decommissioning practices (e.g., Logan 2001, Merrill and Cassaday 2003). The report entitled, National Best Management Practices for Water Quality Management on National Forest System Lands, includes specific road BMPs for controlling erosion and sediment delivery into waterbodies and maintaining water quality (USDA FS 2012). These BMPs cover road system planning, design, construction, maintenance, and decommissioning as well as other transportation-related activities.

Forest Service BMPs - Implementation and Effectiveness

While national BMPs have been established, the effectiveness of individual BMPs, and whether they are implemented at all, is in question. Furthermore, design features are increasingly replacing BMPs for project-level mitigation of road-related environmental impacts. These design features are not consistent among projects, but rather adapted from forest plans and state BMPs, rather than national Forest Service guidelines. Design features need to be standardized, and their rate of implementation and effectiveness systematically reviewed.

When considering how effective BMPs are at controlling nonpoint pollution on roads, both the rate of implementation, and their effectiveness should both be considered. The Forest Service tracks the rate of implementation and the relative effectiveness of BMPs from in-house audits. This information is summarized in the *National BMP Monitoring Summary Report* with the most recent data being the fiscal years 2013-2014 (Carlson et al. 2015). The rating categories for implementation are "fully implemented," "mostly implemented," "marginally implemented," "not implemented," and "no BMPs." "No BMPs" represents a failure to consider BMPs in the planning process. More than a hundred evaluations on roads were conducted in FY2014. Of these evaluations, only about one third of the road BMPs were found to be "fully implemented" (Carlson et al. 2015, p. 12).

The monitoring audit also rated the relative effectiveness of the BMP. The rating categories for effectiveness are "effective," "mostly effective," "marginally effective," and "not effective."

"Effective" indicates no adverse impacts to water from project or activities were evident. When treated roads were evaluated for effectiveness, almost half of the road BMPs were scored as either "marginally effective" or "not effective" (Carlson et al. 2015, p. 13). However, BMPs for completed road decommissioning projects showed approximately 60 percent were effective and mostly effective combined, but it was unclear what specific BMPs account for this success (Carlson et al. 2015, p. 35). As explained below, road recontouring that restores natural hillside slopes is a more effective treatment compared to those that leave road features intact.

A recent technical report by the Forest Service entitled, *Effectiveness of Best Management Practices that Have Application to Forest Roads: A Literature Synthesis* summarized research and monitoring on the effectiveness of different BMP treatments for road construction, presence and use (Edwards et al. 2016). They found that while several studies have found some road BMPs are effective at reducing delivery of sediment to streams, the degree of each treatment has not been rigorously evaluated (Edwards et al. 2016). Few road BMPS have been evaluated under a variety of conditions, and much more research is needed to determine the site-specific suitability of different BMPs (Edwards et al. 2016, also see Anderson et al. 2011).

Edwards et al. (2016) cites several reasons for why BMPs may not be as effective as commonly thought. Most watershed-scale studies are short-term and do not account for variation over time, sediment measurements taken at the mouth of a watershed do not account for in-channel sediment storage and lag times, and it is impossible to measure the impact of individual BMPs when taken at the watershed scale. When individual BMPs are examined there is rarely broad-scale testing in different geologic, topographic, physiological, and climatic conditions. Further, Edwards et al. (2016) observes, "The similarity of forest road BMPs used in many different states' forestry BMP manuals and handbooks suggests a degree of confidence validation that may not be justified," because they rely on just a single study. Therefore, BMP effectiveness would require matching the site conditions found in that single study, a factor land managers rarely consider.

Climate change will further put into question the effectiveness of many road BMPs (Edwards et al. 2016). While the impacts of climate will vary from region to region (Furniss et al. 2010), more extreme weather is expected across the country which will increase the frequency of flooding, soil erosion, stream channel erosion, and variability of streamflow (Furniss et al. 2010). BMPs designed to limit erosion and stream sediment for current weather conditions may not be effective in the future. Edwards et al. (2016) states, "More-intense events, more frequent events, and longer duration events that accompany climate change may demonstrate that BMPs perform even more poorly in these situations. Research is urgently needed to identify BMP weaknesses under extreme events so that refinements, modifications, and development of BMPs do not lag behind the need."

The uncertainties about BMP effectiveness as a result of climate change, compounded by the inconsistencies revealed by BMP evaluations, suggest that the Forest Service cannot simply rely on them, or design features/criteria, as a means to mitigate project-level activities. This is especially relevant where the Forest Service relies on the use of BMPs instead of fully analyzing potentially harmful environmental consequences from road design, construction, maintenance or use, in studies and/or programmatic and site-specific NEPA analyses.

D. Effectiveness of Road Decommissioning Treatments

In order to truly achieve a sustainable minimum road system, the Forest Service must effectively remove unneeded roads. According to the Forest Service, the objective of road decommissioning is to "stabilize, restore, and revegetate unneeded roads to a more natural state to protect and enhance NFS lands" (FSM 7734.0). However, rather than actively removing roads, the Forest Service is increasingly relying on abandoning roads to reach decommissioning treatment objectives (Apodaca et al.2018). Simply closing or abandoning roads will lead to continued resource damage. Other treatments such as ripping the roadbed or installing drainage such as waterbars or dips, have limited and often short-term benefits to natural resources (e.g., Luce 1997, Switalski et al. 2004, Nelson et al. 2010). Recontouring roads is the only proven method to attain the intended outcome of road decommissioning.

Several studies have documented the benefits of fully recontouring roads for ecological restoration. Lloyd et al. (2013) found that rooting depths were much deeper in recontoured roads than in abandoned roads in Idaho, and soil organic matter was an order of magnitude higher on recontoured roads than abandoned roads. Further studies show that soil carbon storage is much higher on recontoured roads as well. A study in Northern California found that recontouring roads resulted in higher soil organic carbon than ripping the roads (Seney and Madej 2015). Higher tree growth and wildlife use has also been found on and near recontoured roads than ripped or abandoned roads (Kolka and Smidt 2004, Switalski and Nelson 2011). Switalski and Nelson (2011) found increased use by black bears on recontoured roads than closed or abandoned roads due to increased food availability and increased habitat security. In addition, removing culverts at stream crossings results in restoring aquatic connectivity and expanding habitat (Erkinaro et al. 2017).

Legacy Roads Monitoring Project

Since 2008, the Forest Service Rocky Mountain Research Station has conducted systematic monitoring on the effectiveness of decommissioned roads in reducing hydrologic and geomorphic impacts from the Forest Service road network. One intent of the monitoring project was to gauge the success of the Legacy Roads and Trails Program that Congress established to provide dedicated funding for the treatment and removal of unnecessary forest roads. The monitoring found that recontouring roads and restoring stream crossings results in dramatic declines in road-generated sediment. Storm-proofing treatments lead to fewer benefits, and on control sites (untreated or abandoned roads), high levels of sediment delivery continued, and the risk of culvert failures remained. For example, a study on the Lolo Creek Watershed on the Clearwater National Forest found a 97% reduction in road/stream connectivity following road recontour (Cissel et al. 2011). Using field observations and the Geomorphic Roads Analysis and Inventory Package (GRAIP), they found a reduction of fine sediments from 38.1 tonnes/year to 1.3 tonnes/year along 3.5 miles of road. Furthermore, they found that restoring road/stream crossings eliminated the risk of culverts plugging, stream diversions, and fill lost at culverts (Table 8).

On the other hand, monitoring conducted on the Caribou-Targhee National Forest found only a 59% reduction of fine sediment delivery from a combination of storm proofing (installation of drain dips), ripping, tilling, and outsloping techniques. There was a reduction of 34.9 tons/year to 14.1 ton/year – leaving a significant amount of sediment continuing to be delivered to streams. Additionally, some stream crossing culverts were not treated and the risk of plugging remained leaving 330 m^3 of fill material at risk. While trail conversion and decommissioning treatments reduced slope failure risks, in some cases storage treatments actually increased the risk of failure (Nelson et al. 2010). Additional monitoring studies conducted in Montana, Idaho, Washington, Oregon, and Utah have similar results.[10]

Table 8. Summary of GRAIP road risk predictions for a watershed on the Clearwater National Forest road decommissioning treatment project (reprinted from Cissel et al. 2011).

IMPACT/RISK TYPE	EFFECT OF TREATMENT: INITIAL GRAIP PREDICTION
Road-stream hydrologic connectivity	-97%, -2510 m
Fine Sediment Delivery	-97%, -36.8 tonnes/yr.
Landslide Risk	Reduced to near natural condition
Gully Risk	Reduced from very low to negligible
Stream Crossing Risk -plug potential -fill at risk -diversion potential	 -100% eliminated at 9 sites -100%, 268 m^3 fill removed -100%, eliminated at 3 sites
Drain Point Problems	17 problems removed, 4 new problems

[10] For reports visit https://www.fs.fed.us/GRAIP/LegacyRoadsMonitoringStudies.shtml

The Forest Service recognizes that fundamental to road decommissioning is revegetating the roadbed. FSM 7734 states, "Decommission a road by reestablishing vegetation and, if necessary, initiating restoration of ecological processes interrupted or adversely impacted by the unneeded road." However, roads are inherently difficult to revegetate because of compaction, lack of soil and organic material, low native seedbank, and presence of noxious weeds (Simmers and Galatowitsch 2010, Ramlow et al. 2018). Many recently acquired industrial timberlands (e.g. Legacy Lands) have road systems with limited canopy cover, little woody debris available, and a large weed seedbank. Thus, revegetation is going to be particularly challenging on these lands.

