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## Do Ponds on Golf Courses Provide Suitable Habitat for Wetland-Dependent Animals in Suburban Areas? An Assessment of Turtle Abundances

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**ABSTRACT.**—Golf courses represent a common type of anthropogenically modified habitat in suburban environments. Golf courses may provide suitable habitat for semi-aquatic animals in suburban areas, yet studies comparing animal abundances in golf course ponds with other pond types in suburban environments are somewhat limited. In this study, we compared turtle abundances in golf course ponds with ponds found in residential areas and ponds found in rural (farm) areas and examined the relationship between turtle abundance and residential land-cover within individual golf courses. We captured turtles in 10 golf course ponds, 5 ponds surrounded by residential development, and 5 ponds located on farms. We estimated abundance and the effects of pond area, pond type (i.e., residential, golf, farm) and percentage of residential development within golf course boundaries. Using binomial mixture models and Bayesian inference, we found that ponds surrounded by residential development had lower abundances of Painted Turtles (*Chrysemys picta*) and Sliders (*Trachemys scripta*) than ponds located on golf courses and farms. Additionally, golf courses that have a greater amount of residential development within course boundaries had fewer turtles than courses that contained minimal residential development. Our results suggest that golf courses can offer suitable habitat for semi-aquatic turtles in suburban areas. However, residential development within golf course boundaries appears to have a negative effect on local abundances. Thus, if golf courses are to be seen as reserves for wetland-dependent animals, golf courses with low housing density should be considered as a more preferable option than courses associated with extensive residential development.

Animal conservation and management requires an in-depth knowledge of populations in a wide range of habitats, including those habitats highly altered by urban and suburban development (Adams and Lindsey, 2009). Urbanization has led to drastic changes in natural habitats and is a major factor contributing to the exceptional loss of wetlands and associated fauna in many areas of the United States (Hefner and Brown, 1985; Ricciardi and Rasmussen, 1999). Yet, urbanization has led to gains of some types of aquatic habitats (Dahl, 2006, 2011). For example, from 1998 to 2004, over 280,000 hectares of freshwater ponds have been created in the lower 48 United States due, in part, to the construction of storm water detention ponds, ponds in suburban parks, and ponds on recreational lands (Tilton, 1995; Dahl, 2006).

Golf courses represent a ubiquitous type of recreational land in suburban areas. For example, in the United States over 900,000 ha are dedicated to golf courses (GCSAA, 2007). Golf courses often include ponds and other aquatic habitats in the form of water hazards. Additionally, most golf courses are composed of >70% rough “out-of-play” areas (GCSAA, 2007) that may serve to buffer aquatic habitats from surrounding suburban environments or provide diverse habitats for a variety of organisms (Tanner and Gange, 2005). Furthermore, in suburban areas the total area dedicated to golf courses may exceed most nature reserves in overall size (Colding et al., 2006). Several recent investigations have indicated that golf courses and their aquatic habitats support diverse animal populations including amphibians (Scott et al., 2002; McDonough and Paton, 2007; Colding et al., 2009), invertebrates (Colding et al., 2009), birds (Terman, 1997; Jones et al., 2005; Rodewald et al., 2005; Tanner and Gange 2005; Sorace and Visentin 2007), and other taxa (Blair and Launer 1997; Tanner and Gange, 2005; Hodgkison et al., 2007). These studies suggest that golf courses may provide a refuge for many semi-aquatic species in suburban environments.

The potential of golf courses as habitat for semi-aquatic wildlife relies on the assumption that the benefits to wildlife outweigh the costs associated with inhabiting a highly managed area. Indeed, golf courses have been criticized strongly for their use of large quantities of water, application of herbicides and fertilizers, and establishment of exotic vegetation (Pearce, 1993; Warnken et al., 2001). Additionally, it is common for golf course designers to integrate residential development within the golf course itself (Nicholls and Crompton, 2007). In fact, it has been estimated that, during the 1990s, 46% of new golf courses were associated with real estate development (Mulvihill, 2001). Residential development, including roads, driveways, and buildings, within golf course boundaries likely fragments the terrestrial habitat critical for most semi-aquatic animals (Burke and Gibbons, 1995; Gibbons, 2003; Semlitsch and Bodie, 2003).

