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Managing the Southern Pine Forest—Retained Wetland Interface for Wildlife Diversity: Research Priorities

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Abstract Forest certification programs require program participants to manage for biodiversity using science-based information. Management at the interface of retained wetland features and plantations provides opportunities to enhance wildlife diversity on commercial pine forest lands. We review the scientific literature to document how wildlife in managed pine forests might benefit from retention of isolated wetlands and riparian zones, and potential effects of forest management on conservation of wetland-associated wildlife on managed pine forests of the southern USA. We suggest research goals and methodologies to address information gaps critical to improved management. Many available studies lacked inferential power, and most depended on measures of diversity, richness, or abundance rather than community similarity or demographic measures of fitness. Observational studies have yielded potential hypotheses that should be tested with manipulative experiments. Demographic measures of fitness should replace potentially misleading measures of abundance or density, and diversity measures supplemented with comparisons of community similarity. Researchers should institute long-term studies to account for temporal variability. Multi-scale analyses would help determine appropriate management scale for isolated wetlands and the utility of riparian areas and associated streamside management zones as dispersal corridors. Landscape-level models would facilitate long-term planning and provide a framework for adaptive management.

Keywords Forest management · Isolated wetlands · Riparian areas · Streamside management zones · Sustainable forestry

Introduction

Forest management effects on biodiversity are of increasing interest to many segments of the public, especially as more area comes under intensive management for fiber production (Guynn et al. 2004; Brockerhoff et al. 2008). Plantation management in particular has been of interest among biologists because of the perception, earned or not, that monocultures managed primarily for wood products lack key components of habitat for some wildlife species (Stephens and Wagner 2007). Recently, sustainable forestry certification systems have emerged to ensure that the ecological effects of management are noted, studied, and those results integrated back into improved management models. Certification systems typically contain provisions related to the management and biodiversity contributions of retained habitat features, including wetlands. The contributions of wetland habitat features to vertebrate diversity in managed pine (*Pinus* spp.) forests are important to understand and document so that certification goals can be met.

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The U.S. Supreme Court decision, Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers (No. 99-1178) reduced protection for isolated wetlands under the Clean Water Act by requiring them to have a "significant nexus" to navigable waters. This and subsequent rulings have created a situation where the requirements for what constitutes a jurisdictional wetland are unclear (Frankel 2007; Murphy and Johnson 2007; Leibowitz et al. 2008). Forest land certification systems may provide the greatest level of protection for isolated wetlands on commercial forests. For example, the Sustainable Forestry Initiative® (SFI) requires program participants to institute programs for identification, management, and protection of water bodies and riparian areas (SFI Sustainable Forestry Board 2004). Evidence of wetland contributions to wildlife diversity can encourage appropriate steps to determine management actions that conserve those contributions.

Timber management in and around wetland features is guided by state-specific Best Management Practices (BMPs), with the primary goal of protecting water quality (Blinn and Kilgore 2001; Aust and Blinn 2004). Buffer zones are commonly required to protect water bodies from potential negative impacts of forest management, and can be managed to benefit wildlife. Many commercial forest land owners institute buffers in accordance with state-mandated BMPs as part of their forest certification requirements. However, BMPs do not generally consider wildlife habitat directly, so management practices in and around streams and isolated wetlands may not conserve habitat necessary for associated wildlife communities.

We instituted this review to gather and synthesize information from the scientific literature to determine the state of knowledge regarding contributions of retained wetland features on vertebrate diversity and the effects of timber management on the conservation of those contributions in managed pine forests of the southern United States. This information is especially important given that 16% of all southern timber lands are under pine plantation management (Wear et al. 2007). We focus specifically on two features: isolated wetlands and riparian areas. Isolated wetlands are subject to a variety of management actions, ranging from buffer zones to complete overstory removal, that modify their relationship with associated uplands. Riparian areas in managed pine forests interface with uplands through the medium of streamside management zones (SMZs), and we address them primarily in that context. Following a brief description of each wetland type, we discuss their relationships with vertebrate wildlife in southern pine forests and the known effects of forest management on associated wildlife communities. We also identify information gaps and suggest future research priorities to enable forest managers to better conserve wildlife diversity along the pine-wetland interface.

