

Amphibians and Fire in Longleaf Pine Ecosystems: Response to Schurbon and Fauth

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Introduction

Longleaf pine (*Pinus palustris* Mill.) savannas harbor a substantial proportion of the biodiversity of the southeastern United States, including a high proportion of species with endangered status (Means & Grow 1985; Noss 1988; Platt 1999; Means 2004), yet <3% of this once-widespread ecosystem remains (Ware et al. 1993). Longleaf pine savannas are home to at least 35 amphibian, 46 reptile, 45 bird, and 36 mammal species, many of which depend almost exclusively on longleaf pine savannas (Means 2004). Many researchers have concluded that these and other animals benefit from maintenance of longleaf pine savanna by prescribed burning (Mushinsky 1985; Engstrom 1993; James et al. 1997). Because natural fires no longer sweep through remaining tracts of longleaf pine, land managers must apply prescribed burns to simulate the rejuvenating effects of fire (Brennan et al. 1998). In the past two decades, the frequency of prescribed burns has been recommended on ever shorter intervals to restore habitat features eroded by decades of fire suppression (Engstrom et al. 1984; Cox et al. 1987; Weigl et al. 1989; Engstrom 1993; Tucker & Robinson 2003; Means 2004).

Schurbon and Fauth (2003) suggest decreasing the frequency of prescribed burns from the current 2–3 years to 3–7 years in order to “better maintain diverse amphibian and plant” assemblages in southern pine forests. Their suggestion was based on a 1-year study of amphibians sampled at 15 ponds within the Francis Marion National Forest, South Carolina. We believe that their study design and sampling effort were inadequate and, therefore, that their conclusions may be invalid. For reasons elaborated below, we discourage land managers from following their recommendation and suggest that a 1- to 3-year fire-return interval be the goal for prescribed fire in pine forests of the southeastern U.S. coastal plain.

Burn Treatment

Schurbon and Fauth claim that frequent prescribed burns are not beneficial to all amphibians of longleaf pine ecosystems, but they did not study frequency of burning. They studied time since last burn, a very different parameter. For example, one would expect differences in amphibian species richness and abundance between two ponds, both 3 years after burning, if the uplands around one had been subject to fire suppression for 30 years and the uplands at the other had burned every 3 years during the same period. The importance of fire-return interval in longleaf pine savannas was recognized when fire suppression resulted in habitat losses and declines in game species (Stoddard 1935; Croker & Boyer 1975; Croker 1987). Fire-return intervals of more than a decade resulted in the replacement of herbaceous ground cover and small pines with hardwood shrubs and trees (reviewed in Platt 1999). Amphibian and other vertebrate specialists of longleaf pine savannas cannot survive in hardwood forests (Means 2004). Not taking into account fire-return intervals on their burn plots renders their conclusions more applicable to the question of what happens to amphibians when fire is suppressed, rather than to the question of what effect fire-return frequency has on amphibians. It is entirely possible that a consistent fire-return interval of 5–12 years would result in a permanent shift in amphibian community composition with the loss of numerous anuran species.

The statistically significant differences Schurbon and Fauth found in amphibian species richness among the different treatments could have been produced by variation in the season of burn, which was not taken into account. Many researchers have shown substantial vegetative response to season of fire (Olson & Platt 1995; Menges & Hawkes 1998; Drewa et al. 2002a, 2002b), and we would expect amphibians to be affected as well, at least indirectly, because habitat structure is important

to them. We know nothing about whether amphibians migrate more successfully over recently burned sites or through a standing crop of several years' litter and recent growth, which provides protection from predators and desiccation. Dormant-season (winter) burns potentially affect winter breeders more than growing-season burns affect summer breeders because natural fires are ignited during the late spring and summer lightning season. Because of the lack of lightning in winter, winter fires are rare to nonexistent. If any of the amphibian responses to fire in their study were, in fact, attributable to fire, the responses may have been to season and not periodicity.

Effect of Upland Habitat

Schurbon and Fauth did not control for upland vegetation type or land-use history surrounding each study pond. On a large national forest such as the Francis Marion, many different silvicultural treatments are scattered over the landscape. Whether the uplands surrounding a pond are vegetated in native longleaf pine savanna or in a slash pine plantation or have been mechanically site prepared and clearcut will have a great effect on the amphibian fauna available to breed in ponds (Enge & Marion 1986; Means 2004; Means & Means 2004). In short, Schurbon and Fauth ignored the landscape context of their study ponds.

In South Carolina the longleaf pine specialist amphibians are *Ambystoma cingulatum*, *A. mabeei*, *A. talpoideum*, *A. tigrinum*, *Eurycea quadridigitata*, *Bufo quercicus*, *Hyla femoralis*, *H. gratiosa*, *H. squirella*, *Pseudacris nigrita*, *P. ocularis*, *P. ornata*, *Rana capito*, and *Scaphiopus holbrooki*, all of which breed in the type of ponds studied by Schurbon and Fauth. Five of these species were missing from their samples (*A. cingulatum*, *A. tigrinum*, *E. quadridigitata*, *H. gratiosa*, *R. capito*), and six others had some of the lowest total occurrences they reported (*A. mabeei*, *A. talpoideum*, *S. holbrooki*, *B. quercicus*, *P. nigrita*, *P. ornata*) (Schurbon 2000). At least six species they reported (*A. opacum*, *A. maculatum*, *Plethodon variolatus*, *H. chrysozelis*, *P. crucifer*, *R. clamitans*) occur primarily in hardwood forests and are not characteristic of longleaf pine ecosystems.