Consistent application of BMPs that direct recontouring roads for decommissioning will be essential to ensure the treatments best achieve improvements in ecological conditions. More than any other treatment, road recontouring ensures complete decompaction of the roadbed, incorporates native soils that were side-cast during construction, and prevents motorized use. This in turn increases plant rooting depths, soil carbon storage, tree growth, and wildlife use. Any earth disturbing activity can create conditions favorable to noxious weeds, so treating weeds before any treatment and ensuring quick revegetation can limit weeds spread. Applying road recontour BMPs that also mitigate risks associated with noxious weed expansion will help prevent their spread

Conclusion

Numerous studies show that roads and motorized trails negatively impact the ecological integrity of terrestrial and aquatic ecosystems and watersheds. There is ample evidence to confirm the harm to wildlife, aquatic species, water quality, and natural processes from forest roads and motorized use. In addition, the evolving science surrounding roads and wildfire demonstrate a direct link between access and human-caused ignitions, and also suggests that land managers must consider how roads affect fire behavior. Minimizing these impacts by reducing road densities could be an effective solution.

An increasing body of literature exists demonstrating that not only is the Forest Service's transportation infrastructure highly vulnerable to climate change, but also that roads exacerbate climate change's harmful effects to other resources. The agency itself has published multiple reports and guidelines for adaptation, yet few forests are fully translating the information into tangible actions. The Forest Service must implement climate change adaptations as soon as possible, including protecting and expanding intact forests as part of a growing effort to promote natural climate change solutions. Opportunities exist to reduce fragmentation, sequester carbon, and expand roadless areas by implementing a minimum road system.

The Forest Service must fulfil its mandate to achieve an ecologically and economically sustainable forest road system by fully complying with the Roads Rule's requirement to identify a minimum road system. Inconsistent policy interpretations, inadequate travel analysis reports and lack of accountability has largely left this goal wholly out of reach. Yet this work remains vitally important, especially in the context of climate change. The Forest Service should reinvigorate its efforts to comply with the rule's requirements. Towards this end, the agency must include current science, particularly related to future climate conditions. All road and motorized trail densities should be included in the analysis. When the agency actually does identify a minimum road system and proposes to remove unneeded roads, it must carefully evaluate the effectiveness of all proposed BMPs and design features, and fully implement the most effective decommissioning treatments to maximize restoring ecological integrity to the area. These actions will ensure the Forest Service finally achieves its goal to establish a truly sustainable forest road system.

Recontoured road, Olympic National Forest - Skokomish Watershed, 2017. By WildEarth Guardians

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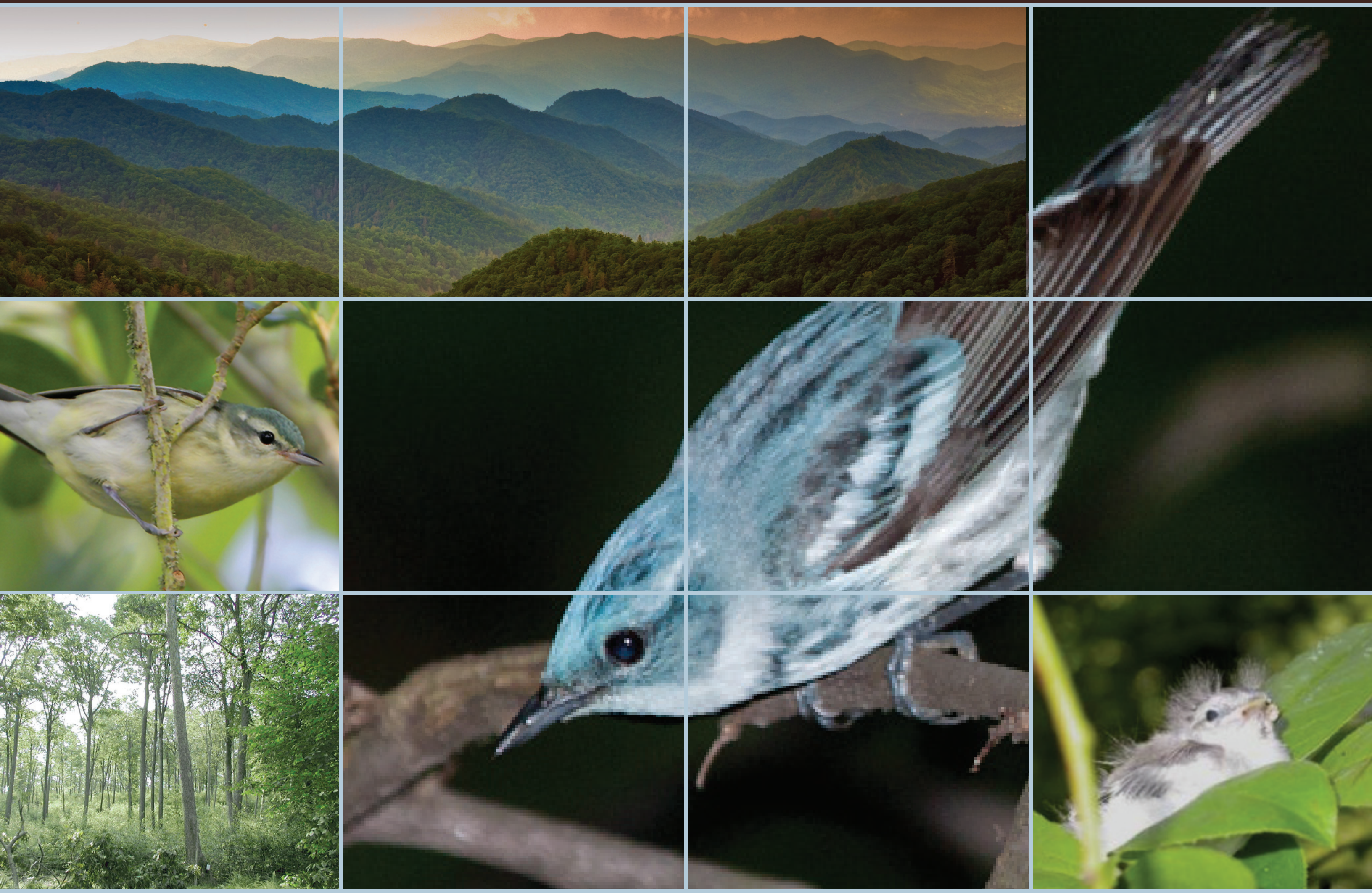
CERULEAN WARBLER

Management Guidelines for Enhancing Breeding Habitat in Appalachian Hardwood Forests



February, 2013

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Photos: Cover: Clockwise: Appalacian landscape, Charlie Choc; Cerulean Warbler, Bill Hubick, http://www.billhubick.com; Fledgling, Marja Bakermans; Shelterwood, Scott Stoleson; Female, Bill Hubick

Appalachian breeding habitat. Than Boves

Table of Contents

Male Cerulean Warbler. Marja Bakermans

Introduction

The Cerulean Warbler (*Setophaga cerulea*) is a migratory songbird that breeds in mature deciduous forests of eastern North America. Cerulean Warblers (hereafter, ceruleans) require heavily forested landscapes for nesting and, within Appalachian forests, primarily occur on ridge tops and steep, upper slopes. They are generally associated with oak-dominated (*Quercus* spp.) stands that contain gaps in the forest canopy, that have large diameter trees (>16 inches diameter breast height (dbh)), and that have well-developed understory-and upper-canopy layers. Ceruleans primarily use the mid- and upper-canopy where they glean insects from the surface of leaves and conceal their open cup nests. Because they are severely declining across much of their range (Fig. 1), habitat management is a high priority. Management for this species can also improve conditions for a number of other wildlife species that depend on the same structure.

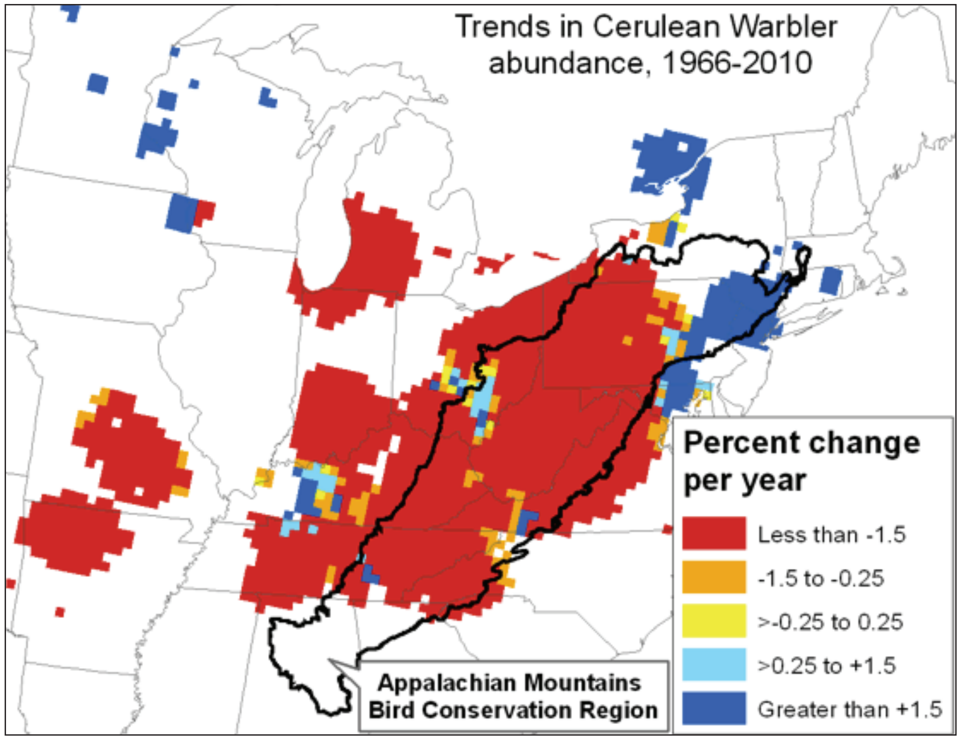


Figure 1. Cerulean Warbler distribution and trends in abundance across their breeding range from Breeding Bird Survey data (1966-2010; Sauer et al. 2011). The Appalachian Mountains Bird Conservation Region boundary is in black.