Isolated Wetlands

Description

The term "isolated wetlands" refers to relatively small, non-permanent wetlands, natural or anthropogenic, that are geographically disjunct from other wetlands (Tiner 2003). In the Southeast, isolated wetlands include: Carolina bays, pocosins, Coastal Plains ponds, gum ponds, cypress domes, sinkhole wetlands, woodland vernal ponds, inter- and intra-dunal wetlands, seepage slope wetlands, inactive floodplain wetlands, natural ponds, and excavated ponds (see Tiner 2003 for descriptions). Perhaps because of controversy over Federal legislation and regulations, much of the recent research has focused on isolated wetlands <4 ha in surface area (Semlitsch and Bodie 1998; Gibbs 2000; Snodgrass et al. 2000b; Zedler 2003). Given the variety of isolated wetlands under this umbrella, generalizations will not be entirely applicable for all types of isolated wetlands.

The water source for isolated wetlands varies from rainwater to spring-fed. Subsurface flow or intermittent overflow may link isolated wetlands hydrologically with other wetlands (Leibowitz 2003; Tiner 2003; Whigham and Jordan 2003). Hydrology of isolated wetlands is often ephemeral (Snodgrass et al. 2000a; Brooks and Hayashi 2002), although some may dry completely only following long periods of drought (Sharitz 2003). Most rely on precipitation events for filling, and are thus most likely to fill in late winter to early spring, followed by loss of water through evapotranspiration leading to complete drying by mid- to late summer, though occasional late-season precipitation may lead to refilling for a time (Semlitsch et al. 1996).

Vegetation cover in isolated wetlands varies widely, from none in open ponds to tree overstories in cypress domes, pocosins, and gum ponds (Tiner 2003). Vegetation moderates wetland temperatures through shading, and provides organic inputs while reducing sedimentation and nutrient inputs from land erosion (Castelle et al. 1994). Hydrology, soil texture, depression size and disturbance history have been suggested as primary drivers of plant succession (Kirkman et al. 2000; De Steven and Toner 2004; Casey and Ewel 2006). Succession may progress from open water through forested cover, influencing composition of associated vertebrate communities (Skelly et al. 1999).

Wildlife Relationships with Isolated Wetlands

Forest management effects upon mammalian and avian use of isolated wetlands are not emphasized in the literature. Clark et al. (1985) trapped or observed 40 mammalian species in pocosins, Carolina bays, and associated communities, which can serve as refuges for black bear (*Ursus americanus*; Richardson and Gibbons 1993), and smaller mammals such as bobcat (*Lynx rufus*) and marsh rabbit (*Sylvilagus palustris*; Monschein 1981). Isolated wetlands provide commuting and foraging areas for bats in southern forests (Wilhide et al. 1998; Menzel et al. 2005a, b). Mitchell et al. (1995) captured 10 small mammal species in undisturbed pocosin forests in North Carolina, and did not detect differences in small communities between undisturbed pocosin forests and pine plantations possibly because pocosin-like habitat persisted in managed stands. Wetlands are important to waterfowl and other avian species. Seventy percent of avian species associated with temperate forest habitats in southeastern North Carolina were detected in Carolina bays (Mamo and Bolen 1999). Seasonal drying of isolated wetlands concentrates prey species for wading birds (Ogden et al. 1976; Kushlan 1979), including populations of the endangered wood stork (Coulter and Bryan 1993), and small ponds may support large wading bird rookeries (Moler and Franz 1987; Richardson and Gibbons 1993). During periods of drought, isolated ponds may provide refuge to bird species dispersing from affected areas (Beissinger and Takekawa 1983).