The presence of these species and the authors' statements that some ponds were located in hardwood stands, pine stands, and near a river-bottom hardwood bordered by small pine stands indicates so much variability in the terrestrial vegetation that we do not see how their results reflect amphibian responses to fire in true longleaf pine ecosystems. Moreover, despite the concerns we addressed above (and ignoring their data for the aquatic *Siren* and *Amphiuma*), Schurbon and Fauth (2003) encountered 19 out of 23 (82.6%) terrestrial species on their study sites that were 0–3 years postfire. In other words,

the majority of the species they observed apparently were present under the very fire regimes they advised against.

Variation in Reproductive Phenology and Detectability

Numbers of amphibian breeding adults and emigrating metamorphs at isolated ponds in southern pine forests vary considerably from one year to the next (Pechmann et al. 1991; Semlitsch et al. 1996). The number of species detected at specific breeding ponds and abundance also exhibit considerable annual and spatial variation (Dodd 1992; Semlitsch et al. 1996; Russell et al. 2002; Means & Means 2004). There is additional within-season variation in amphibian use of breeding ponds due to differences in reproductive phenology among species (Dodd & Charest 1988; Bridges & Dorcas 2000; Paton & Crouch 2002). As a result of this potentially extreme variation, estimates of amphibian species richness and abundance are quite dependent on sampling effort. We are not convinced that a 1-year study with three replicates per treatment and the degree of sampling effort expended by Schurbon and Fauth is adequate to account for this variation.

Sampling Effort

We believe the sampling effort employed by Schurbon & Fauth was inadequate and biased their results. Most of their data were generated from three partial drift fences placed around each of the 15 ponds. Based on our calculations from their published methods, these fences sampled <15% of the circumference at each pond. This left more than 85% of the circumference for amphibians to enter and exit the pond without being detected. Because movements of immigrating and emigrating amphibians at breeding ponds often are not random (Dodd & Cade 1998; Madison & Farrand 1998; DeMaynadier & Hunter 1999; Malmgren 2002; Johnson 2003), many individuals likely entered and exited the ponds undetected. By placing small fragments of a complete drift fence around only a small percentage of their study ponds' margins, Schurbon and Fauth could have missed recording some species at some ponds where they were present. Furthermore, they opened their traps at the drift fences for only 32 days during their 1-year study. Although they opened traps during and immediately preceding rainfall events, when amphibians are most active, this represents a sampling effort of <9% of the available days during a 1-year study.

Their efforts to sample amphibians in the ponds and to document chorusing frogs were also inadequate. Aquatic censuses were conducted only twice in a 24-hour period each time. Because aquatic salamanders are a significant component of the species richness of pond-breeding amphibians in South Carolina (Semlitsch et al. 1996; Russell

et al. 2002), and some have been shown to act as keystone predators (Fauth & Resetarits 1991; Fauth 1999; Eason & Fauth 2001), the extremely limited effort to detect such species in their study ponds may have significantly biased their results. They augmented trapping with call censuses to detect chorusing anurans after only four major rainfall events (Schurbon 2000). Each pond was visited for 5 minutes, and it seems that they did not visit ponds in random order and did not state what time of day or night they did the surveys, both of which can affect the ability of call surveys to detect anurans (Bridges & Dorcas 2000; Crouch & Paton 2002). This potentially biased their species-richness values because “calling censuses detected species not captured in treefrog shelters or that bypassed drift fences” (Schurbon & Fauth 2003).

Pond Hydroperiod and Rainfall

Hydroperiod is a primary source of variation in amphibian community structure, and it is highly correlated with rainfall, which has a strong influence on the timing of amphibian breeding migrations (Pechmann et al. 1989; Semlitsch et al. 1996; Skelly 2001). Thus, rainfall and pond hydroperiod have great impacts on the occurrence and detectability of amphibian species. Hydroperiods of the ponds studied by Schurbon and Fauth varied considerably. Because the Julian date of inundation has a profound influence on amphibian reproduction, pond hydroperiod should have been held constant among replicates. Although Schurbon and Fauth did not measure rainfall, it is safe to assume that there was also considerable variation in rainfall among the 15 study ponds. Despite the fact that these environmental covariates are known to explain a substantial proportion of variation in amphibian community structure, Schurbon and Fauth did not incorporate rainfall or hydroperiod directly in their statistical analyses and did not consider the influence of these factors on the probability of species detection.

Abundance, Counts, and Lack of Independence

Schurbon and Fauth used a concept of “abundance” based on count data without attempting to determine associated detection probabilities. They used a “minimum abundance” variable (that is, counts) to predict amphibian “responses” to the number of years since the last burn. In effect, Schurbon and Fauth confuse the concept of abundance with using counts as an index of population size. Abundance and counts are actually related, as $\hat{N} = C/\hat{p}$, where \hat{N} is abundance, C is counts, and \hat{p} is detection probability. They are not equivalent.

