Chapter 11: Reptile and Amphibian Response to Hardwood Forest Management and Early Successional Habitats

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Abstract Herpetofauna responses to forest management and early successional habitats are influenced by species-specific adaptations to historical disturbance regimes. It can take decades for woodland salamander diversity to recover after heavy overstory removal for even-aged forest regeneration or hot fires that yield higher light, drier microclimates, and reduced leaf litter cover, but some frog and toad species may tolerate or even increase after disturbances. In particular, disturbances that retain some canopy cover, such as selection harvests or low intensity burns, can mitigate effects on terrestrial salamanders. The same early successional conditions that are detrimental to salamanders can benefit many reptile species, such as fence lizards (*Sceloporus undulatus*). Maintaining stand age diversity across central hardwood forest landscapes, including retention of mature forest communities, should provide habitats for both early successional wildlife and mature forest species.

11.1 Relevant environmental changes following disturbance

The extent and frequency of historical disturbances in central hardwood forests varied widely depending on slope position, aspect, stand age, and stand composition (White et al., Chapter 3). Gap phase disturbances following wind events, ice storms, and insect outbreaks were more common than the large-scale changes that followed hurricanes and wildfires in other regions of North America (White et al., Chapter 3). Amphibian and reptile species associated with mature hardwood forest presumably were common across much of the landscape, whereas those associated with early succession habitats were much more variable because they depended upon infrequent natural disturbance to create ephemeral patches of suitable habitat (Greenberg 2001).

Natural and anthropogenic disturbances that create young forest by removing or reducing canopy cover can greatly alter the microclimate at or just below the soil surface, where most amphibian and reptile species reside (but see Brooks and Kyker-Snowman 2008). Following overstory removal, light penetration increases, raising soil temperatures and evaporation rates and decreasing litter depth and moisture until it is replenished by leaf-fall and shade from the recovering vegetation (Greenberg et al., Chapter 8). Fire also can consume leaf litter and reduce

leaf-fall input levels (Petranka et al. 1994). Increased light levels near the ground promote development of a grass and forb layer and establishment of shrubs or regenerating trees (Russell et al. 2004). These environmental changes can alter herpetofaunal movement patterns, survival rates, and prey abundance (Moseley et al. 2004).

Down wood or coarse woody debris (CWD) is used by many reptile and amphibian species for mating sites, egg-laying, feeding, and thermoregulation (Whiles and Grubaugh 1996). Down wood volume typically follows a U-shaped chrono-sequence in central hardwood forests, with highest levels in the 5 to 10 years following disturbance (i.e., downed trees following windthrow or logging slash following timber harvest) and again during late-succession or old growth stages when aging trees senesce (Gore and Patterson 1986). However, larger, more decayed logs may be more abundant in mature or old growth hardwood forest (Petranka et al. 1994). Webster and Jenkens (2005) reported that primary forests in the Southern Appalachians contained more large-diameter, highly decayed CWD compared to forests subjected to anthropogenic disturbances. Furthermore, among sites with similar disturbance histories, higher levels of CWD were associated with mesic conditions and higher elevations (Webster and Jenkens 2005, Keyser, Chapter 15). Therefore, reptile and amphibian species that use down wood heavily may be most abundant early (e.g., some reptiles) or late (e.g., salamanders) in stand development. However, the degree to which salamanders and other amphibians specifically rely on CWD is likely influenced by the availability of other surface cover. For example, salamanders may use cover objects less in undisturbed stands with intact leaf litter and vegetation cover compared to stands where leaf litter and vegetative cover is reduced from prescribed burning and herbivory (Ford et al. 2010).

Machinery associated with timber harvest operations can cause soil compaction or erosion. Disturbances of the subterranean environment, as occurs with most types of mechanical site preparation, can cause direct mortality or degrade habitat conditions for fossorial snakes and other species that spend portions of their life cycle below ground (Russell et al. 2004, Todd and Andrews 2008). However, mechanical site preparation and other forms of intensive forest management are uncommon in the Central Hardwood Region as compared to other regions such as the southeastern Coastal Plain (e.g., Russell et al. 2002).

Amphibians and reptiles often are generically lumped together as "herpetofauna," but in fact are as phylogenetically distinct from one another as are mammals and birds. Amphibians (class *Amphibia*) have permeable, moist skin that is used for respiration and increases vulnerability to desiccation. Amphibians have a two-stage or "biphasic" life cycle that includes morphologically distinct larval and adult stages. Most require water for egg deposition and development of larvae, which eventually metamorphose into adults that can be largely terrestrial (Duellman and Trueb 1986). Amphibian taxa vary considerably in their vulnerability to desiccation. For example, some frogs and toads can tolerate higher temperatures (Stebbins and Cohen 1995) and can store and reabsorb larger amounts of water in their bladders than salamanders (Zug 1993). Some salamanders are lungless, and some are completely terrestrial (deMaynadier and Hunter 1995). Many amphibian species have small home ranges (Duellman and Trueb 1986) and poor dispersal capabilities (Sinsch 1990). Conversely, most reptiles (class *Reptilia*) require warm temperatures (associated with higher light levels) for egg incubation and successful development of hatchlings (Deeming and Ferguson 1991). Reptiles have dry scaly skin that protects them from desiccation. Clearly, response to disturbance and early successional habitats might be expected to differ between the two taxonomic classes, and among species within them. Within *Amphibia*, salamanders tend to decline following disturbances that reduce

canopy cover because of their increased risk of desiccation, whereas some toad and frog species may tolerate higher temperatures and lower moisture in early successional habitats (Russell et al. 2004). Many reptile species increase in recently disturbed areas, likely because of improved opportunity for thermoregulation and foraging (Russell et al. 2004).

11.2 Amphibian and reptile response to timber harvest
11.2.1 Amphibian response

Heavy overstory removal for forest regeneration treatments (e.g., clearcut or shelterwood regeneration harvests) can adversely affect amphibians, especially terrestrial salamanders (Pough et al. 1987, Petranka et al. 1993, 1994, deMaynadier and Hunter 1995, Ash 1997, Harpole and Haas 1999, Reichenbach and Sattler 2007). Canopy removal results in higher light levels, a warmer, drier microclimate, and reduced leaf litter cover, which could cause salamanders to desiccate (deMaynadier and Hunter 1995, Renken 2006). In the Southern Appalachians, terrestrial salamander abundance declines following clearcutting (Ash 1988, Ash 1997, Petranka et al. 1993, Petranka et al. 1994; but see Adams et al. 1996).