Adult Cerulean Warbler feeding chick. Wayne Miller

Goals

This document provides land managers in the Appalachian Region with guidelines for retaining and enhancing habitat for Cerulean Warblers and a diverse bird community based on the current available science. They are intended for use by federal, state and private foresters, biologists, and other land managers. These management guidelines are based to a large extent on the recently completed Cooperative Cerulean Warbler Forest Management Project (CWFMP) but also incorporate relevant findings from other research projects. All literature incorporated into this document is listed in the Reference section. The guidelines apply primarily to upland oak-dominated habitats where the majority of the research reported was completed.

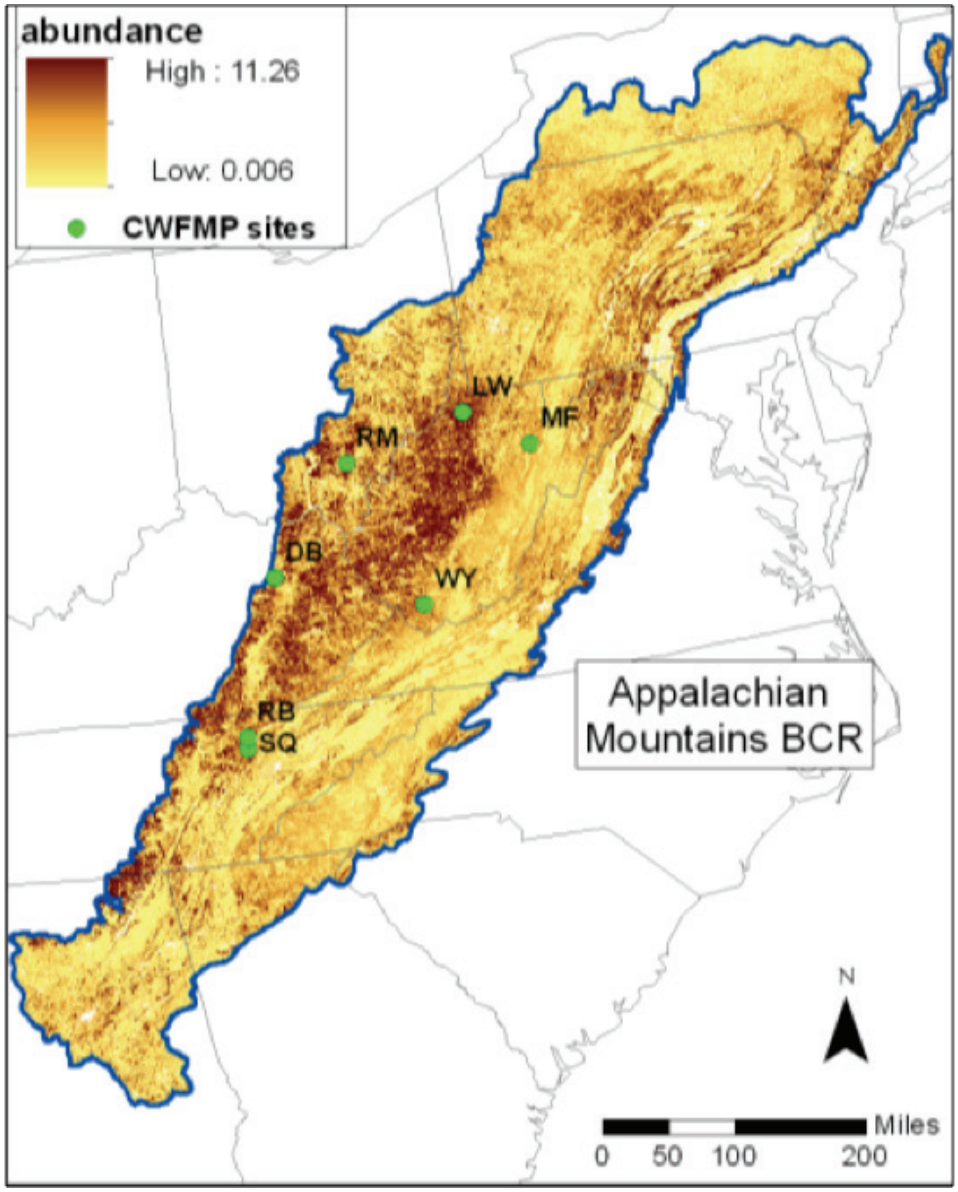


Figure 2. Cerulean Warbler abundance (number per route) estimated from Breeding Bird Survey data for the Appalachian Mountains Bird Conservation Region (BCR) (adapted from Shumar 2009). Study areas from the Cerulean Warbler Forest Management Project (CWFMP) are in the core range of the species.

Conservation

Male Cerulean Warbler. Than Boves

About 80% of the total cerulean population breeds within the Appalachian Mountains Bird Conservation Region (BCR; Fig. 1), and they are particularly abundant within the central part of the region (Fig. 2). Declines have occurred across most of their range (Fig. 1). A range-wide loss of ~70% of the population (Fig. 3) led to their designation as a species of national conservation concern by the U.S. Fish and Wildlife Service (USFWS) and as a Continental Watch List species by Partners in Flight.

Cerulean declines are primarily related to the loss and reduced suitability of habitat on breeding, migration, and wintering grounds. On breeding grounds, the second growth forests that occur throughout most forested landscapes often lack the complex forest structure favored by ceruleans. Old-growth forests naturally develop a more open and complex canopy structure, as well as multi-layered shrub and mid-story layers. Maintaining older, structurally diverse forest within cerulean breeding range may be important to sustain populations in the long-term and to support the ecosystems on which they and other organisms depend. In managed forests, however, foresters and landowners can use silviculture as a tool to develop stands with structural and compositional characteristics that are favorable for cerulean and associated species. Partial harvesting to benefit ceruleans can be consistent with forest management goals such as promoting oak regeneration and managing for a diverse wildlife community.

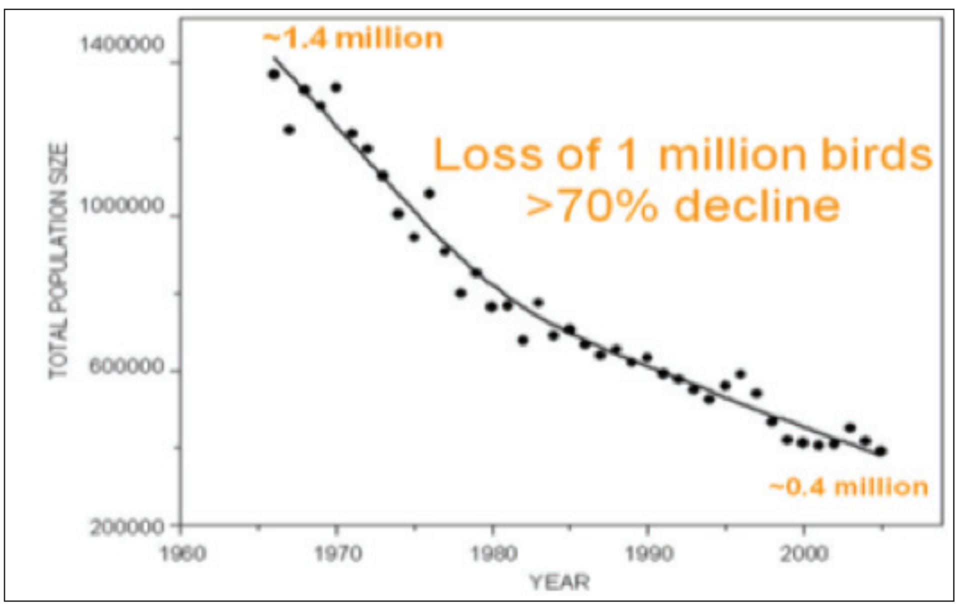


Figure 3. Cerulean Warbler population decline modeled using Breeding Bird Survey data from 1966-2006 (W. Thogmartin, unpubl. analyses).

Cerulean Warbler Habitat Association

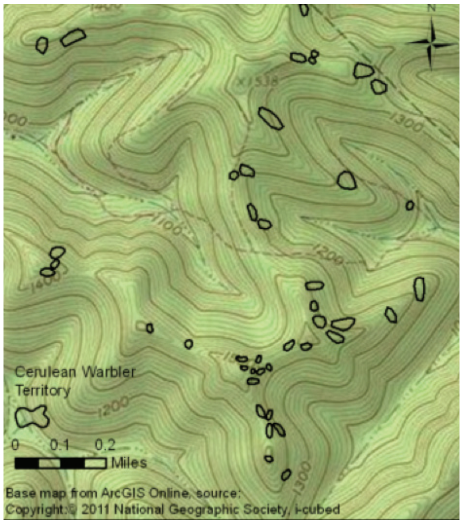


Figure 4. Cerulean Warbler territories on a topographic map of the Lewis Wetzel Wildlife Management Area, West Virginia, showing territories aligned along ridgelines and clustering near areas of local relief.

Cerulean breeding density is variable across the Appalachian region (Fig. 2). Their distribution is often patchy in part due to the patchy nature of canopy disturbance in mature forests and their strong association with ridge tops. In a southern West Virginia study, for example, they occurred at 40% of randomly placed sample points.