Herpetofauna are the vertebrates most closely associated with isolated wetlands. Lack of large predatory fish in seasonal ponds removes a significant source of predation on amphibian larvae (Wilbur 1980; Porej and Hetherington 2005), and many amphibian species are adapted specifically to breed in temporary ponds (Wellborn et al. 1996). At least 10 anuran and five salamander species in the Southeastern Coastal Plain are dependent on isolated wetlands for breeding sites (Moler and Franz 1987). Surveys have shown great herpetofaunal species richness in isolated wetlands imbedded in pine forests. Dodd (1992) monitored herpetofaunal use of a temporary pond in a north Florida longleaf pine sandhills community, capturing 16 amphibian and 26 reptile species. A survey of 444 seasonally flooded ponds on private, industrial forest lands over 35 counties in south Georgia, south Alabama, and north Florida identified 16 salamander, 24 anuran, 34 reptile, and 37 fish species (Wigley et al. 1999). Moreover, pond size may not predict species richness. Wetland size was not related consistently to richness, diversity, or evenness of herpetofauna, nor were herpetofaunal communities more similar at ponds of similar size in an industrial pine forest in the South Carolina Coastal Plain (Russell et al. 2002a).

Wildlife species associated with isolated ponds are also dependent on upland habitat (Dodd 1996; Buhlmann and Gibbons 2001; Gibbons 2003; Regosin et al. 2005). Semlitsch and Bodie (2003) estimated core habitat for amphibian populations ranging from 159–290 m from the wetland edge, and 127–289 m for reptiles, suggesting that forest management within these distances may have serious influence on herpetofauna. Several studies have investigated the impacts of upland management surrounding isolated wetlands on herpetofauna in southern pine forests (Table 1). In general, the studies seem to indicate that upland management may have little effect on herpetofaunal diversity, but that individual species and community composition can be greatly affected. It must be noted that many of these studies have limitations on their inferential power due to pseudo-replication, short duration, lack of pre-treatment data, or observational rather than experimental nature. However, they do provide fertile ground for further experimentation to confirm or adjust their findings.

Russell et al. (2004) pointed out that southeastern Coastal Plain forests, which have evolved with fire to create large areas of sub-climax pine forests, are likely to harbor herpetofaunal communities different than those in hardwood dominated landscapes such as the southern Appalachians, where many studies of forest management impacts on herpetofauna have taken place. Experiments in Coastal Plain forests have found little reason as yet for concern over standard silvicultural practices in pines. Mole salamanders (*Ambystoma talpoideum*) in South Carolina reportedly may be able to maintain themselves in clearcut areas as long as there is sufficient cover in the form of coarse woody debris (CWD) or leaf litter (Chazal and Niewiarowski 1998). Russell et al. (2002b) found only temporary declines in abundance of both snakes and turtles in clearcut and site-prepared stands adjacent to small ponds on Coastal Plain industrial pine forests; other herpetofaunal groups were either unaffected, or, in the case of bronze frogs (*Rana clamitans clamitans*), may have benefited from treatment. These studies, while encouraging, do not yet represent a broad consensus of opinion. Observations of great potential impact on individual species, such as those by Means et al. (1996), caution against blanket acceptance and widespread application of a few short-term studies.

Microclimate may be an important factor for determining amphibian abundance and distribution (deMaynadier and Hunter 1995; Harper and Guynn 1999; Goldstein et al. 2005). Microsite variables may explain the seeming resilience of some herpetofauna in disturbed pine uplands. Chazal and Niewiarowski (1998) hypothesized that the similarities between sites in soil moisture, organic content, and litter contributed to the lack of treatment effect on mole salamanders exposed to conditions in a 4-month-old clearcut compared with salamanders penned in an adjacent, undisturbed 40-year-old pine forest. Mosely et al. (2004) suggested that either CWD or pine litter of sufficient depth contributed to adequate microclimate conditions for mole salamanders under mature pines. Conversely, Russell et al. (2002a, b) found no correlation between either litter or CWD and reptile or amphibian diversity before or after harvest and site preparation; however, no comparisons were attempted at the species level, so microhabitat effects may have been obscured by opposing species preferences.