There has been considerable debate about the time that it takes salamander populations to recover to pre-disturbance levels following canopy removal (Ash and Pollock 1999, Petranka 1999). Estimates range from approximately 20 years to more than 100 years (Table 11.1). Discrepancies in documented recovery periods likely are related to differences in study designs, salamander communities, and site and landscape characteristics. But, research suggests that post-disturbance recovery of salamander abundance is closely correlated with litter layer recovery (Pough et al. 1987, Ash 1997, Crawford and Semlitsch 2008a). Longer recovery periods may be required on drier aspects and ridge tops than on mesic sites where soil moisture remains relatively high even after disturbance (Harper and Guynn 1999, Petranka 1999). However, the former sites generally are poorer sites for woodland salamanders. Disturbances that retain heavy canopy cover such as midstory removal, selection harvest, firewood cutting, thinning, and heavy browsing by white-tailed deer (*Odocoileus virginianus*) are less likely to affect salamander abundance (Pough et al. 1987, Adams et al. 1996, Messere and Ducey 1998, Brooks 1999, Ford et al. 2000, Harpole and Haas 1999, Moseley et al. 2003, Knapp et al. 2003, Homyack and Haas 2009, Semlitsch et al. 2009). Yet, salamander density may decline following partial canopy reduction (e.g., Duguay and Wood 2002), and reductions in canopy cover by as little as 41% can cause local declines in salamander abundance (Knapp et al. 2003).

The exact mechanisms for the disappearance of terrestrial salamanders from disturbed sites remain in question. Semlitsch et al. (2008) proposed three hypotheses to explain amphibian declines following timber harvest: (1) retreat to underground refugia; (2) mortality from desiccation or starvation; and (3) evacuation to adjacent forest. Although a percentage of pond-breeding mole salamanders (*Ambystoma* spp.) may disperse out of disturbed environments, it is not known how they fare once they reach adjacent forest (Semlitsch et al. 2008). Mortality is the most likely cause of declines in terrestrial salamander density following clearcutting because plethodontid salamanders primarily are surface feeders and individuals eventually would starve unless they came to the surface where they could desiccate. Adult plethodontid salamanders lack lungs and depend on cutaneous respiration for gas exchange. Because moist skin is necessary to facilitate respiration, salamanders are most active where the forest floor is moist or at night when relative humidity is highest (Petranka et al. 1993). Salamander desiccation results from reduced leaf litter cover and depth, and higher ground temperatures following clearcutting, rather than changes in soil moisture (Pough et al. 1987, Ash 1997, Rothermel and Luhring 2005).

Rothermel and Luhring (2005) showed that salamander survival was 100% in uncut forest, but individuals could survive in clearcuts only by gaining access to protective underground burrows. Some researchers have speculated that salamanders are unlikely to evacuate to adjacent forested areas that already are saturated with territorial adults (e.g., Petranka 1999). For example, Bartman et al. (2001) did not detect any post-harvest emigration of plethodontid salamanders from sites that had been subjected to shelterwood harvests in western North Carolina. Interestingly, Ash (1997) speculated that adult salamanders disperse into early successional habitats such as clearcuts to avoid competition from smaller or immature salamanders that are restricted to mature forests with abundant, moist litter.

Juvenile frogs and salamanders typically exhibit higher rates of mortality than adults following canopy removal because their high surface:volume ratios make them prone to desiccation (Jaeger 1980, Ash et al. 2003, Marsh and Goicochea 2003). Additionally, the high adult:juvenile ratio of salamanders in clearcuts indicates low reproduction by adults or higher rates of mortality in juveniles (Ash 1997, Ash et al. 2003). Adults of some salamander species are better adapted to withstand the hot, dry conditions of recently disturbed sites or more exposed ridge top environments (Ash 1997, Ash et al. 2003, Ford et al. 2010). For example, Ford et al. (2010) reported that larger-bodied slimy salamanders (*Plethodon glutinosis*) were less affected by leaf litter reduction following fire than smaller-bodied red-backed salamanders (*P. cinereus*) or mountain dusky salamanders (*Desmognathus ocropheaus*). Riedel et al. (2008) documented high densities of both adult and juvenile eastern red-backed salamanders within former deciduous forests of West Virginia that had been converted to silvopastures, traditional pastures, and ungrazed meadows, indicating that this species may be more resilient to the creation of early successional habitats than previously thought. Interestingly, the physiological condition and sex ratios of salamanders within these open, early successional habitats were similar to those of salamanders in adjacent mature forest, although adults were significantly more abundant than juveniles (Riedel 2006). Riedel et al. (2008) suggested that the presence of artificial cover in these open, early successional habitats, in combination with moisture trapped by dense herbaceous vegetation, facilitated woodland salamander persistence. In addition, Marsh et al. (2004) showed that dispersal of *P. cinereus* was not limited by the presence of forest cover, and suggested that this species may be relatively insensitive to the creation of small, intervening, open habitats within deciduous forests such as fields, power line corridors, and even small residential areas. Accordingly, at least some species of woodland salamanders may tolerate the creation of small patches of early successional habitats within mature deciduous forests (Marsh et al. 2004, Riedel et al. 2008, Moseley et al. 2009), yet others can be highly sensitive to forest road edges (Semlitsch et al. 2007). However, individuals forced to forage in areas with reduced cover may be more exposed to predation (Moseley et al. 2004).

Timber harvest can affect stream-breeding salamanders by eliminating terrestrial habitat for adults and by degrading aquatic habitats required for larval development (Perkins and Hunter 2006, Crawford and Semlitsch 2008a, Peterman and Semlitsch 2009). Adult stream-breeding salamanders (e.g., *Desmognathus* and *Eurycea*) use terrestrial habitats some distance away from streams for foraging and overwintering habitat (Ashton and Ashton 1978, Crawford and Semlitsch 2007). Similar to terrestrial salamanders, adult stream salamander (e.g., Blue Ridge two-lined salamander [*E. wilderae*]) abundance may be reduced following timber harvest because of decrease in leaf litter depth, soil moisture, and overstory cover (Crawford and Semlitsch 2008a, b). Increased water temperatures and reduced litter input following canopy removal and sedimentation from logging roads (Vose and Ford Chapter 14) are detrimental to

larval salamanders that occur in the streams (Semlitsch 2000, Peterman and Semlitsch 2009). Stream sedimentation can fill interstitial spaces between rocks at the stream bottom, thus potentially reducing abundance of salamanders that use the spaces for cover, such as *Eurycea* and *Desmognathus* species (Lowe and Bolger 2002, Miller et al. 2007, Moseley et al. 2008, Peterman and Semlitsch 2009). However, retention of an uncut riparian buffer may mitigate the effects of clearcut harvests on larval salamanders (Peterman and Semlitsch 2009).