In amphibian populations, the number of individuals often fluctuates widely over time (Pechmann et al. 1991; Pechmann & Wilbur 1994; Marsh 2001; Green 2003) be-

cause of the many variables that influence activity and thus detection probabilities (Dodd 2003). Because counts represent some unknown fraction of the sampled population, estimates of species-specific detection probabilities are required to estimate abundance of species. Unless detection probabilities are not significantly different, there is little reason to suppose that count data represent even a minimal index of abundance (Thompson et al. 1998; Hyde & Simons 2001; Schmidt 2003).

Additionally, it is likely that Schurbon and Fauth’s count data are not independent because animals were not marked. Because many individuals use the same retreat sites, such as PVC pipe shelters, throughout a season (e.g., Boughton et al. 2000) and may be repeatedly captured in pitfalls, the same individuals could have been counted numerous times during their study. In addition, amphibians may grow substantially from one season to the next, also making it possible to count an individual more than once within a 1-year sampling period. Because of these problems, Schurbon and Fauth’s “abundance” data cannot be used as a valid response variable to measure the effects of prescribed burns on amphibian populations. As a result, their species evenness values are also suspect.

Applying Recommendations Elsewhere

Schurbon and Fauth contend that random selection of sites allows them to extrapolate their results throughout the southeastern coastal plain longleaf pine forests. The sampled population (FMNF) undoubtedly contains a subset of the available habitats and amphibian species contained within the entire longleaf pine forest community. However, the restriction of sampling effort to a small number of ponds at a single geographic site ensures that certain elements of the entire forest community—amphibian species, habitats, burn histories, and other environmental variables—had a probability of zero of being sampled by Schurbon and Fauth. This sampling bias imposes severe limitations on the ability of the authors to extrapolate their results beyond FMNF. Furthermore, unless the U.S. Forest Service randomly assigned burn treatments spatially across the forest, Schurbon and Fauth in fact did not sample a truly random selection of ponds, possibly further restricting the extent of their sampling inference.

Schurbon and Fauth cannot extrapolate their results because they have not sampled a representative portion of the entire longleaf pine community. In effect, they have extended their recommendations far beyond the range of valid statistical inference (Thompson et al. 1998; Williams et al. 2002). As Thompson et al. (1998) stated, “An important point here is that we can only make statistically based statements about the animals within the sampling frame or sampled populations.” We contend that the widespread advocacy of a land-management practice based on a single study, such as the burn frequencies

discussed by Schurbon and Fauth, is not valid without a significant amount of additional research.

Conclusion

Six species of salamanders and 11 frogs that are specialists in longleaf pine ecosystems do not occur in hardwood forests (Guyer & Bailey 1993; Dodd 1995; Means 2004). Many studies have demonstrated how rapidly southern temperate hardwoods (Platt & Schwartz 1990) or evergreen wetland shrubs (Drewa et al. 2000a, b) invade longleaf pine savanna (Waldrop et al. 1992; Hermann 1995; Glitzenstein et al. 1995, 2003) in the United States. Other studies have shown how rapidly longleaf pine-adapted vertebrates decline when the fire-return interval is extended (Engstrom et al. 1984; Tucker & Robinson 2003; U.S. Fish and Wildlife Service 2003; Means 2004).

This is important because terrestrial vertebrates such as amphibians respond to vegetation in specific ways. In closed-canopy forests, where most of the primary productivity takes place high in the trees, vertebrates are volant, scansorial, and arboreal. Herbivores that feed far above ground in the branches are limited in body size. Terrestrial amphibians and reptiles must feed on detrital invertebrates and are therefore specialized to burrow in litter and feed on a detritivore-based food web. In open-canopied longleaf pine savannas, by contrast, a large percentage of ecosystem primary productivity takes place at ground level, supporting a diverse invertebrate herbivore fauna. Insectivorous amphibians feed on a different species ensemble of arthropods on the ground and in the low herbaceous vegetation. The consequences for plant succession of increasing the fire-return interval therefore have a strong impact on the types of amphibians found on a given site.

Because of the poor study design and sampling effort of Schurbon and Fauth, their results may not represent amphibian response to fire or, at the very least, to periodicity of fire. Land managers should not be quick to accept their results and conclusions. We suggest that better-designed studies are required to assess amphibian responses to fire. Meanwhile, because longleaf pine savannas have experienced decades of fire exclusion, managers should not be restricted in the use of frequent prescribed burns to control hardwood encroachment in restoring longleaf pine savannas and adjacent vegetation that depend on fire in the longleaf ecosystems (Means 1996).

Ultimately, site-specific, high amphibian diversity, per se, or even abundance is not important in longleaf pine management. What is important is returning the few remaining longleaf pine ecosystems to some semblance of their natural condition and allowing component species to adjust their abundance and distribution throughout the longleaf pine landscape to the population levels and dispersion patterns to which they are adapted.

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