Landscape and Topography

Small forest tract size and the presence of large-scale edge (e.g., agricultural lands, mountaintop mines) can limit use of a site by ceruleans. Although the minimum forest tract size required by ceruleans to breed successfully is not known, smaller, more fragmented forest patches tend to have lower densities of territories and lower nest success. Ceruleans will use relatively small forest patches (~25 ac), but typically in landscapes that are primarily forested (e.g. >75% forest cover within ~6 miles of the project area). In landscapes with a relatively low proportion of forest cover (e.g. those that are dominated by agriculture), ceruleans are less likely to occur within small forest tracts. In the heavily deforested Mississippi Alluvial Valley, ceruleans require ~4000 acre tracts, in the highly fragmented Mid-Atlantic region ~1730 acres, and in the more forested Ohio Hills ~60 acres.

Ceruleans are often associated with canopy gaps and also use internal forest edges including narrow roads, narrow utility rights-of-way, narrow-cut strip mines, edges of small timber harvests, and trails. However, they are less abundant near abrupt or "hard" edges between forest cover and large expanses of open land (e.g., commercial, residential, and industrial development). In southern West Virginia, for example, cerulean abundance decreased near mountaintop mine edges and in northern West Virginia, they avoided edges of a large powerline right-of-way that was ~75 feet wide.

In the Appalachians, ceruleans primarily occur along ridges and steep, upper slopes and appear to cluster near areas of local relief such as knobs and bluffs (Fig. 4). The soil characteristics and topography of these features contribute to stratification of canopy trees so that ridge top forests often have a complex overstory structure containing large oaks with expansive crowns. Thus, ridge top forests often offer the structure and composition sought by breeding ceruleans. Within ridge top forests, ceruleans often favor mesic, north- and northeast-facing slopes, although other aspects are used. In some sections of the Appalachians (e.g. Delaware River valley), ceruleans are most dense at lower slope positions and along major waterways.

Appalachian landscape. Than Boves

Minimum patch size used by ceruleans depends on the amount of forest cover in the landscape.

Stand structure and Composition

Before extensive clearcutting in the late 19th and early 20th century, tree mortality from old age, windthrow, ice storm damage, and fire contributed to the development of structurally complex and relatively open stands in which oaks were dominant. In the even-aged stands that developed following those extensive harvests, natural canopy disturbances tended to be unevenly distributed and relatively small thereby creating a relatively homogenous canopy structure (e.g., a closed canopy forest with an undeveloped understory and/or mid-story).

Important Components of Cerulean Habitat

Large Diameter Trees

Ceruleans place territories and nests in hardwood forests with well-spaced, large diameter trees (>16 inches dbh). Nests are typically in the largest trees available at a site.

Canopy Gaps and Structure

Ceruleans favor the complex canopy structure characteristic of uneven-aged stands and old growth forest. Canopy gaps allow mid- and upper-canopy trees the growing space to form long horizontal branches and develop dense foliage. Tree species composition is relatively diverse with shade-intolerant species abundant in the overstory.

Upland forest used by Cerulean Warbler. Marja Bakermans

Heterogenous stand structure including large trees, canopy gaps, and understory vegetation promote density and reproductive success of ceruleans.

A relatively open canopy structure provides ceruleans with dominant trees (i.e., taller than the surrounding canopy) where exposed perches aid the birds in broadcasting their song and whose expansive crowns offer ample foliage in which to forage and conceal nests. Nests are often placed along flat lateral branches that extend over a relatively open midstory and a relatively dense understory, conditions that occur adjacent to a regenerating canopy gap. Ceruleans preferentially use canopy gaps ~400-1000 ft^2 in size and that contain vegetative growth within them.

Oaks and Hickories

In the Appalachians, ceruleans are strongly associated with stands in which oaks and hickories (*Carya* spp.) predominate. They preferentially forage and nest in white (*Q. alba*) and chestnut oak (*Q. montana*), but they avoid red maple (*Acer rubrum*) and oaks from the red oak group (scarlet (*Q. coccinea*), black (*Q. velutina*), and northern (*Q. rubra*) and southern red oak (*Q. falcata*). On sites dominated by species other than oaks, ceruleans preferentially used black cherry (*Prunus serotina*) and black locust (*Robinia pseudoacacia*) in West Virginia and American elm (*Ulmus americana*) and sycamore (*Platanus occidentalis*) in Ohio for various activities.

Grapevines

Grapevines provide a favored source of nest material. Cerulean nest success was positively associated with density of grapevines (*Vitis spp.*) in Ohio perhaps because vines add complexity to the canopy and, consequently, reduce the search-efficiency of nest predators. In Maryland, fledglings often were observed perching within clumps of grapevines.

Understory Vegetation

Density and nest success of ceruleans have been positively associated with understory vegetation. In Ohio, vegetation surrounding nest locations had 24% greater understory vegetation density than random locations in the stand. A high density of understory vegetation is beneficial to ceruleans because 1) females frequently drop to the understory for intensive foraging bouts during incubation and brooding, and 2) fledgling birds often seek the dense vegetation for protection from predators.

Leave some grapevines to provide nest material.

Female Cerulean Warbler incubating; note grapevine bark on the nest rim. This is a typical location for nests, i.e. on a lateral branch, next to a vertical twig, with an umbrella of leaves above the nest. Than Boves

Cerulean Warbler fledgling in thick understory vegetation. Marja Bakermans

Cerulean Warbler nest of grapevine and other materials. Marja Bakermans

Cooperative Cerulean Warbler Forest Management Project

The Cooperative Cerulean Warbler Forest Management Project (CWFMP), implemented under the auspices of the Cerulean Warbler Technical Group, was initiated to allow the scientific and management communities to test ideas about the habitat needs of ceruleans through experimental manipulations of timber harvest. The objective of the CWFMP was to study the response of ceruleans and the overall bird community to three silvicultural treatments and an unharvested control, collectively representing a canopy disturbance gradient. Seven study sites, each containing the four treatments, were established within mixed-mesophytic forest in Tennessee, Ohio, Kentucky, and West Virginia (Fig. 2). Sites were closed-canopy mature forest and located in heavily forested regions; forest cover within six miles of study areas averaged 83%. All stands were oak dominant.

Pre-harvest, West Virginia LW study area, basal area = 121 ft^2/acre Patrick McElhone

Light harvest in 2007 (1 yr post-harvest), West Virginia LW study area, RBA=83.6 ft^2/acre. Patrick McElhone

Treatment plots were 50 acres in size and included an unharvested plot, a light harvest, a medium harvest, and a heavy harvest (Fig. 5). In harvested plots, treatments included a 25-acre harvest and a 25-acre section of undisturbed forest that bordered the harvest (hereafter buffers). Light harvests were single tree removals and residual basal area (RBA) averaged 93 ft^2/acre (range 84-106) resulting in stands that had~80% stocking. The goal of medium harvests was to thin the stand to

Medium harvest in 2010 (4 yrs post harvest), West Virginia LW study area, RBA=45.5 ft²/acre. Jim Sheehan

Heavy harvest in 2008 (2 yrs post-harvest), Tennessee, RB study area. residual basal area (RBA)=34.5 ft²/acre. Than Boves

a residual stocking of 60-70% and favor the crown release of the best quality dominants and codominants. All other commercial stems (>6 inches dbh) were removed. The heavy harvests were applied with the objective of creating an understocked residual stand comprised of scattered dominants and co-dominants with all other commercial stems (>6 inches dbh) removed. After harvesting, the medium harvest had average RBA of 62 ft^2/acre (range 46-81) resulting in ~55% stocking. The heavy harvests had average RBA of 27 ft^2/acre (range 12-34). Basal area for unharvested plots averaged 117 ft^2/acre (range 95-138) with ~100% stocking.

The CWFMP is the largest forest management experiment ever conducted to evaluate cerulean warbler and associated songbird response to forest management. The results of the study demonstrate the initial response of ceruleans (first four years post-harvest) to forest management. Additional studies are needed to track cerulean response over the life of a managed stand to fully characterize the nature of the changes in habitat structure that occur in these stands and how ceruleans respond to these changes.

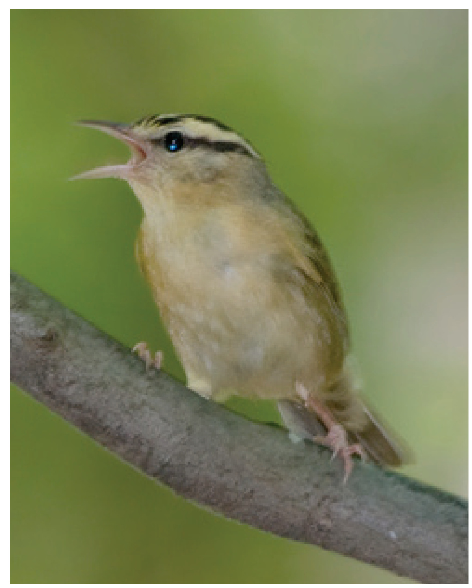
During two pre-harvest field seasons (2005-2006) and four post-harvest field seasons (2007-2010), data were collected on cerulean nest success, territory density, and habitat use. We also measured composition and relative abundance of the overall bird community to characterize response to partial harvesting and mapped territories of six other focal species in addition to Cerulean Warbler: Hooded Warbler (*Setophaga citrina*), Kentucky Warbler (*Geothlypis formosus*), Ovenbird (*Seiurus aurocapillus*), Scarlet Tanager (*Piranga olivacea*), Wood Thrush (*Hylocichla mustelina*), and Worm-eating Warbler (*Helmitheros vermivorus*).