Frogs and toads tend to be more tolerant of canopy removal and elevated ground temperatures than salamanders (Gibbs 1998, Ross et al. 2000, Russell et al. 2004, Patrick et al. 2006). Additionally, tadpoles of some frog species may develop faster or survive better in ponds within clearcuts (Semlitsch et al. 2009, Felix et al. 2010). Some anuran species likely are attracted to the higher coverage of herbaceous vegetation around ponds in open environments (Felix et al. 2010). Response to canopy removal around breeding ponds differs among anuran species. Species associated with open habitats, such as gray treefrogs (*Hyla versicolor*), deposit more eggs in ponds in areas with heavy canopy removal. In contrast, species that require cooler water temperatures, such as mountain chorus frogs (*Pseudacris brachyphona*) and spotted salamanders (*Ambystoma maculatum*) only deposit eggs where at least 75% of the canopy is retained (Semlitsch et al. 2009, Felix et al. 2010). However, gray treefrogs oviposited more eggs in ponds in clearcuts close to forest edge than in ponds 50 m into clearcuts (Hocking and Semlitsch 2007), because adult treefrogs require mature trees for foraging (Johnson et al. 2007, 2008). Adult wood frogs (*Rana sylvatica*) were able to travel through clearcuts when dispersing between breeding ponds and non-breeding habitats in mature forest, but their rate of travel increased in response to the degraded micro-climatic conditions (Rittenhouse and Semlitsch 2009). Some anurans, especially juveniles, may experience increased predation or desiccation risks following timber harvests (Patrick et al. 2006, Rittenhouse and Semlitsch 2009, Rittenhouse et al. 2009). Species response to the creation of young forest may vary regionally. For example, adult wood frogs did not use hot, dry clearcuts in Missouri but did use moist areas within clearcuts as non-breeding habitat in Maine (Patrick et al. 2006, Rittenhouse and Semlitsch 2009).

11.2.2 Reptile response

The same conditions following timber harvest that may be detrimental to amphibians appear to benefit many reptiles (Greenberg 2002, Adams et al. 1996). Most reptile species require the warm temperatures associated with higher light levels for egg incubation and successful development of hatchlings (Goin and Goin 1971, Deeming and Ferguson 1991). The hotter, drier microclimate in open, disturbed sites also may facilitate movement and thermoregulation for many reptile species (Greenberg 2001). Lizards, particularly fence lizards (*Sceloporus undulatus*), generally increase following canopy reduction (McLeod and Gates 1998, Greenberg 2001, Renken et al. 2004). Following timber harvests, Renken et al. (2004) determined that juvenile abundance of *S. undulatus* was twice as high as that of adults, suggesting that the lizards experienced an immediate boost in reproductive rates in disturbed sites or that the recently disturbed sites were colonized primarily by juveniles. In predominantly forested landscapes in Pennsylvania, snake abundance and richness increased with decreasing tree basal area (Ross et al. 2000).

However, there is evidence that some forest-dwelling reptile species may decline following timber harvest (Russell et al. 2004). In Coastal Plain pine forests, abundance of several small-bodied leaf litter snake species was lower in clearcuts than unharvested and

thinned pine stands, but snake abundance was highest in thinned stands where habitat heterogeneity and presumably prey abundance was highest (Todd and Andrews 2008). In contrast with the management of deciduous forests, the intensive mechanical site preparation associated with Coastal Plain pine management not only removes surface cover used by small-bodied snakes but also likely results in direct destruction of nest sites (Russell et al. 2002).

11.3 Response to prescribed fire

Over the past 500 years, fire was a common forest disturbance across much of the Central Hardwood Region (Spetich et al. Chapter 4). Fire effects on vegetation structure likely varied with fire intensity and frequency, which in turn was influenced by topography, weather conditions, and population distribution of Native Americans or European settlers who intentionally burned to promote forage for game or livestock (Spetich et al. Chapter 4). Hot fires certainly reduced leaf litter and often killed overstory trees, creating patchy, heterogeneous early successional conditions with some snags and trees remaining. In contrast, cool, patchy burns likely had minimal impact on overstory trees or leaf litter depth and cover, but reduced shrub cover or killed midstory trees where it burned. In ecosystems such as longleaf pine-wiregrass or sand pine-scrub where lightning-ignited fires created and maintained "fire climax" habitat conditions, many species of reptiles and amphibians are behaviorally adapted to survive wildfire or prescribed burns, and require fire maintained habitat conditions (Russell et al. 1999; Greenberg 2002). Less is known about fire effects on herpetofauna of upland hardwood forest, where the majority of fires were historically human-caused. Fire is thought to have little direct effect on amphibians and reptiles, but the likelihood of individual mortality during a fire depends on the species' behavior, fire intensity, and season of burn (Russell et al. 1999). Negative indirect impacts of prescribed fire likely are most severe for species that require leaf litter or other forest debris that is consumed (Russell et al. 1999).

Relatively few studies have addressed fire effects on herpetofauna in hardwood forests (Russell et al. 2004, Renken 2006). Several studies have reported no difference between amphibian populations on prescribed burned sites and unburned controls (Ford et al. 1999, Floyd et al. 2001, Moseley et al. 2003, Keyser et al. 2004, Greenberg and Waldrop 2008, Ford et al. 2010, Matthews et al. 2010). Others have indicated that toad abundance may increase following fire (Kirkland et al. 1996, Greenberg and Waldrop 2008). Conversely, intense prescribed fires that cause immediate or delayed reduction in canopy cover following overstory tree mortality can produce micro-habitat changes near the forest floor (e.g., reduced leaf litter cover and depth, more sunlight, higher ground temperatures) that negatively impact salamander populations (Matthews et al. 2010).

Reptiles, lizards in particular, may increase after prescribed burns, especially after hot fires that reduce canopy cover (Moseley et al. 2003, Keyser et al. 2004; Greenberg and Waldrop 2008, Matthews et al. 2010). Litter removal, midstory and canopy reduction, and higher ground temperatures following intense fires likely create thermoregulatory conditions favorable for lizards (Moseley et al. 2003). Overstory mortality following intense fires also generates down wood that may be used as basking sites by lizards and large-bodied snakes (Matthews et al. 2010). However, it is not known whether these same changes negatively affect small-bodied fossorial snakes that depend on leaf litter.

Prescribed fire effects on wetland and stream-associated amphibians in central hardwood forests have not been well studied (Renken 2006). Intense fires that kill trees and reduce canopy

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cover in the uplands adjacent to streams or amphibian breeding ponds could result in higher water temperatures increased sedimentation rates, or runoff of ash that changes water pH, potentially killing amphibian adults, eggs, or larvae (Renken 2006). However, other temperature and sediment-sensitive aquatic vertebrates in the Appalachians, such as brook trout (*Salvelinus fontinalis*), have been reported to respond positively to adjacent forest disturbances, presumably in response to greater abundance of macroinvertebrate prey after partial canopy removal (Nislow and Lowe 2006). In short, more research is needed on the effects of fire and other forest disturbances on aquatic and riparian reptiles and amphibians in central hardwood forests.

