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Anuran occupancy and breeding site use of aquatic systems in a managed pine landscape

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ABSTRACT

Although the southeastern United States has >12 million ha of intensively managed pine forest, we have a poor understanding of how aquatic systems embedded in managed landscapes contribute to biodiversity. Further, the influence of local- and landscape-scale environmental factors on occupancy of aquatic habitat types by wetland-breeding species in managed forests is unclear. Thus, we investigated anuran occupancy across three ephemeral aquatic system types (altered sites, unaltered sites, and roadside ditches) embedded in an intensively managed pine landscape in eastern North Carolina, USA. These aquatic systems varied in management, disturbance intensity, and landscape context. Altered sites are actively managed as part of the surrounding plantation, unaltered sites are avoided by silvicultural activities (set aside), and ditches receive maintenance as part of routine forest management. We examined occupancy of anuran species at 53 aquatic sites surrounded by early-, mid-, or late-rotation aged stands. During January–July 2013–2014, we conducted repeated call surveys for anurans at aquatic sites and detected 14 species. We used single-species, multi-season occupancy models to examine associations between species occupancy and site- and landscape-scale habitat characteristics for 9 commonly encountered anurans as a function of aquatic system type and stand age class while accounting for imperfect detection. Detection probabilities by species ranged from 0.27 to 0.53 and increased seasonally through the year for most anurans. Species occupancy ranged in 2013 from 0.28 to 0.81 and in 2014 from 0.33 to 0.82. Species occupancy varied by aquatic system type, but the stand age surrounding an aquatic site had little effect on occupancy for most anurans we modeled. Our results indicate that local-scale factors commonly had a larger influence than landscape context on anuran occupancy. We detected evidence of breeding across all aquatic systems and stand age classes, suggesting at least a subset of species are calling and reproductively active. Our study highlights how novel landscape structure and reconfigured ephemeral aquatic systems embedded in intensively managed forests can support anuran occupancy across a range of disturbance intensities.

1. Introduction

Pine plantations in the southern United States are among the most intensively managed in the world (Schultz, 1997). Since 1952, land area covered by plantations has increased by 1672% to cover >12.9 million ha, with management intensity also increasing (Jokela et al., 2004; Fox et al., 2007). Typical silvicultural operations include mechanical and chemical site preparation, fertilization, thinning, clear-cut harvesting, and declining rotation lengths (Jokela et al., 2010) that produce a landscape mosaic of forest patches varying in age and structural conditions and benefit many species (Wigley et al., 2000; Hartley, 2002; Brockerhoff et al., 2008). Similarly, expansive areas of wet pine flats and pocosins wetlands were reconfigured in many areas of the Atlantic Coastal Plain by ditching and draining to remove excess water for supporting forestry, agriculture, and development (Cashin et al., 1992). However, forest landowners also set aside areas of ecological importance, including aquatic habitats, rare ecotypes, and riparian zones, which can still be abundant on reconfigured landscapes (Jones et al., 2010; Leonard et al., 2012). Thus, naturally derived aquatic habitat types and networks of ditches occur through large areas of managed forest, yet their contributions to amphibian diversity and occupancy are only beginning to be understood (Homyack et al., 2014, 2016).

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Many amphibians depend on aquatic sites surrounded by terrestrial habitat to complete their lifecycle (Semlitsch and Bodie, 2003). Aquatic sites provide habitat for breeding, larval development, and over-wintering, while adjacent terrestrial habitat provides adults and dispersing juveniles with foraging opportunities, escape cover, and hibernation or estivation sites. Among aquatic habitat types, small ephemeral pools that are often difficult to detect play critical ecological roles by supporting high species diversity and abundance of amphibians (Russell et al., 2002; Semlitsch and Bodie, 2003; Gibbons et al., 2006). Although natural forests typically support greater amphibian richness than altered areas (Guerry and Hunter, 2002; Denton and Richter, 2013; Walls et al., 2014), there is abundant evidence that pine plantations contribute to wildlife diversity and population maintenance (Wigley et al., 2000; Hartley, 2002; Brockerhoff et al., 2008; Homyack et al., 2014).

Prior research on the contributions of altered aquatic habitat types suggests that some anuran species can exhibit behavioral plasticity and use a variety of habitat types. For example, *Hyla* and *Pseudacris* species were more abundant in harvested gaps relative to undisturbed bottomland wetlands (Cromer et al., 2002) and other anurans used ditch systems in reconfigured landscapes, even where natural wetlands were present (Mazerolle, 2004; Homyack et al., 2014). Further, other altered wetlands and incidentally created aquatic habitat types, such as machinery ruts and blocked drainages, provide oviposition sites, but these may sometimes serve as population sinks (Adam and Lacki, 1993; DiMauro and Hunter, 2002).