Kentucky Warbler. Bill Hubick

Ovenbird. William Majoros

Scarlet Tanager. Bill Hubick

Wood Thrush. USFWS



Worm-eating Warbler. Bill Hubick

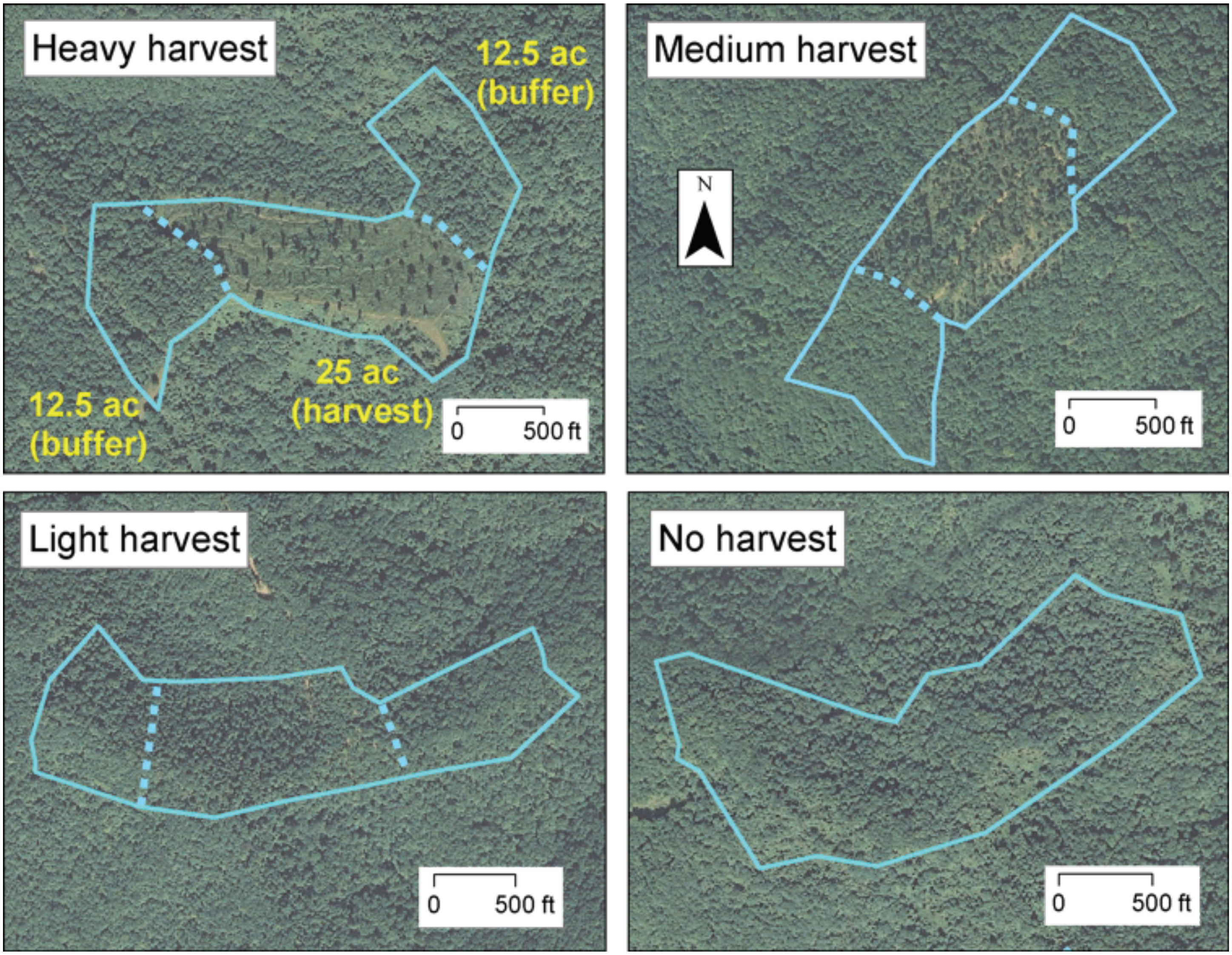


Figure 5. Plot layout in the CWFMP showing harvests and unharvested buffer areas one year after harvests were implemented on LW in WV.

Findings Relevant to Silvicultural Prescriptions

Short-term Response of Cerulean Warblers to Harvests

Territory Density

■ Across all harvests, cerulean territory density generally increased or was maintained and rarely decreased from pre-harvest densities (Fig. 6 top). The modeled response indicated that annual increases occurred (Fig. 7).

■ The largest and most consistent increases occurred when RBA was between ~40 and 90 ft²/ac (Fig 6 top, Fig 7). An extreme increase occurred in a harvest ~45 ft²/ac RBA where ceruleans were absent preharvest; post-harvest territories here were densely clustered.

■ Territory density increases that occurred at low levels of RBA (<40 ft²/ac) were typically delayed 2-3 years, likely in response to the time needed for understory foliage and structural development to occur in the residual stand. Within these heavy harvests, territories were often situated along the harvest edge (Fig. 8) and nests were rarely located within the harvest.

■ Single tree selection harvests with RBA >90 ft²/ac produced little increase in cerulean territory density (Fig 6 top).

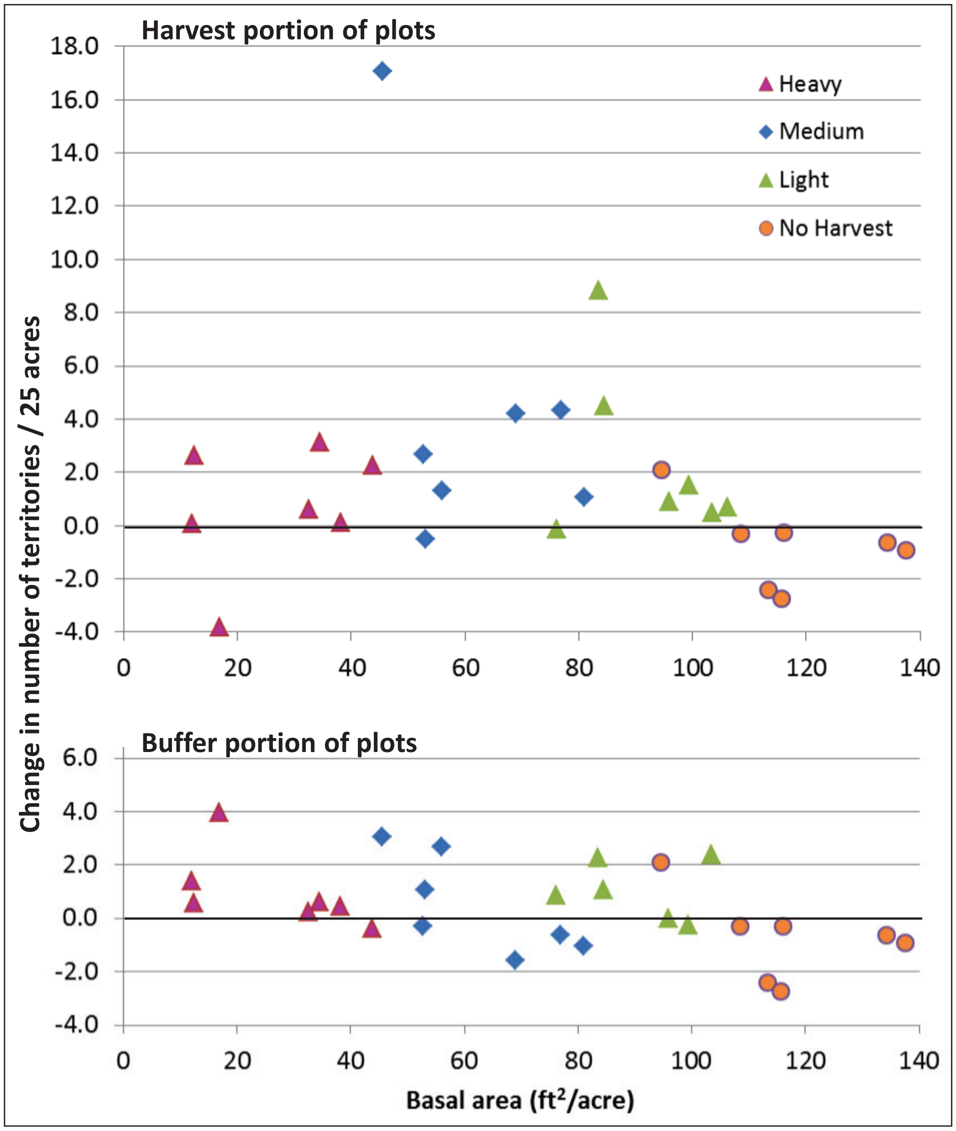


Figure 6. Mean change in number of cerulean warbler territories per 25 ac from 2006 (pre-harvest) to 2007-2010 (post-harvest) relative to post-harvest basal area and harvest intensity. Top figure is within harvests and bottom figure is within unharvested buffers. Points above the 0 line indicate plots with a mean increase in number of territories.

Ceruleans favor residual basal area of ~40 to 90 ft²/acre of canopy trees.

■ Although the territory density response to harvests was generally positive (Fig. 6 top, Fig. 7) it was variable across study sites likely due to differences in pre-harvest cerulean densities, topography, and forest structure and composition.

■ In the majority of unharvested buffers (Fig. 6 bottom), cerulean territory density mostly increased or was maintained regardless of intensity of the adjacent harvest.

■ Some degree of thinning in the canopy of oak-dominated stands with basal area >~130 ft²/ac would likely benefit ceruleans because territory density generally was low on these highly stocked stands (Fig 7).