11.4 Mitigation strategies

11.4.1 Stream and wetland buffers

Riparian buffers between upland timber harvests and adjacent streams or wetlands have been recommended to mitigate impacts on sensitive amphibian species (Semlitsch 2000). Buffers shade water, contribute leaf litter to streams, filter sediment, provide terrestrial habitats for biphasic amphibians and reptiles, and possibly provide refuge for individuals dispersing out of harvested areas (Mitchell et al. 1997, Semlitsch 2000, Perkins and Hunter 2006). Crawford and Semlitsch (2007) recommended a 92-m buffer adjacent to Southern Appalachian streams to provide core habitat free of edge effects for the widest ranging stream salamander species. Effects of timber harvest on sensitive amphibian species may extend at least 25 m into adjacent mature forest, possibly because of the reduced canopy and litter cover along edges created by timber harvests (deMaynadier and Hunter 1998). To provide the core biphasic habitat needs, Semlitsch and Bodie (2003) recommended 159-290 m buffers for amphibians and 127-289 m buffers for reptiles around wetlands and streams. However, it has been speculated that narrower 30-m buffers may provide adequate protection to larval salamanders (Peterman and Semlitsch 2009). Alternatively, a two-tiered approach has been recommended to protect aquatic herpetofauna, with unharvested 10-25 m buffers around streams surrounded by a wider partial harvest zone (deMaynadier and Hunter 1995). To date, however, the actual community and demographic responses of stream-dwelling herpetofauna to adjacent forest disturbance remain poorly characterized. Therefore, few data are available to evaluate the efficacy of specific buffer widths recommended to protect herpetofauna within deciduous forests of the Central Hardwood Region.

11.4.2 Coarse woody debris retention

Salamander populations are positively linked to CWD abundance, especially on drier sites and where leaf litter cover is sparse, so retention of CWD may help mitigate the effects of disturbance on amphibians and provide critical habitat or refuge to a number of reptile species (Pough et al. 1987, Petranka et al. 1994, Brooks 1999, Herbeck and Larsen 1999, Russell et al. 2004). Retention of CWD and brush piles in clearcuts may decrease the proportion of salamanders leaving clearcuts and could contribute to increased juvenile amphibian survival by providing cool, moist refugia (Patrick et al. 2006, Rittenhouse et al. 2008, Semlitsch et al. 2008). Todd and Andrews (2008) captured more small snakes in clearcuts with CWD retention than in clearcuts without retention. However, CWD retention appears to provide only short-term benefits to sensitive amphibians by providing refuge from desiccating conditions immediately

post-harvest, and may not prevent declines (Mosely et al. 2004, Semlitsch et al. 2009). Coarse woody debris diameter and degree of decay is generally much lower, and thus not used by salamanders, in recently harvested sites than in old growth stands (Herbeck and Larsen 1999). Additionally, several studies failed to show benefits of CWD retention for amphibians (Greenberg 2001, Ford et al. 2002a, Rothermel and Luhring 2005, Rittenhouse and Semlitsch 2009). Similarly, higher abundance of lizards and snakes in small canopy gaps was not related to CWD abundance (Greenberg 2001).

11.4.3 Overstory retention

Small forest openings such as group selection harvests and wind-created downburst gaps with multiple treefalls, or partial harvests that retain a large percentage of the overstory, can mitigate the negative effects of timber harvest on amphibians by maintaining shade and leaf litter input and providing refuge and recolonization sources (Pough et al. 1987, Ford et al. 2000, Greenberg 2001, Lowe and Bolger 2002, Homyack and Haas 2009). Overstory retention adjacent to wetlands can be critical to maintaining connectivity between aquatic reproduction sites and other habitat features required by amphibians, as many, especially salamanders, avoid timber harvests when emigrating from breeding pools (Todd et al. 2009). In Maine, partial harvests adjacent to headwater streams had less effect on amphibian communities than clearcuts (Perkins and Hunter 2006). Increased growth of herbaceous plants or shrubs near the forest floor following small overstory reductions might improve habitat conditions for some herpetofaunal species and mitigate changes to the microclimate that are problematic for disturbance-sensitive species such as salamanders (Ross et al. 2000, Semlitsch et al. 2009). Retention of at least 50% of the overstory is recommended to minimize negative effects on amphibian populations (Ross et al. 2000, Semlitsch et al. 2009). However, as little as 41% reduction in the overstory may result in declines in the abundance of plethodontid woodland salamanders similar to clearcuts (Knapp et al. 2003). Group-selection harvests require more frequent stand entries across a larger land base to extract the same amount of wood fiber as a clearcut (Homyack and Haas 2009). We suggest that the relationships between partial overstory reduction and response by amphibian populations require more study.

11.4.4 Small stand sizes and longer rotations

Smaller harvest units may help to minimize the deleterious effects of timber harvest on wood frogs and other sensitive amphibians, especially juveniles (Patrick et al. 2006, Rittenhouse and Semlitsch 2009). The distance that dispersing individuals must traverse across smaller clearcuts could lessen the risks of desiccation and predation. Additionally, small timber harvests may facilitate evacuation by individuals from harvested areas into adjacent uncut areas (Semlitsch et al. 2008). Ford et al. (2002a) demonstrated that the amount of cove hardwood habitat surrounding harvested patches is an important determinant of woodland salamander population response to the disturbance, so designated no-harvest areas on the landscape could serve as sources for repopulating nearby harvest units (Petranka et al. 2004). Additionally, breeding pools in small timber harvest openings could provide ideal locations for rapid larval development for larvae of some disturbance-adapted or early successional amphibians and be in close proximity to the mature forest required by adults (Barry et al. 2008, Semlitsch et al. 2009). Further, small harvest openings (< 2 ha) provide habitat for lizards and other reptiles (Greenberg

2001). Similar to group-selection harvests, however, harvest of the same timber volume in smaller units requires more roads, potentially leading to sediment loading in streams and disturbance to a larger percentage of the land base.

Increasing the rotation length of managed forest stands would ensure that a portion of the landscape contained large trees, high accumulations of large diameter CWD, and other structural characteristics associated with late-seral forest (Herbeck and Larsen 1999). Alternatively, employing forest management practices that retain and enhance structural components of habitats important for herpetofauna (e.g., retention of CWD, green and legacy tree retention, selection harvest systems) may provide suitable conditions for these species while contributing to economic and other resource objectives. Additionally, management practices that mimic historical disturbance regimes may be used to promote a diversity of cover types across the landscape, which in turn would provide habitat for a variety of reptiles and amphibians. Examples of historical disturbance conditions include more frequent prescribed fires on xeric ridge tops in the Southern Appalachians and less disturbance on moist, north-facing slopes and ravines.