At the global scale, herpetofauna are impacted negatively by habitat alteration, road mortality, disease, and other factors (Gibbons et al., 2000; Blaustein et al., 2011, 2012), but managed forests may provide a refuge from some salient threats (O'Bryan et al., 2016). Although numerous studies have examined effects of forest management in the Southern United States on amphibians (Cushman, 2006; Semlitsch et al., 2009), less is known about species-specific responses to different management strategies applied to wetlands within the forested matrix. In many forested systems of the Atlantic Coastal Plain, three kinds of aquatic systems occur on pine landscapes: unaltered sites that are avoided by silvicultural activities, altered sites that are actively managed as part of the surrounding plantation and not set aside, and roadside ditches. We investigated anuran occupancy and breeding effort in these aquatic system types embedded in an intensively managed pine landscape in the Atlantic Coastal Plain of North Carolina. We used occupancy models to examine the influence of forest structure and habitat characteristics at the site and landscape scale while accounting for imperfect detection. We hypothesized that roadside ditches would support occupancy of generalist species, and that altered and unaltered sites also would support occupancy of smaller frogs (e.g., *Hyla* and *Pseudacris* spp.) that have more specific habitat requirements, such as increased aquatic vegetation and a lack of predatory fish.

2. Materials and methods

2.1. Study area and aquatic system types

We conducted our study in an intensively managed loblolly pine (*Pinus taeda*) landscape in the Coastal Plain of North Carolina (Fig. 1). Plantation silviculture involved clearcutting mature stands (25–35 years old), followed by mechanical (V-shearing and bedding) and chemical (banded or broadcast herbicide prescribed at the stand-level) site preparation, loblolly pine seedlings planted at 1100 trees/ha, fertilization, and a commercial thinning entry (Homyack et al., 2014). Embedded in and adjacent to these stands exists a complex ditch network used to lower the water table to promote pine growth and improve operability. In addition, embedded in plantations are several types of ephemeral aquatic systems (Leonard et al., 2012; O'Bryan et al., 2016). The surrounding landscape was a mixture of forest, agriculture lands, and low-density residential housing.

From this landscape we selected and sampled the three dominant aquatic system types, which vary in the intensity of disturbance received during forest management activities: (1) unaltered sites, (2) altered sites, and (3) roadside ditches. Unaltered sites were excluded from harvesting, site preparation, and planting activities, and thus remained relatively undisturbed, often with a hardwood canopy (aquatic set-asides). Altered sites were typically small temporary depressions that received the same silvicultural regime as the surrounding stand and thus had been subject to harvesting, site preparation, and planting. Ditch sites were maintained approximately 5 years prior to this study and were adjacent to a single plantation. Because ditches were continuous linear systems, we randomly selected a 150-m transect/site by generating a random center point and sampled 75 m in either direction.

We selected aquatic sites embedded in (altered and unaltered sites) or adjacent to (roadside ditches) plantations across a range of stand ages. We stratified aquatic study sites by structural condition of the surrounding plantation (early, mid-, or late rotation age) and by landscape context (sites surrounded by forest to a mixture of forest, agriculture, and rural housing). Early rotation stands were open-canopy, regenerating clearcuts planted from 2008 to 2013 (mean = 2.4 years old, SE = 0.3). Mid-rotation stands were commercially thinned 2008–2013 to approximately 210 trees/ha and were 12–24 years old (mean = 15.5 years old, SE = 0.9). Late rotation stands were commercially thinned 1991–2003 (with the exception of one site thinned in 2008, but maintained structural conditions similar to other late aged stands) and were 21–40 years old (mean = 28.1 years old, SE = 1.4).

We identified potential altered aquatic sites using GIS data from previous remote sensing research that predicted the geographic location of small depressions in the study area (see Leonard et al., 2012 for details). We identified potential unaltered sites with GIS data and imagery, and identified potential roadside ditch sites using local forestry records. We cross-referenced potential sites with harvest plans and excluded those with a harvest planned during the project duration. From this pool, we visited sites within each category in a random order. Due to high variability in hydroperiod (i.e., sites may not have had water during site visits), we selected sites based on presence of aquatic vegetation, visible ground depressions, and/or standing water, and selected those $\geqslant$500 m away from the nearest site. We continued selecting sites until we reached our target sample size. We selected 51 of the 110 sites we visited in 2013 and added 2 additional sites in 2014 (one site in 2013 was not used in 2014). These included 16 unaltered aquatic sites (size range = 0.05–2.25 ha), 18 altered aquatic sites (size range = 0.02–0.86 ha), and 19 roadside ditches.

2.2. Aquatic site characteristics

At each site, we quantified habitat characteristics before (March) and after leaf-out (May) in 2013 and 2014. We measured the size of altered and unaltered sites by walking the perimeter of each site with a Garmin e-trex10 GPS unit (Garmin, Chicago, IL) with a WAAS accuracy of <3 m. We delineated perimeters of altered and unaltered sites by identifying the high water line, ground depressions, or abrupt changes in vegetation. We estimated the size of a ditch by measuring the width of the ditch at its widest point within the 150-m transect, then calculated 150 m $*$ ditch width. We estimated water depth (cm) at the deepest point and canopy openness using a