Cerulean Warbler male with color bands. Matt Shumar

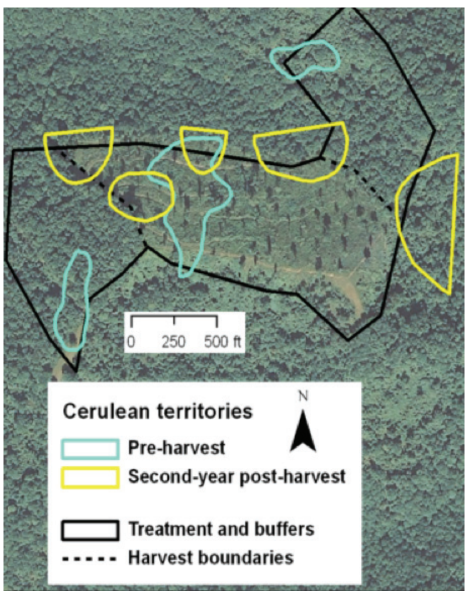


Figure 8. Cerulean Warbler territories aligned along the edge of a 20 acre heavy harvest with 12.5 ft²/ac of residual basal area. Territories before the harvest are shown in blue and after harvest are in yellow. The birds used little of the interior of the cut.

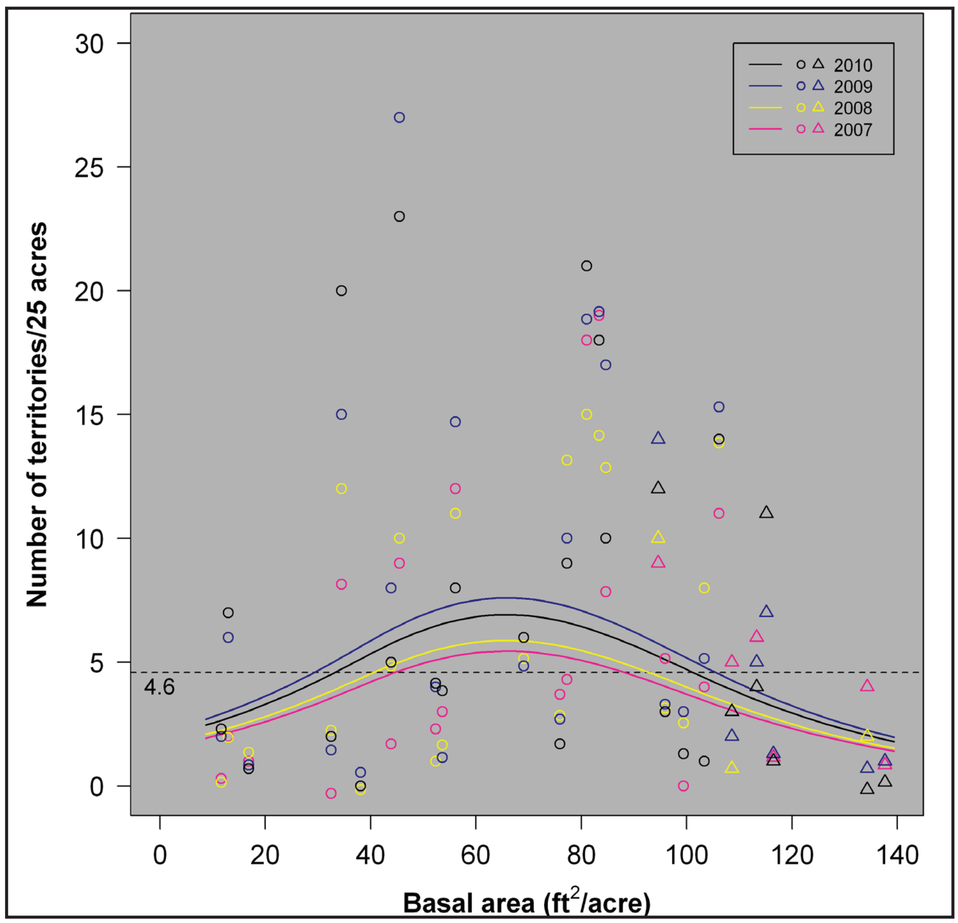


Figure 7. Annual number of post-harvest (2007-2010) cerulean warbler territories per 25 acres (circles=harvests; triangles=no-harvest control) relative to post-harvest basal area. Curved lines are the annual post-harvest predicted response for a plot with 4.6 pre-harvest territories/25 acres (the pre-harvest mean indicated by the thin dotted horizontal line).

Nest Success

■ Nest success varied strongly by study site and year and was relatively low at many of the study areas. Harvest intensity had less influence on nest success than study area and year.

■ Unharvested buffers adjacent to the harvests had nest success similar to that of the unharvested control stands.

■ Of the three harvest treatments, medium harvests had higher nest success than light or heavy harvests (Fig. 9). However, unharvested control stands in the South region (the two Tennessee study areas) had higher nest success than any harvest.

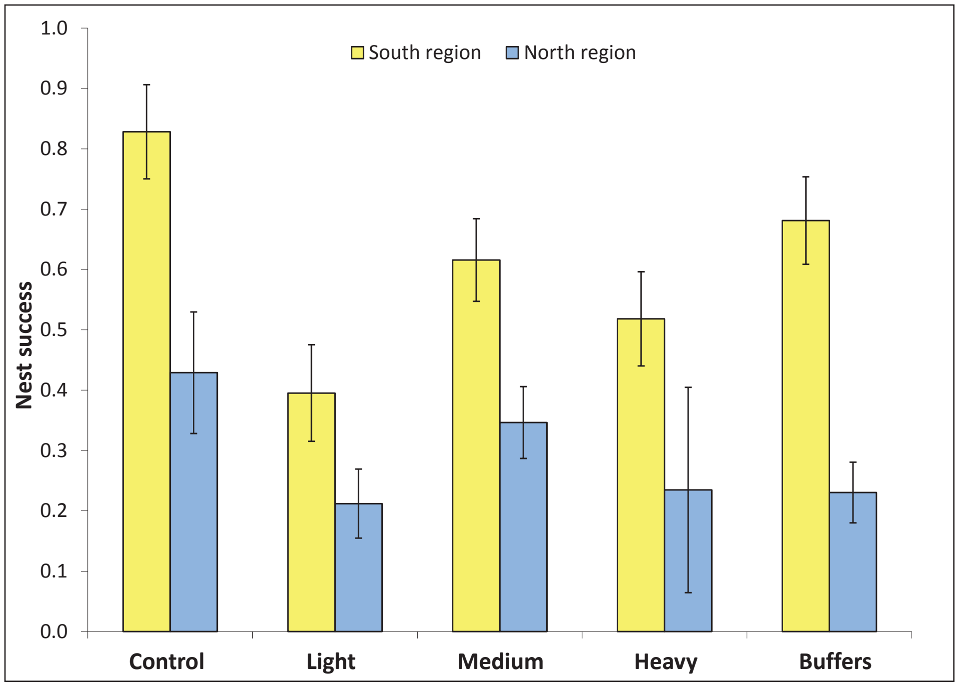


Figure 9. Cerulean Warbler nest success (with standard error bars) for the no harvest control, the three harvest treatments, and the unharvested buffers.

Male Cerulean Warbler with newly hatched chicks. Ohio DNR

Habitat Use

■ For nest trees, ceruleans preferred white oak, sugar maple (*A. saccharum*), and cucumber magnolia (*Magnolia acuminata*) as nest trees and avoided red maple and oaks from the red oak group (scarlet, black, and northern and southern red oak) (Fig. 10).

■ For foraging, they preferred sugar maple, chestnut oak, and hickories and again avoided oaks from the red oak group (Fig. 11).

■ Ceruleans placed their nests in trees that averaged 15-19 inches dbh across the study areas. Nest trees were larger than random trees within the territory. Vegetation structure adjacent to nest trees had less mid-canopy cover and more understory cover than generally available within the surrounding territory. These conditions are characteristic of canopy gaps that have some vegetative growth within them.

White oaks, hickories, and sugar maples are favored for nesting and foraging.

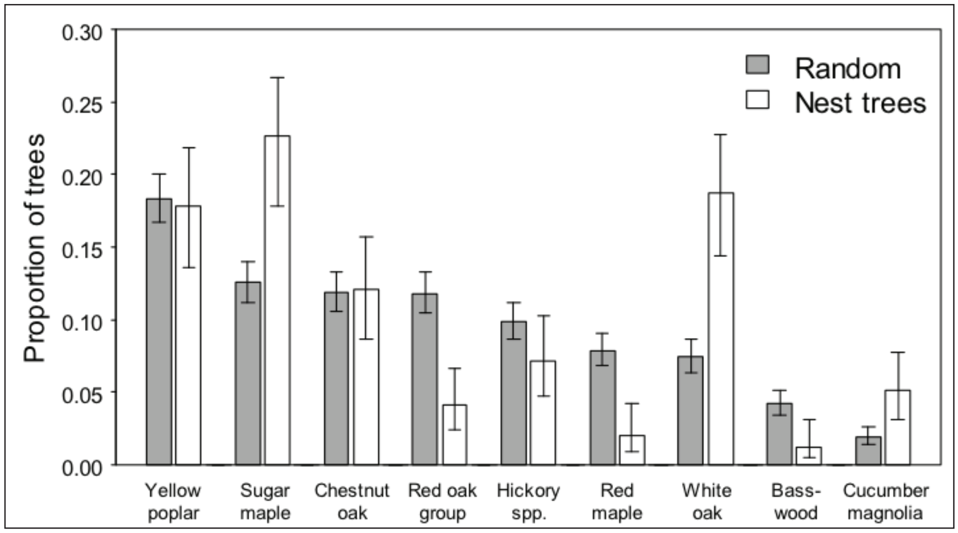


Figure 10. Nest tree selection by Cerulean Warblers at all study areas (pooled) in the Appalachian Mountains, 2008–2010. For each tree species, bars and 95% confidence intervals are the proportion of total trees within randomly sampled plots (gray) and the proportion of total nest trees (white). Red oak group includes northern red (Quercus rubra), black (Q. velutina), and scarlet (Q. coccinea) oak, and hickory species include mockernut (Carya tomentosa), bitternut (C. cordiformis), pignut (C. glabra), and shellbark (C. laciniosa) hickory. Only the most common tree species are shown.