11.5 Research challenges

More focus on reptile response to disturbance. Reptile response to disturbance from forest management has been studied much less than amphibian response. For example, a database search of journal articles using the keywords *salamander* and *clearcut* generated 64 citations; conversely, a search using the same database with the keyword *lizard* in place of *salamander* generated three citations and replacement of *salamander* with *snake* generated one citation. We can only speculate that the cause for the discrepancy is due in large part to the direction of response by amphibians and reptiles in previous studies. Because amphibians, especially woodland plethodontid salamanders, typically decline locally following disturbance, they have received the majority of research emphasis in the past two decades. However, some reptile species such as small fossorial snakes similarly show negative response; other reptile species, such as fence lizards increase in abundance following disturbance. We suggest that there may be a bias in the scientific literature attributable to a greater attraction by scientists to studying taxa that respond negatively to forest management, and journals to accept manuscripts that report significant results.

Longer study durations. Deleterious effects of canopy reduction on salamanders and other amphibians may be delayed for up to five years after timber harvest (Ash 1988, Reichenbach and Sattler 2007, Homyack and Haas 2009). Some species may experience a greater time lag in the demographic changes that occur following disturbance (Homyack and Haas 2009). Greenberg and Waldrop (2008) reported that a single prescribed burn that killed trees and reduced canopy cover did not reduce the relative abundance of terrestrial salamanders (*Plethodon* spp.), but salamander abundance was lower in the same treatment units compared to control plots after a second burn 5 years later in the same study area (Matthews et al. 2010). The delayed changes in salamander abundance following the fuel reduction treatments could either have been a result of additive effects of the treatments on environmental conditions, or the result of delayed changes in demographic parameters (Matthews et al. 2010). Lastly, long-term studies also should address the effects of forest management on population demography at large spatial scales (Homyack and Haas 2009).

More accurate assessment of detection bias. Most reptile and amphibian studies assume that sampled individuals represent the entire population (deMaynadier and Hunter 1995). This assumption is unlikely for salamanders because surface populations represent only a small percentage of the total population (Bailey et al. 2004a). Additionally, detection probabilities often differ among treatment areas because of variable habitat conditions, which in turn could influence abundance estimates for reptile and amphibian populations (Bailey et al. 2004b). For example, reduction of leaf litter from prescribed fire or timber harvest could cause individual salamanders to move more frequently and for longer periods (Mosely et al. 2004), or cause them to aggregate under coverboards being used to assess population response to burning or other disturbances (Ford et al. 2010). Few studies of reptile and amphibian response to forest management have accounted for detection bias (except see Bailey et al. 2004b, Ford et al. 2010). Mark-recapture methodologies can be used to account for detection probability, but recapture rates, especially with terrestrial salamanders, can be low and capture-recapture methods can be costly when used in large-scale field experiments (Bailey et al. 2004a). In the case of large-scale studies, researchers can use a double-sampling design that uses capture-recapture analysis on a subset of sites to estimate detection probability and calibrate counts for the complete set of sampling locations (Bailey et al. 2004a, Bailey et al. 2004c).

More focus on site conditions, landscape position, and abiotic features. There is evidence that elevation, slope, concavity, and other landform characteristics may be important determinants of woodland salamander occurrence and abundance in central hardwood forests (Ford et al. 2002a, b). Many studies of amphibian response to forest management do not account for landscape position and associated conditions such as moist, concave, lower-slope positions with a thicker leaf litter layer and drier, warmer ridge tops or south-facing slopes that could influence amphibian or reptile species composition and their response to disturbance. When compared to other vertebrates, patterns of amphibian distribution across landscape scales remain poorly known (Johnson et al. 2002, Dillard et al. 2008a). Because amphibians have limited dispersal abilities and small home ranges, site-specific habitat factors often are assumed to have an overriding influence on patterns of amphibian distribution. However, there is increasing evidence that abiotic habitat characteristics measured at broad spatial scales are important predictors of amphibian occurrence and abundance within forest ecosystems. Although disturbance and succession of vegetation exert a strong influence on amphibian distribution and abundance (deMaynadier and Hunter 1995, Russell et al. 2004), recent research indicates that the importance of abiotic habitat features such as geology, topography, and climate have not been sufficiently recognized (Russell et al. 2005, Harper 2007, Dillard et al. 2008a, b). For example, Dillard et al. (2008a, b) showed that elevation, slope, aspect, and parent geology were better predictors of the occurrence of the threatened Cheat Mountain salamander (*P. nettingi*) in deciduous forests of West Virginia than were the composition or successional stage of overstory vegetation. Moseley et al. (2009) determined that the effects of canopy openings (e.g., edge effects) on woodland salamanders within deciduous forests of West Virginia depended on site aspect.

Landscape-level population effects. Most studies of amphibian and reptile response to forest management have been conducted at the scale of an individual stream, forest stand, or wetland. Therefore, more research is needed to assess the persistence of reptile and amphibian

communities at the landscape or watershed scale (Perkins and Hunter 2006). Renken et al. (2004) recorded similar responses by reptiles and amphibians to clearcuts as in other studies, but the researchers failed to detect larger-scale impacts given the relatively small percentage of the landscape that was harvested. Ford et al. (2002a) suggested that salamander populations in small, isolated cove hardwood stands might be more vulnerable to extirpation by timber harvests than populations in larger, less isolated coves. Because juvenile amphibians are more susceptible to habitat change, management activities that fragment habitats likely will have the greatest impact on species for which juveniles conduct the majority of dispersal among breeding and non-breeding locations (Patrick et al. 2008). Some amphibian species avoid roads likely because of reduced soil moisture and cover, so landscape-level conservation strategies should account for these increasingly prominent movement barriers (Gibbs 1998, Marsh and Beckman 2004, Semlitsch et al. 2007). In contrast, anecdotal evidence indicates that secondary forest roads and trails with little use may not have negative impacts on herpetofauna and in some cases be used as habitat (e.g., Dillard et al. 2008c). More information is needed to better understand how landscape factors influence amphibian and reptile response to the creation of early successional habitats in upland hardwood forest (Ford et al. 2002a).