Table 1 Studies of upland management impacts on herpetofauna associated with isolated wetlands in southern pines

Study and context	Experimental design	Key results and observations	Comments
Chazal and Niewiarowski (1998)	Monitored mole salamanders (*Ambystoma talpoideum*) placed in 10×10-m enclosures for 5–6 months. Analyzed for both demographic and environmental parameters.	No differences found for any demographic parameter, including: recapture rate, body length and mass, nonpolar lipids for carcasses and eggs, clutch size, and time to emergence.	Salamanders in clearcuts were not exposed to the actual disturbance.
Clearcut and 40-yr-old loblolly plantation, within 300 m of a Carolina bay in South Carolina.		Authors conjectured that similarities between sites in soil moisture, organic content, and litter may have contributed to the lack of treatment effect.	Pseudo-replicated.
Russell et al. (2002b)	Divided area surrounding wetlands into clearcut, clearcut with site preparation, and reference stands. Herpetofaunal communities sampled before treatment and 0.5 years and 1.5 years post-treatment.	Abundance of snakes and turtles was reduced in both site prepared and clearcut stands relative to controls 0.5 months post-treatment; abundance of both were similar to control stands 1.5 years following treatment. Bronze frogs increased in site prepared and clearcut areas 1.5 years post-treatment.	Pre- and post-treatment sampling period of 18 months.
18–25-yr-old industrial pine forests surrounding 5 small isolated wetlands in coastal South Carolina.		No treatment effect was detected for overall abundance, species richness, or community similarity of any reptile or amphibian group. Found no evidence to suggest amphibians emigrated from clearcuts into reference stands.	
Enge and Marion (1986)	Watersheds assigned to reference, and to clearcut with either minimal or intensive mechanical site preparation. Sampled 2–3 years post-treatment.	Species richness did not differ among treatments for amphibians; reptile species richness was lower in the intensively prepared site.	Pseudo-replicated.
3 watersheds in 40-yr-old slash pine in Florida.		Abundance of all herpetofauna was 3× greater in the reference than in either treatment plot, and was still significant after removal of the most abundant species, which was found almost exclusively in the control forest.	No pre-treatment comparison.
Means et al. (1996)	Observational study monitored cross-highway movement of flatwoods salamander (*Ambystoma cingulatum*) for 3 periods over 23 years following conversion of one area to intensively managed plantations.	Across road migration decreased from 7.9/hr to 0.1/hr.	No pre-treatment observations.
Intensive slash pine silviculture across highway from National Forest in Florida		Authors discounted drought, road mortality, collecting, and acid rain as possible causes, and theorized that slash pine silviculture had degraded salamander habitat by altering hydrology, soil infrastructure, plant communities, and terrain.	No replication or randomization.
Greenberg 2001	Surveyed gopher frogs (*Rana capito aesopus*) at 8 ponds embedded in either hardwood-invaded or savanna-like longleaf pine upland matrices.	Adult use of ponds did not differ by upland type. Juvenile recruitment was greater at ponds in savanna-like matrix. Recruitment was affected by both landscape and within-pond scale factors.	Based on 5 years of monitoring.
Greenberg and Tanner 2005	Surveyed oak toads (*Bufo quercicus*) at 8 ponds embedded in either hardwood-invaded or savanna-like longleaf pine upland matrices.	Both adult pond use and juvenile production were greater at ponds in the hardwood-invaded matrix.	Based on 10 years of continuous monitoring.

Hanlin et al. (2000)	Surveyed amphibian populations daily in each forest type for 3 years following restoration of the bay. Compared results among years and to surveys performed 18 years earlier (Bennett et al. 1980).	The hardwood site had the greatest abundance for all years combined, but also had the lowest diversity. The slash pine site expressed the highest diversity for each year and overall.	No randomization.
3 forests types (40-yr-old slash pine, 41-yr-old loblolly pine, and >60-yr-old hardwood) surrounding a restored Carolina bay in South Carolina.		83% reduction in overall abundance (regardless of forest type) between summer surveys in 1977–1978 and 1994–1996. The authors attributed this to pond restoration activities, which eliminated leaf litter and woody debris from the bay and its margins, as well as altering adjacent vegetation from mesic shrubs to an open herbaceous community.	Pseudo-replicated.