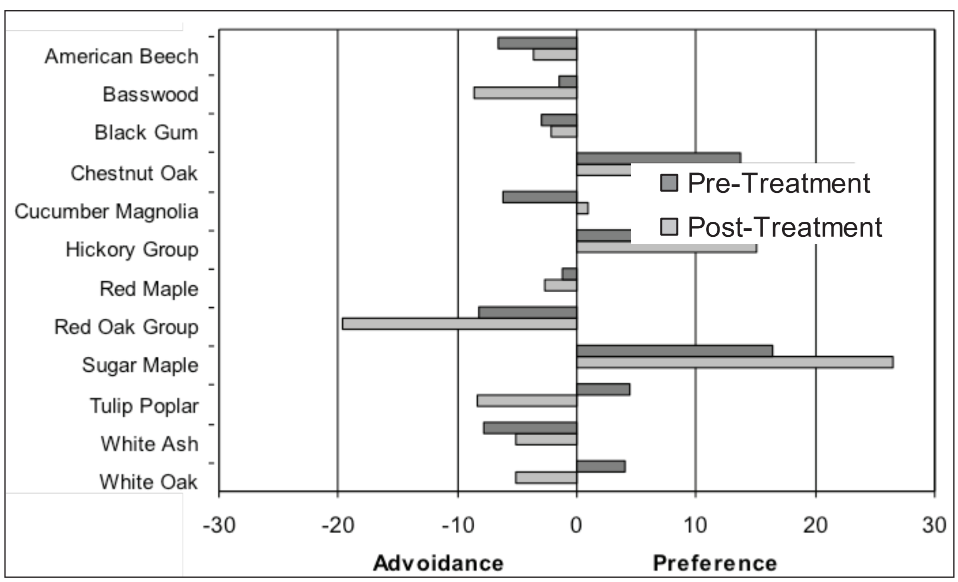


Figure 11. Pre-harvest (2006) and post-harvest (2007) indices of tree species preference and avoidance by Cerulean Warblers for the 12 most commonly available tree species.

Changes in Allied Bird Communities

Appalachian forests are considered some of the most biologically diverse temperate forests in the world. They provide breeding habitat for many avian species including those dependent on closed-canopy forest, others that require young forest habitat, and some species that require mature forest with canopy gaps. Consequently, individual species responded in various ways to different levels of RBA (Table 1).

■ Ovenbird, a species that nests and forages on the ground, had its greatest abundance at high RBA (>90 ft^2/ac; Fig. 12). An immediate negative response to canopy removal persisted four years after harvests in heavy and medium harvests. Ovenbirds occurred at moderate densities in light harvests (>85 ft^2/ac).

■ Species that nest in the midstory of older forests such as Wood Thrush and Acadian Flycatcher (*Empidonax virescens*), also had immediate and persistent reductions in abundance in response to canopy removal in heavy and medium harvests. This was likely in response to midstory removal and the open canopy and dense understory conditions that developed in response to these harvest levels.

■ Heavy and medium harvests increased abundance and diversity of shrub-nesting species including Hooded Warbler (Fig. 12), Indigo Bunting (*Passerina cyanea*), Yellow-breasted Chat (*Icteria virens*), Kentucky Warbler, and Eastern Towhee (*Pipilo erythrophthalmus*). These species are associated with low RBA and high shrub cover. Response of some species, e.g. Hooded Warbler and Kentucky Warbler, was delayed until dense shrub cover developed.

■ Certain canopy-nesting species such as Cerulean Warbler and Blue-gray Gnatcatcher (*Polioptila caerulea*) generally increased in abundance at intermediate levels of RBA across the study sites while Eastern Wood Pewee (*Contopus virens*) increased only in Ohio at intermediate RBA. Some canopy-nesters that are less sensitive to small-scale harvesting, like Scarlet Tanager, had similar abundance across the range of harvest intensities.

These short term effects are from small-scale harvesting (~25 ac) within relatively continuous mature forest. Avian species may respond differently to larger harvests, more extensive harvesting, or harvesting within landscapes with less forest cover.

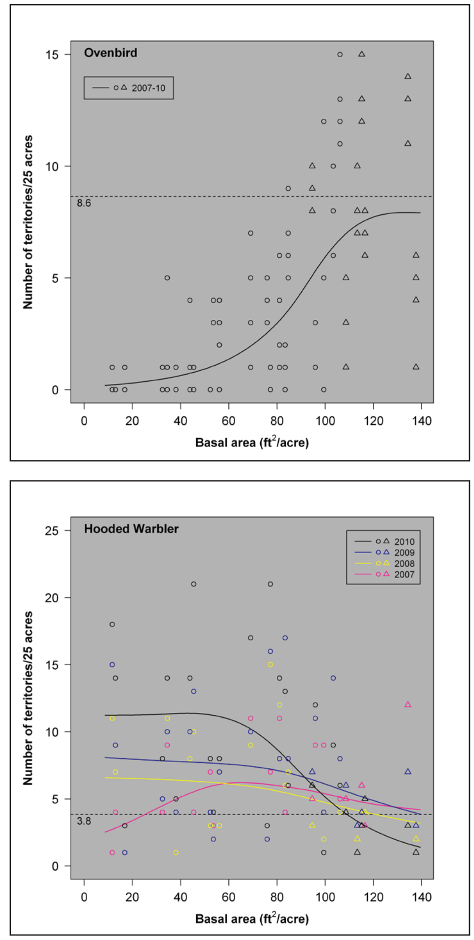


Figure 12. Number of post-harvest (2007-2010) Ovenbird and Hooded Warbler territories per 25 acres (circles=harvests; triangles=no-harvest control) relative to post-harvest basal area. Negative (Ovenbirds) and positive (Hooded Warbler) predicted responses to basal area are shown by curved lines (the pre-harvest mean indicated by the thin horizontal line). For Hooded Warbler, there was an annual increasing response during 1 to 4 years post-harvest.

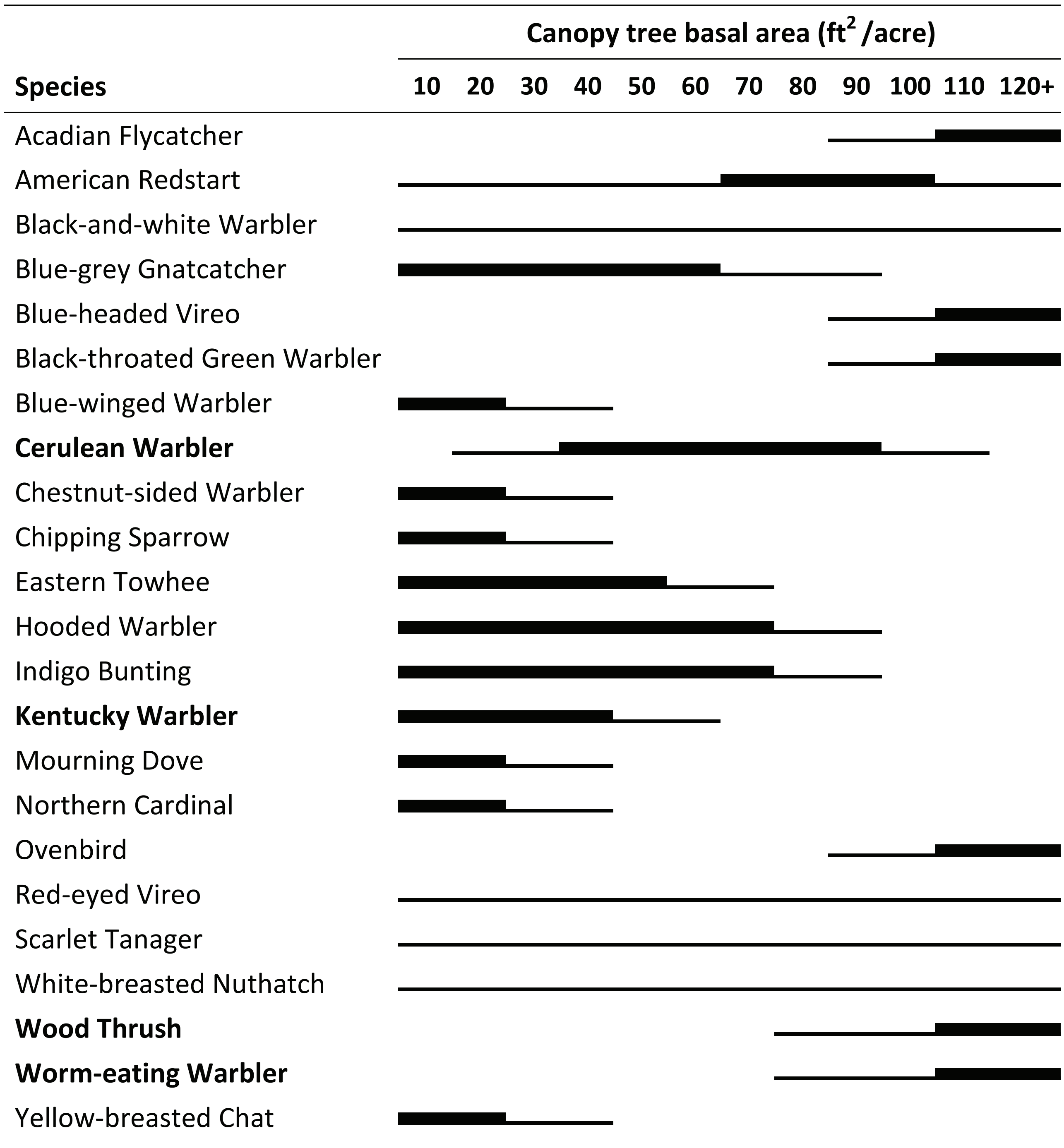


Table 1. Suitable and optimal (thickest line) basal areas for migratory songbirds that were common at CWFMP study sites. Bolded species are USFWS Birds of Management Concern. Relative abundance and/or territory density for a given species was highest under optimal basal area ranges and the species was present under suitable ranges.