11.6 Literature cited

Adams JP, Lacki MJ, Baker MD (1996) Response of herpetofauna to silvicultural prescriptions in the Daniel Boone National Forest, Kentucky. Proc Annu Conf Southeast Assoc Fish and Wildl Agencies 50:312-320

Ash AN (1988) Disappearance of salamanders from clearcut plots. J Elisha Mitchell Sci Soc 104:116-122

Ash AN (1997) Disappearance and return of plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Con Bio 11:983-989

Ash AN, Bruce RC, Castanet J, Francillon-Vieillot H (2003) Population parameters of *Plethodon metcalfi* on a 10-year-old clearcut and in nearby forest in the southern Blue Ridge Mountains. J Herp 37:445-452

Ash AN, Pollock KH (1999) Clearcuts, salamanders, and field studies. Con Bio 13:206-208

Ashton RE, Ashton PS (1978) Movements and winter behavior of *Eurycea bislineata* (Amphibia, Urodela, Plethodontidae). J Herp 12:295-298

Bailey LL, Simons TR, Pollock KH (2004a) Estimating detection probability parameters for Plethodon salamanders using the robust capture-recapture design. J Wildl Manage 68:1-13

Bailey LL, Simons TR, Pollock KH (2004b) Spatial and temporal variation in detection probability of Plethodon salamanders using the robust capture-recapture design. J Wildl Manage 68:14-24

Bailey LL, Simons TR, Pollock KH (2004c) Estimating site occupancy and species detection probability parameters for terrestrial salamanders. Ecol Applic 14:692-702

Barry DS, Pauley TK, Maerz JC (2008) Amphibian use of man-made pools on clearcuts in the Allegheny Mountains of West Virginia, USA. Appl Herp 5:121-128

Bartman CE, Parker KC, Laerm J, McCay TS (2001) Short-term response of Jordan's salamander to a shelterwood timber harvest in western North Carolina. Phys Geog 22:154-166

Brooks RT (1999) Residual effects of thinning and high white-tailed deer densities on northern redback salamanders in southern New England oak forests. J Wildl Manage 63:1172-1180

Brooks RT, Kyker-Snowman TD (2008) Forest floor temperature and relative humidity following timber harvesting in southern New England, USA. For Ecol Manage 254:65-73

Crawford JA, Semlitsch RD (2007) Estimation of core terrestrial habitat for stream-breeding salamanders and delineaton of riparian buffers for protection of biodiversity. Con Bio 21:152-158

Crawford JA, Semlitsch RD (2008a) Post-disturbance effects of even-aged timber harvest on stream salamanders in southern Appalachian forests. Ani Con doi:10.1111/j.1469-1795.2008.00191.x

Crawford JA, Semlitsch RD (2008b) Abiotic factors influencing abundance and microhabitat use of stream salamanders in southern Appalachian forests. For Ecol Manage 255:1841-1847

Deeming DC, Ferguson MWJ (1991) Physiological effects of incubation temperature on embryonic development in reptiles and birds. In: Deeming DC, Ferguson MWJ (eds) Egg Incubation: Its Effects on Embryonic Development in Birds and Reptiles. Cambridge University Press, New York

deMaynadier PG, Hunter ML Jr (1995) The relationship between forest management and amphibian ecology: a review of the North American literature. Environmental Reviews 3:230-261

deMaynadier PG, Hunter ML Jr (1998) Effects of silvicultural edges on distribution and abundance of amphibians in Maine. Con Bio 12:340-352

Dillard LO, Russell KR, Ford WM (2008a) Macrohabitat models of occurrence for the threatened Cheat Mountain Salamander, *Plethodon nettingi*. Appl Herp 5:201-224

Dillard LO, Russell KR, Ford WM (2008b) Site level habitat models for the endemic, threatened Cheat Mountain Salamander: the importance of geophysical and biotic attributes for predicting occurrence. Biodiv and Con 17:1475-1492

Dillard, LO, Russell, KR, Ford WM (2008c). *Plethodon nettingi* habitat use and ovoposition. Herp Rev 39(3):335-336

Duguay JP, Wood PB (2002) Salamander abundance in regenerating forest stands on the Monongahela National Forest, West Virginia. For Sci 48:331-335

Duellman WE, Trueb L (1986) Biology of Amphibians. McGraw-Hill, New York

Felix ZI, Wang Y, Schweitzer CJ (2010) Effects of experimental canopy manipulation on amphibian egg deposition. J Wild Manage 74:496-503

Floyd TM, Russell KR, Moorman CE, Van Lear DH, Guynn, DC Jr, Lanham JD (2001) Effects of prescribed fire on herpetofauna within hardwood forests of the upper Piedmont of South Carolina: a preliminary analysis. In: Outcalt, KW (ed), Proc Eleventh Biennial Southern Silvicultural Research Conference. Gen. Tech. Rep SRS-48. USDA Forest Service Southern Research Station, Asheville, NC

Ford WM, Chapman BR, Menzel MA, Odom RH (2002a) Stand age and habitat influences on salamanders in Appalachian cove hardwood forests. For Ecol Manage 155:131-141.

Ford WM, Menzel MA, Odom RH (2002b) Elevation, aspect and cove size effects on woodland salamanders in the southern Appalachians. Southeast Nat 1:315-324

Ford WM, Menzel MA, McGill DW, Laerm J, McCay TS (1999) Effects of a community restoration fire on small mammals and herpetofauna in the southern Appalachians. For Ecol Manage 114:233-243

Ford WM, Menzel MA, McCay TS, Gassett JW, Laerm J (2000) Woodland salamander and small mammal responses to alternative silvicultural practices in the southern Appalachians of North Carolina. Proc Annu Conf Southeast Assoc Fish and Wildl Agencies 54:241-250

Ford WM, Rodrigue JL, Rowan EL, Castleberry SB, Schuler TM (2010) Woodland salamander response to two prescribed fires in the central Appalachians. For Ecol Manage 260:1003-1009

Gibbs JP (1998) Amphibian movements in response to forest edges, roads, and streambeds in southern New England. J Wild Manage 62:584-589

Goin CJ, Goin OB (1971) Introduction to Herpetology. W.H. Freeman and Company, San Francisco

Gore JA, Patterson WA III (1986) Mass of downed wood in northern hardwood forests in New Hampshire: potential effects of forest management. Can J For Res 16:335-339

Greenberg CH (2001) Response of reptile and amphibian communities to canopy gaps created by wind disturbance in the southern Appalachians. For Ecol Manage 148:135-144

Greenberg CH (2002) Fire, habitat structure and herpetofauna in the Southeast. In: Ford WM, Russell KR, Moorman CE (Eds.), The Role of Fire in Nongame Wildlife Management and Community Restoration: Traditional Uses and New Directions. Gen Tech Rep NE-288, USDA Forest Service Northeastern Forest Experiment Station, Newtown Square, Pennsylvania

Greenberg CH, Waldrop TA (2008) Short-term response of reptiles and amphibians to prescribed fire and mechanical fuel reduction in a southern Appalachian upland hardwood forest. For Ecol Manage 255:2883-2893