Riparian Areas

Description

Riparian areas are transitional areas between perennial, intermittent, and ephemeral streams and terrestrial ecosystems that exhibit gradients in biophysical conditions, ecological processes, and biota. Riparian areas are beneficial for a wide spectrum of taxa, and often contain a different species pool if not greater species richness relative to uplands (Sabo et al. 2005; Palmer and Bennett 2006). Streamside vegetation protects aquatic systems from upland disturbance by trapping sediments and chemicals, stabilizing stream banks, and moderating water temperature extremes with shade (Lee et al. 2004). Streamside management zones (SMZs) are areas of vegetation retained along water channels primarily to protect water quality. Although Best Management Practices for SMZs vary somewhat among states, recommended zone width is generally dependent on slope and stream classification. Thus, SMZs will contain varying proportions of riparian and upland vegetation, and may be entirely one or the other. However, SMZs represent the most common transition from stream to upland on managed forests, and riparian areas nearly always interact with managed pine through the artificial construct of the SMZ.

Wildlife Relationships with SMZs

Comparisons of vertebrate communities between landscapes with and without riparian areas have been reported in many regions, with breeding bird communities particularly well-studied (Pais et al. 1988; Doyle 1990; Whitaker and Montevecchi 1997; Wiebe and Martin 1998; Bub et al. 2004; Lehmkuhl et al. 2007). However, direct comparisons are lacking for southern pine forest landscapes, and much of our supposition that riparian areas increase species richness and diversity is inferred from our knowledge of individual species biology. For example, Azevedo et al. (2006) modeled SFI practices in an East Texas watershed and reported that implementation of SMZs was particularly beneficial in improving habitat suitability for an array of indicator wildlife species.

Recommendations for SMZ widths to protect water quality may not be appropriate for meeting some objectives for wildlife communities. Managers typically determine appropriate and unique buffer widths on the basis of location, channel size, riparian slope, disturbance intensity, and management goals, as well as landscape context. Wenger (1999) recommended that, for optimum wildlife benefits, buffers should be at least 100 m to each side in some locations throughout a landscape. Wider buffers (150–400 m; Semlitsch and Bodie 2003) may be more appropriate for some management purposes, for example, connecting wetlands to high quality adjacent sites. On the other hand, narrower buffers (30 m wide; Wenger 1999; Lee et al. 2004) may be all that is necessary to meet some biodiversity-related objectives and to preserve water quality. Wigley and Melchiors (1994) urged caution when recommending riparian buffer widths for meeting biodiversity objectives, and concluded that variable-width riparian buffers may be more desirable in many cases to allow operational flexibility and incorporation of important habitat features, such as mature trees and snags, that may not fit within a fixed width buffer. Indeed, a mixture of protected widths, from 100 m on each side for interior species to near-streamside harvesting for species that may use sparse vegetation such as nesting turtles (Russell et al. 2004), probably matches the historical mosaic created by fire and wind disturbance.

Few studies have directly examined herpetofauna use of SMZs bordering southern pines (Table 2). Rudolph and Dickson (1990) documented more herpetofauna in wider zones of 50–95 m than in narrow zones adjacent to young loblolly pine plantations (2–4 years) in Texas. Talley and Crisman (2006) only detected differences in larval salamanders 1 year after timber harvest and site preparation compared to reference conditions along streams in two watersheds in Georgia. Although Fogarty (2005) studied older mixed forest stands (>25 years) extending at least 120 m in width from a stream's edge rather than buffers, he recommended buffers of 25–50 m due to high amphibian concentration at these widths in Mississippi. In contrast, reptiles were distributed evenly throughout riparian areas.