Management Considerations

Cerulean Warblers occur on forested lands throughout its range. Landowners desirous of keeping their lands in forested condition can do so using the economic benefits derived from productive forest management. In mature forest stands that have high cerulean densities and high nest success, the no-harvest option is most favorable for sustaining cerulean populations. In actively managed forests, there are opportunities to use forest management practices to mimic the structure and natural disturbance regimes of old-growth forests to enhance habitat for this species. The results from the CWFMP indicate that retaining RBA levels of ~40-90 ft^2/acre after harvesting trees in 25 acre harvest units in oak-dominated stands creates a forest structure that is generally favorable for ceruleans. Small-sized harvest stands (~10-27 acres) and their edges are not avoided by ceruleans.

Female Cerulean Warbler. Ohio DNR

In addition to enhancing stand conditions for ceruleans, small-scale harvests that result in intermediate levels of RBA are consistent with promoting oak regeneration and a diverse wildlife community. These harvests create habitat for early-successional birds, many of which are experiencing long-term population declines. For example, in northeast Pennsylvania, stands of regenerating timber attract Cerulean Warblers to use both the mature forest edge and adjacent residual trees in the harvest while providing breeding habitat for Golden-winged Warblers (*Vermivora chrysoptera*). Opening the canopy also can enhance habitat for many species of forest-dwelling bats. A study of bat use of the CWFMP treatments found increased bat foraging activity within partial harvests than in unharvested plots.

Important considerations for implementing harvests for ceruleans include the following:

Landscape-scale Considerations

Forest Cover

Some studies of forest songbirds have found decreased nest success in landscapes with a low proportion of forest cover. In heavily forested regions, the abundance and productivity of ceruleans and other forest songbirds appear to be more heavily influenced by stand structure than by landscape or edge effects. Thus, habitat enhancements for ceruleans located in heavily forested regions (>70% forest cover at the six mile scale) are more likely to be effective at attracting ceruleans and landscape context may have less influence on reproductive success.

Scale of Harvesting

Even in heavily forested regions, maintaining a significant portion of the management area as mature forest cover is important for sustaining populations of forest-interior birds because many forest-interior birds are sensitive to the amount of mature forest cover at larger spatial scales. In addition, several mature forest dependent species (e.g., Wood Thrush, Worm-eating Warbler, and Acadian Flycatcher) are likely to decrease in abundance at intermediate levels of RBA. Thus, where these species are high priority, maintaining about 50% of large forest blocks in the >50 year-old age class will provide structural complexity yet retain closed-canopy forest availability.

Stand-scale Considerations

Local Cerulean Density

Where cerulean density is relatively high (>5 territories/25 acre), immediate habitat enhancements are not necessary because harvesting may reduce reproductive success which may outweigh any increases in cerulean breeding density. Ideal locations to focus management efforts are where local cerulean densities are low (<5 territories/25 acre). If no ceruleans are present near the management site (within ~5 miles), they may be less likely to colonize the managed area.

White Oak dominated habitat. Fran Trudeau

Retain large diameter white and chestnut oak trees in any management scenario.

White Oak Dominance

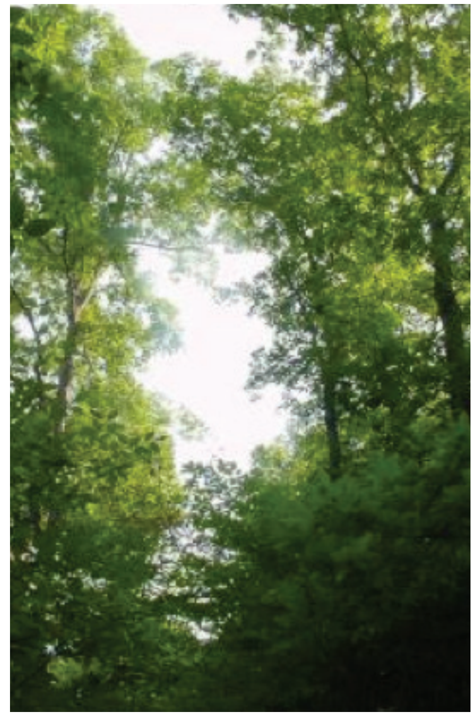
Maintaining white and chestnut oak dominance in the residual stand is a primary consideration in implementing management strategies for ceruleans. Thus, site productivity and the presence of sufficient advance regeneration of white and chestnut oaks are important considerations in management. Where feasible, favor white oak, chestnut oak, hickories, and sugar maple in the residual stand and do not retain red maple or red oaks. Retain some of the largest diameter individuals of the preferred species as residual trees. Prescribed fire at regular intervals may be necessary to promote oak regeneration, maintain small canopy gaps, and facilitate understory vegetation diversity.

Topography

In much of the Appalachians, harvests located along ridgetops and upper slopes are likely to be more effective in attracting ceruleans. Mesic, north- and east-facing slopes are often favored by ceruleans although other aspects are used.

Size of Canopy Gaps

Ceruleans preferentially use canopy gaps that are ~400-1000 ft^2 in size, particularly those with advanced vegetative growth within them. Thus, group-selection harvests that allow already established regeneration to grow into a stratified canopy may benefit this species.



Canopy gap in West Virginia. Scott Bosworth

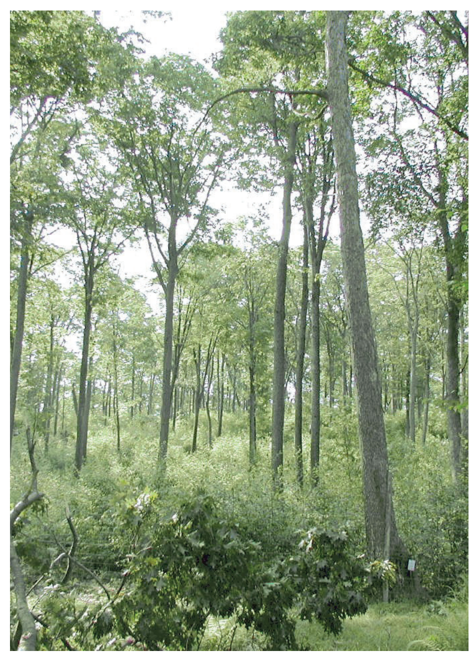
Temporal and Silvicultural Considerations

A number of different silvicultural practices could achieve residual basal areas in the harvested stand that are suitable for cerulean warblers (~40-90 ft^2/acre). Some additional considerations for various silvicultural treatments are below.

■ *Single-tree selection harvests* (our light harvest treatment) were less effective in increasing cerulean numbers and rapid canopy closure may limit the duration of suitable habitat. Single-tree selection with RBA above ~90 ft^2/acre also led to lesser nest success than harvests with lesser RBA. However, if single-tree harvest is favored by a landowner for providing income, cerulean densities would still be maintained particularly if non-preferred trees are removed and preferred oaks are retained.

■ *Group selection as part of an uneven-aged system* can improve cerulean habitat and would likely be effective longer than single-tree selection. The small group openings provide for diverse canopy structure and understory development. This approach has been shown to advance stands toward late successional structure beneficial to many avian species.

■ *Shelterwood harvests* are often compatible with promoting oak regeneration and, in the CWFMP, generally resulted in increased cerulean density and intermediate levels of nest success. However, complete overstory removal during the second stage of a shelterwood harvest will substantially reduce numbers of mature forest species including Cerulean Warbler, Wood Thrush, Acadian Flycatcher, and Worm-eating Warbler. If managing for forest birds, retain the residual canopy as long as possible and until adjacent habitat has been enhanced with shelterwood or other types of harvests and colonized by ceruleans.



Shelterwood harvest. Scott Stoleson

■ *Thinnings* as part of intermediate harvest treatments would open the canopy and provide the structure favored by ceruleans. These could take the form of a crown thinning or shelterwood seed cut.

■ *Modified even-age regeneration* can be used to create future opportunities for cerulean habitat improvement. Leaving large-diameter residual stems in a harvest unit can lead to development of two-aged stands. Such stands achieve more complex canopy structure earlier in their development than similar single-aged stands and the residual stems allow for some use of the stand by forest birds. Ceruleans had increased density in RBA of >~40 ft^2/acre.

■ *Crop-tree release* is a practice that is used to accelerate development of crop-trees on higher quality sites. The practice is typically applied in 15 to 20 year-old stands. It can allow for earlier canopy differentiation by accelerating growth of dominant stems. Impact on habitat suitability for ceruleans will not be immediate, but benefits should be seen as the stand develops and where earlier entry into the stand for commercial harvest is made possible.

Complex canopy structure in a deferment cut creates future opportunites for Cerulean Warbler habitat improvements. Doug Becker

Summary

Forest management that incorporates these guidelines and that is applied to oak-dominated stands in the Appalachian region can enhance habitat for Cerulean Warblers and other avian species, as well as other wildlife. Managers can choose a range of residual basal area targets depending on their priority avian species of interest.

For ceruleans, the RBA target range of ~40-90 ft^2/acre results in the most increases for the longest time period. A variety of silvicultural approaches can achieve this range. Where cerulean densities are high (>5 territories/20 acres), habitat management is not likely to be needed.

Landscape considerations are also important. These recommendations may be most beneficial in areas with high forest cover. They have not been tested in landscapes where forest cover is low.

Sitting pretty. Bill Hubick

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Appalachian landscape. Charlie Choc