Harper CA, Guynn DC Jr (1999) Factors affecting salamander density and distribution within four forest types in the southern Appalachian mountains. For Ecol Manage 114:245-252

Harper EB (2007) The role of terrestrial habitat in the population dynamics and conservation of pond-breeding amphibians. Dissertation. University of Missouri, Columbia

Harpole DN, Haas CA (1999) Effects of seven silvicultural treatments on terrestrial salamanders. For Ecol Manage 114:349-356

Herbeck LA, Larsen DR (1999) Plethodontid salamander response to silvicultural practices in Missouri Ozark forests. Con Bio 13:623-632

Hocking DJ, Semlitsch RD (2007) Effects of timber harvest on breeding-site selection by gray treefrogs (*Hyla versicolor*). Bio Con 138:506-513

Homyack JA, Haas CA (2009) Long-term effects of experimental forest harvesting on abundance and reproductive demography of terrestrial salamanders. Biol Con 142:110-121

Jaeger RG (1980) Microhabitats of a terrestrial forest salamander. Copeia 1980:265-268

Johnson CM, Johnson LB, Richards C, Beasley V (2002) Predicting the occurrence of amphibians: an assessment of multiple-scale models. In: Scott JM, Heglund PJ, Haufler JB, Morrison ML, Raphael, MG, Wall WB, Sampson F (eds) Predicting Species Occurrences: Issues of Accuracy and Scale. Island Press, Washington DC

Johnson JR, Knouft JH, Semlitsch RD (2007) Sex and seasonal differences in the spatialterrestrial distribution of gray treefrog (*Hyla versicolor*) populations. Bio Con 140:250-258

Johnson JR, Mahan RD, Semlitsch RD (2008) Seasonal terrestrial microhabitat use by gray treefrogs (*Hyla versicolor*) in Missouri oak-hickory forests. Herpetologica 64:259-269

Keyser PD, Sausville DJ, Ford WM, Schwab DJ, Brose PH (2004) Prescribed fire impacts to amphibians and reptiles in shelterwood-harvested oak-dominated forests. Vir J Sci 55:159-168

Kirkland GL Jr, Snoddy HW, Miller TL (1996) Impact of fire on small mammals and amphibians in a central Appalachian deciduous forest. Am Midl Nat 135:253-260

Knapp SM, Haas CA, Harpole DN, Kirkpatrick RL (2003) Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Con Bio 17:752-762

Lowe WH, Bolger DT (2002) Local and landscape-scale predictors of salamander abundance in New Hampshire headwater streams. Con Bio 16:183-193

Marsh DM, Goicochea MA (2003) Monitoring terrestrial salamanders: biases cause by intense sampling and choice of cover objects. J Herp 37:460-466

Marsh DM, Beckman NG (2004) Effects of forest roads on the abundance and activity of terrestrial salamanders. Ecol Applic 14:1882-1891

Marsh DM, Thakur KA, Bulka KC, Clarke LB (2004) Dispersal and colonization through open fields by a terrestrial, woodland salamander. Ecol 85:3396-3405

Matthews CE, Moorman CE, Greenberg CH, Waldrop TA (2010) Response of herpetofauna to repeated fires and fuel reduction treatments. J Wildl Manage 74:1301-1310.

McLeod RF, Gates JE (1998) Response of herpetofaunal communities to forest cutting and burning at Chesapeake Farms, Maryland. Am Midl Nat 139:164-177

Messere M, Ducey P (1998) Forest floor distribution of northern redback salamanders, *Plethodon cinereus*, in relation to canopy gaps: first year following selective logging. For Ecol Manage 107:319-324

Mitchell JC, Rineharat SC, Pagels JF, Buhlmann KA, Pague CA (1997) Factors influencing amphibian and small mammal assemblages in central Appalachian forests. For Ecol Manage 96:65-76

Miller JE, Hess GE, Moorman CE (2007) Southern two-lined salamanders in urbanizing watersheds. Urban Ecosys 10:73-85

Moseley KR, Castleberry SB, Schweitzer SH (2003) Effects of prescribed fire on herpetofauna in bottomland hardwood forests. Southeast Nat 2:475-486

Moseley KR, Castleberry SB, Ford WM (2004) Coarse woody debris and pine litter manipulation effects on movement and microhabitat use of *Ambystoma talpoideum* in a *Pinus taeda* stand. For Ecol Manage 191:387-396

Moseley KR, Ford WM, Edwards JW (2009) Local and landscape factors influencing edge effects on woodland salamanders. Envir Mon Assess 151:425-435

Moseley KR, Ford WM, Edwards JW, Schuler TM (2008) Long-term partial cutting impacts on Desmognathus salamander abundance in West Virginia headwater streams. For Ecol Manage 254:300-307

Nislow KH, Lowe WH (2006) Influences of logging history and riparian forest characteristics on macroinvertebrates and brook trout (*Salvelinus fontinalis*) in headwater streams (New Hampshire, U.S.A.). Freshwater Bio 51:388-397

Patrick DA, Calhoun AJK, Hunter ML Jr (2008) The importance of understanding spatial population structure when evaluating the effects of silviculture on spotted salamanders (*Ambystoma maculatum*). Bio Con 141:807-814

Patrick DA, Hunter ML Jr, Calhoun AJK (2006) Effects of experimental forestry treatments on a Maine amphibian community. For Ecol Manage 234:323-332

Perkins DW, Hunter ML Jr (2006) Effects of riparian timber management on amphibians of Maine. J Wildl Manage 70:657-670

Peterman WE, Semlitsch RD (2009) Efficacy of riparian buffers in mitigating local population declines and the effects of even-aged timber harvest on larval salamanders. For Ecol Manage 257:8-14

Petranka JW (1999) Recovery of salamanders after clearcutting in the southern Appalachians: a critique of Ash's estimates. Con Bio 13:203-205

Petranka JW, Eldridge ME, Haley KE (1993) Effects of timber harvesting on southern Appalachian salamanders. Con Bio 7:363-370

Petranka JW, Brannon MP, Hopey ME, Smith CK (1994) Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. For Ecol Manage 67:135-147

Pough FH, Smith EM, Rhodes DH, Collazo A (1987) The abundance of salamanders in forest stands with different histories of disturbance. For Ecol Manage 20:1-9

Reichenbach N, Sattler P (2007) Effects of timbering on *Plethodon hubrichti* over twelve years. J Herp 41:622-629

Riedel BL (2006) Habitat relationships of red-backed salamanders (*Plethodon cinereus*) in Appalachian grazing systems. Thesis, University of Wisconsin – Stevens Point