Avian response to SMZ width in pine systems has been studied in several locations. In east Texas, bird abundance, although not richness, in SMZs adjoining young loblolly plantations increased up to 95 m width (Dickson et al. 1995; Conner et al. 2004). Numerous species exhibited either linear or threshold relationships with SMZ width, with early succession species more prominent in narrower SMZs and mature forest associates in wider ones (Dickson et al. 1995; Conner et al. 2004). Breeding and wintering avian density was greatest in narrow buffers (15–18 m) bordering a pre-canopy closure pine plantation in Georgia, and all SMZs had greater bird density than the pre-canopy closure pine plantation (Thurmond et al. 1995). Although mature riparian control contained the greatest density of forest interior specialists and Neotropical migrant species, SMZs contained edge-interior and edge species not present in plantations (Thurmond et al. 1995). Two of the avian studies were performed in whole (Hodges and Krementz 1996) or in part (Kilgo et al. 1998) along large river bottoms, and thus may not reflect conditions typical of SMZs in most managed pine landscapes, where it is unlikely that harvested bottomlands would be converted to pine silviculture. These studies indicate that richness and abundance of Neotropical migrants along stream bottoms is directly related to area of bottomland vegetation, and thus retention of adequate riparian vegetation within SMZs can increase overall avian richness in managed pine landscapes.

Studies of small mammals associated with SMZs in southern pine landscapes generally suggest that narrow buffer widths are sufficient for studied species. In Arkansas, Miller et al. (2004) found that SMZs <20 m wide in pine plantations had greater small mammal abundance and species richness than wider zones. Additionally, species richness and catch per unit effort were greater in SMZs adjacent to young (pre-canopy closure) and thinned plantations than closed canopy plantations. Thus, the structure of plantations adjacent to SMZs appeared to influence small mammal community structure within the buffers more than buffer width (Miller et al. 2004). In east Texas pine plantations before crown closure, small mammals were more abundant in narrow (<25 m) than medium (30–40 m) or wide (>50 m) buffers, possibly due to dense, brushy vegetation, abundant seeds, and logging slash in the narrow buffers (Dickson and Williamson 1988). In contrast, Thurmond and Miller (1994) documented that total small mammal abundance during summer was greatest in mature riparian forests, whereas species composition varied by both season and habitat type (i.e., mature forest, buffers, and young pine plantation).

Information Gaps and Research Directions

Although the potential contributions of retained wetland features to wildlife diversity in managed pine forests have been well-documented, there is yet a need to examine community response to habitat structure and composition. The effect of natural succession in isolated wetlands on associated wildlife communities has not been systematically studied in the South. Knowledge of community change in relation to succession would provide a baseline for research into active management of isolated wetlands. Studies similar to Skelly et al. (1999) should be conducted, perhaps followed with long-term monitoring. Width and landscape context are commonly addressed in studies of SMZs. However, because SMZs may comprise various proportions of riparian and upland vegetation, it is important to understand also the interaction of width and landscape with SMZ composition. Vegetation differences among the SMZ buffer classes, influenced by riparian zonation and then transition to upland vegetation, need careful consideration and standardization in order to separate the effects of vegetation characteristics from buffer zone width. We would expect a shift in wildlife species from a strictly riparian to a strictly upland SMZ, and the best management for conserving each community may differ markedly.

Table 2 Studies of wildlife response to streamside management zone widths in southern pine forests

Study and context	Species group	Variable and technique	SMZ Width (m)	Key results and observations	Comments
Fogarty (2005) 21 mature mixed forest sites extending from streams in Mississippi.	Amphibians and reptiles	Abundance; visual searches along transects	Transects sampled 0, 25, 50, 75, and 100 m from streams	Amphibian abundance decreased with distance from streams. Reptile abundance was generally even throughout.	Observational study on dispersion from streams.
Rudolph and Dickson (1990)	Amphibians and reptiles	Abundance of taxonomic (4) and ecological (2) groups; drift fences with funnel traps or artificial shelters along transects plus visual searches	Narrow (0–25), medium (30–40), wide (50–95)	Greatest abundance of all groups in wide SMZs.	Only 2 replications of each width.
6 young (2- to 4-year-old) pine plantations in eastern Texas.				In plantations, abundance of reptiles overall and of lizards was greatest next to wide SMZs.	No accounting for differing vegetation and ground cover characteristics. Short-term study (2 years) Differing areal extent of SMZs.
Dickson et al. (1995)	Birds	Abundance; transects	Narrow (15–25), medium (30–40), wide (50–95)	Avian abundance increased with riparian buffer width.	No accounting for differing vegetation characteristics.
9 young (2- to 5-year-old) pine plantations in eastern Texas.					2-year study Bird transects extended beyond SMZ widths. Differing areal extent of SMZs.
Hodges and Krementz (1996) Floodplain swamp adjacent to pine plantations in Georgia.	Birds	Species richness, abundance of 6 focal species; points along transects	Narrow (< 350), medium (400–700), wide (> 1000)	Avian species richness and abundance of 3 species increased with SMZ width.	Unequal subsampling (points per transect) and replication (transects per width class).
Kilgo et al. (1998) Bottomland hardwood forest bordering closed-canopy pine forests or field-scrub in South Carolina.	Birds	Species richness and abundance; point counts	< 50, 50–150, 150–300, 300–1,000, >1,000	Species richness increased with buffer width; abundance was greatest in the narrowest and widest widths.	Possible comparison problems due to varying proportion of edge and areal extent among wide-ranging width classes.
Thurmond et al. (1995)	Birds; spring and winter	Density of total number bird species and ecological groups (4); points along transects	Narrow (15–18), medium (28–30), wide (49–53 m)	Breeding and wintering avian density was greatest in the narrow SMZ.	Pseudoreplicated.
Young pine plantation with chemical site preparation compared to mature forest control in Georgia.				Density was lowest in the plantation; the greatest density of interior species was found in the mature forest control.	Small maximum width. Pines harvested from SMZs. No accounting for differing vegetation characteristics. 2-year study. Varying widths of bird survey transects.
Miller et al. (2004)	Small mammals	Relative abundance, species richness, evenness, diversity; removal trapping along transects	1–20, 21–40, 41–60, 61–100, >100	Narrow SMZs had greater small mammal abundance and species richness than wider SMZs.	Richness and variance possibly affected by sampling intensity (different class width area and number of transect lines).

Table 2 (continued)

Study and context	Species group	Variable and technique	SMZ Width (m)	Key results and observations	Comments
3 types of pine forests (young, closed canopy, and thinned) and natural riparian forests in Arkansas.				Species richness and abundance were greater in SMZs next to young and thinned plantations than in closed canopy plantations.	Pines harvested from SMZs.
Dickson and Williamson (1988) 6 young pine plantations in eastern Texas.	Small mammals	Abundance; live trapping along transects	Narrow (<25), medium (30–40), wide (>50)	Greater abundance of small mammals in narrow SMZs.	Each width replicated only twice. No accounting for differing vegetation characteristics. 2 year study. Differing areal extent of SMZs.
Thurmond and Miller (1994) Young pine plantation with chemical site preparation compared to mature forest control in Georgia.	Small mammals	Abundance; removal traps along transects	Narrow (15–18), medium (28–30), wide (49–53 m)	Abundance and composition varied by treatment type.	Pseudoreplicated. Pines harvested in SMZs. No accounting for differing vegetation characteristics in SMZ classes. Unbalanced sampling, Differing areal extent of SMZs.

Most of the studies we reviewed used measures of abundance, diversity, or richness to differentiate treatment effects. However, none of these metrics indicate the degree to which community composition may be altered by management actions. Likewise, they leave dark the degree to which wetlands actually contribute to diversity and richness in the forest as a whole, which is surely valuable information. Future studies should consider including easily calculated measures of community overlap so that more complete evaluations of potential management effects are available to forest managers. Because herpetofauna populations associated with isolated wetlands can exhibit radical population swings due to annual variance in hydroperiod (Pechmann et al. 1989; Snodgrass et al. 2000a), demographic studies are crucial to take knowledge beyond the potentially misleading results of abundance and density (Van Horne 1983). The potential for SMZs to provide habitat for substantially different communities than surrounding forest (Sabo et al. 2005) needs to be explored more fully, and demographic measures of fitness should be included in studies of species encountered in SMZs to determine whether SMZs act as sources or sinks. In addition, naturally occurring peaks and valleys in populations necessitate studies of several years duration to ensure that results are not clouded by temporal variability.