Renken RB (2006) Does fire affect amphibians and reptiles in eastern U.S. oak forests? In: Dickinson MB (ed) Fire in Eastern Oak Forests: Delivering Science to Land Managers. Gen Tech Rep NRS-P-1. USDA Forest Service Northern Research Station, Newtown Square, Pennsylvania

Renken RB, Gram WK, Fantz DK, Richter SC, Miller TJ, Ricke KB, Russell B, Wang X (2004) Effects of forest management on amphibians and reptiles in Missouri Ozark forests. Cons Bio 18:174-188

Riedel BL, Russell KR, Ford WM, O'Neill KP, Godwin HW (2008) Habitat relationships of eastern red-backed salamanders (Plethodon cinereus) in Appalachian agroforestry and grazing systems. Ag, Ecosyst Envir 124:229-236

Rittenhouse TAG, Semlitsch RD (2009) Behavioral response of migrating wood frogs to experimental timber harvest surrounding wetlands. Can J Zool 87:618-625

Rittenhouse TAG, Harper EB, Rehard LR, Semlitsch RD (2008) The role of microhabitats in the desiccation and survival of anurans in recently harvested oak-hickory forest. Copeia 2008:807-814

Rittenhouse TAG, Semlitsch RD, Thompson FR III (2009) Survival costs associated with wood frog breeding migrations: effects of timber harvest and drought. Ecol 90:1620-1630

Ross B, Fredericksen T, Ross E, Hoffman W, Morrison ML, Beyea J, Lester MB, Johnson BN, Fredericksen NJ (2000) Relative abundance and species richness of herpetofauna in forest stands in Pennsylvania. For Sci 46:139-146

Rothermel BB, Luhring TM (2005) Burrow availability and desiccation risk of mole salamanders (*Ambystoma talpoideum*) in harvested versus unharvested forest stands. J Herp 39:619-626

Russell KR, Van Lear DH, Guynn DC Jr (1999) Prescribed fire effects on herpetofauna: review and management implications. Wildl Soc Bull 27:374-384

Russell KR, Hanlin HH, Wigley TB, Guynn DC Jr (2002) Responses of isolated wetland herpetofauna to upland forest management. J Wildl Manage 66:603-617

Russell KR, Wigley TB, Baughman WM, Hanlin HG, Ford WM (2004) Responses of southeastern amphibians and reptiles to forest management: A review. In: Rauscher, H.M., Johnsen, K. (eds). Southern Forest Science: Past, Present, and Future. Gen Tech Report SRS-75. USDA Forest Service Southern Research Station, Asheville, North Carolina

Russell KR, Mabee TJ, Cole MB, Rochelle MJ (2005) Evaluating biotic and abiotic influences on torrent salamanders in managed forests of western Oregon. Wildl Soc Bull 33:1413-1424

Semlitsch RD (2000) Principles for management of aquatic-breeding amphibians. J Wildl Manage 64:615-631

Semlitsch RD, Bodie JR (2003) Biological criteria for buffer zones around wetlands and riparian habitats for amphibians and reptiles. Con Bio 17:1219-1228

Semlitsch RD, Conner CA, Hocking DJ, Rittenhouse TAJ, Harper EB (2008) Effects of timber harvesting on pond-breeding amphibian persistence: Testing the evacuation hypothesis. Ecol Applic 18:283-289

Semlitsch RD, Ryan TJ, Hamed K, Chatfield M, Drehman B, Pekarek N, Spath M, Watland A (2007) Salamander abundance along road edges and within abandoned logging roads in Appalachian forests. Con Bio 21:159-167.

Semlitsch RD, Todd BD, Blomquist SM, Clhoun AJK, Gibbons JW, Gibbs JP, Graeter GJ, Harper EB, Hocking DJ, Hunter ML Jr, Patrick DA, Rittenhouse RAG, Rothermel BB (2009) Effects of timber harvest on amphibian populations: Understanding mechanisms from forest experiments. BioSci 59:853-862

Sinsch U (1990) Migration and orientation in anural amphibians. Ethol Ecol and Evol 2:65-79

Stebbins RC, Cohen NW (1995) A Natural History of Amphibians. Princeton University Press, Princeton, New Jersey

Todd BD, Andrews KM (2008) Response of a reptile guild to forest harvesting. Con Bio 22:753-761

Todd BD, Luhring TM, Rothermel BB, Gibbons JW (2009) Effects of forest removal on amphibian migrations: implications for habitat and landscape connectivity. J Appl Ecol 46:554-561

Webster CR, Jenkens MA (2005) Coarse woody debris dynamics in the southern Appalachians as affected by topographic position and anthropogenic disturbance history. For Ecol Manage 217:319-330

Whiles MR, Grubaugh JW (1996) Importance of coarse woody debris to southern forest herpetofauna. In: McMinn JW, Crossley DA Jr (Eds). Biodiversity and coarse woody debris in southern forests. Proceedings of the workshop on coarse woody debris in southern forests: effects on biodiversity. Gen Tech Rep SE94. USDA Forest Service Southeastern Forest Experiment Station, Asheville, NC

Zug, GR (1993) Herpetology: An Introductory Biology of Amphibians and Reptiles. Academic Press, Inc., San Diego, California

Table 11.1 Estimated recovery periods for terrestrial plethodontid salamander populations following timber harvest.

Authors	Recovery Period	Disturbance	Comments
Ash 1997	20-24 years	Clearcut	Monitored salamanders in 3 clearcuts using night searches on 225-m^2 plots for 15 years post-harvest and recovery times estimated from regression curves
Harper and Guynn 1999	13-39 years	Clearcut	Used a terrestrial vacuum to sample leaf litter and associated fauna in 120, 0.04-ha plots in 3 stand age classes (0-12, 23-39, and ≥40 years old)
Pough et al. 1987	<60 years	Clearcut	Conducted nighttime surveys for salamanders in 50- x 2-m transects in 4 disturbed stand types of different ages and in 4 paired old-growth sites
Homyack and Haas 2009	>60 years	Various Harvests	Conducted nighttime searches of 15- x 2-m transects for 13 years following 7 canopy removal treatments and estimated population recovery from demographic models
Petranka et al. 1993	50-70 years	Clearcut	Surveyed salamanders in 50- x 50-m plots at 47 sites ranging in age from 2 to 120 years old
Herbeck and Larsen 1999	>80 years	Regeneration cut	Conducted area- and time-constrained searches for salamanders in 21 144-m^2 plots located in 3 age classes (<5, 70-80, >120 years old)
Ford et al. 2002	>85 years	Clearcut	Captured salamanders in drift fence arrays in 13 cove hardwood stands ranging in age from 15 years old to >85 years old
Petranka et al. 1994	120 years	Clearcut	Conducted daytime searches for salamanders in 50 x 50-m plots at 52 forest sites ranging from <5 years old to approximately 200 years old