Many studies of wildlife associated with isolated wetlands have been observational, often without replications or pretreatment data (deMaynadier and Hunter 1995; Russell et al. 2004). While limited in inferential power, they have been useful for producing hypotheses which should now be tested experimentally. For example, Means et al. (1996) hypothesized that silvicultural actions were responsible for fewer observed movements of flatwoods salamander (*Ambystoma cingulatum*) across a road, and that salamander presence in cypress ponds within the plantation matrix was due to a 1-ha unmanaged buffer zone surrounding each pond. A long-term manipulative experiment to investigate buffer zones of different widths or managed with various silvicultural treatments would allow these hypotheses to be tested and would potentially elucidate strategies for integrating salamander management with plantation management.

The effects of forest management actions on microclimate features should be further investigated to determine their impacts on upland herpetofaunal populations. Comparisons of microhabitat use between managed pine and natural pine or pine-hardwood systems coupled with studies of silvicultural impacts on microhabitat elements may indicate which silvicultural methods best provide microhabitat elements important to herpetofauna. Candidate features for study include CWD, hardwood shrubs, soil pH, and leaf litter (deMaynadier and Hunter 1995). Landscape-level diversity may be significantly enhanced by management of habitat elements within SMZs that prove more difficult to accommodate in pine plantations. Because SMZ management may include some level of harvest (Blinn and Kilgore 2001), it is important to quantify the habitat contributions of large trees in SMZs as nesting sites, hunting perches, mast producers, den trees, and sources of snags and coarse woody debris. Comparisons of wildlife presence, abundance, and fitness among stands with differing levels of SMZ harvest would help identify levels of commercial use commensurate with biodiversity goals.

Investigators should consider multiple scale analyses, as the impacts of both isolated wetlands and SMZs are likely to be apparent beyond the immediate environs of the stand in which they occur. Isolated wetlands are subject to wide variance in availability and productivity due to variable precipitation patterns. Recolonization of an area by an extirpated species may depend on periodic emigration from other ponds within the species' dispersal distance. Some reptile species may travel long distances, using several wetland sites over the course of a year (Joyal et al. 2001; Roe et al. 2004), and thus are probably better thought of as a single population than as metapopulations (Smith and Green 2005). Management of single isolated wetlands may therefore be less important (at least for herpetofauna) than management of clusters of ponds (Gibbs 2000; Joyal et al. 2001; Marsh and Trenham 2001) covering the entire hydroperiod gradient (Snodgrass et al. 2000b; Hocking et al. 2008). Research should be undertaken to determine whether and at what level managing clusters of ponds may contribute more to community diversity and stability than managing single ponds. There is minimal research on the value or necessity of SMZs as movement corridors. A significant question is whether special provisions for connectivity are even necessary in managed pine landscapes. Research should explore the potential role of SMZs as movement and dispersal corridors for large mammals, herpetofauna, bats, and birds. Forest land owners would benefit from efforts to model biodiversity at the landscape level to assist with land use planning. Multi-scale analysis would allow creation of landscape-level models and subsequent adaptive management.

Conclusions

Forest certification programs provide impetus to develop effective and sustainable management strategies to conserve wetland-associated wildlife communities and protect ecosystem integrity. Although there is abundant survey data, additional work needs to be done to thoroughly document the potential contributions of retained wetland features to vertebrate biodiversity in managed pine forests of the southern USA. Successful conservation of wetland-associated wildlife communities in managed pine systems will require attention to questions of community ecology and management effects. The current state of knowledge needs to be improved through rigorous, long-term experiments that target management impacts on community composition and fitness as well as measures of richness and abundance. Previous studies have provided hypotheses that should be tested using more powerful methods. Such research will increase opportunities for forest land owners to incorporate and sustain the biodiversity contributions of retained wetland features.

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