Turtles constitute a significant amount of the animal biomass in many types of aquatic environments (Iverson, 1982; Congdon et al., 1986) including those in urban and suburban areas (Conner et al., 2005; Ryan et al., 2008; Eskew et al., 2010a,b). Studies have suggested that turtle populations can persist in suburban environments due to higher growth rates (Budischak et al., 2006; Roe et al., 2011), higher fecundity than populations in rural areas (Lindemann, 1996), or both. However, turtle life history attributes suggest vulnerability to suburbanization. Most semi-aquatic turtles exhibit high adult survival and late sexual maturity; consequently, changes in the adult mortality rate can greatly influence population viability (Congdon et al., 1993). Turtle populations in urban and suburban areas often suffer from high levels of road mortality (Aresco, 2005; Steen et al., 2006) due to movements associated with nesting, dispersal, or hibernation (Bowne et al., 2006; Harden and Dorcas, 2008; Harden et al., 2009; but see Roe et al., 2011). Furthermore, nest predation by human-subsidized predators and local- and landscape-scale habitat degradation in urban and suburban areas likely effect turtle population persistence (Marchand and Litvaitis, 2004). Thus, the long-term viability of many turtle populations in suburban regions appears to be low.

In this study, our objectives were to 1) assess ponds on golf courses as semi-aquatic turtle habitat by comparing abundances of turtles in golf course ponds with abundances found in

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residential and rural ponds; and 2) examine the relationship between turtle abundance and land-cover heterogeneity of individual golf courses, focusing on percent residential land cover within the golf course boundary.

#### MATERIALS AND METHODS

**Study Sites.**—We conducted our study in the Charlotte metropolitan area in North Carolina, USA, specifically within Mecklenburg, Cabarrus, and Iredell counties. This region has seen a tremendous amount of growth over the last 40 yrs; current land cover is dominated by urban–suburban land, agriculture and pasture land, and second-growth forests (Price et al., 2006). We selected 10 golf course ponds to sample turtle populations and simultaneously sampled 10 ponds found off of golf courses which were either surrounded by residential development ( $N = 5$ ) or rural–farm land (i.e., pastures for livestock grazing;  $N = 5$ ). Naturally occurring ponds and wetlands within the Piedmont of North Carolina are rare and, thus, man-made ponds serve as the primary habitat for semi-aquatic turtles (Lindsay and Dorcas, 2001; Failey et al., 2007). Pond size on golf courses ranged from 0.45 ha to 6.13 ha (mean = 1.69 ha, SE = 0.63), whereas ponds on farms ranged from 0.15 ha to 1.54 ha (mean = 0.68 ha, SE = 0.24) and residential ponds ranged from 0.40 ha to 1.00 ha (mean = 0.65 ha, SE = 0.11).

**Turtle Sampling.**—We sampled turtles from early May through August 2010. Specifically, golf course ponds were sampled from 5 May through 28 August, residential ponds were sampled from 17 May until 21 July, and farm ponds were sampled from 6 May until 30 July. In each pond, we placed 10 hoop-net traps baited with sardines in shallow water around the pond perimeter. We checked traps every other day for 10 days. After capture, all turtles were identified to species, sexed using secondary sexual characters (Gibbons and Lovich, 1990; Ernst and Lovich 2009), and individually marked by filing notches in the marginal scutes (Sexton, 1959). We also weighed and measured (i.e., carapace length, plastron length, width, and depth) each turtle. All turtles were released within a few hours after initial capture. For the purposes of this study, we used counts (i.e., all captures) of males and females of each turtle species during each day of checking traps to estimate abundance. Turtles that could not be sexed accurately (i.e., juveniles) were excluded from the analysis.

**Analyses.**—We used binomial mixture models (Royle, 2004) to examine the effects of pond type (i.e., golf vs. farm vs. residential) and habitat covariates on turtle abundances. Binomial mixture models provide estimates of abundance and per-individual detection rate and can incorporate both site and survey covariates. Binomial mixture models assume a specific field sampling protocol consisting of replicate counts at a set of spatially distinct sites ( $i$ ) during temporally indexed surveys ( $j$ ), denoted as  $c_{ij}$  (Royle and Dorazio, 2008). The counts, which are modeled as independent outcomes of binomial sampling with index  $N_i$  and detection probability  $p_i$ , result in a joint probability mass function or a product-binomial model:

$$[y_i|N_i, p_i] = \Pi \text{Bin}(y_{ij}|N_i, p_i). \quad (1)$$

The introduction of additional parameters is needed to specify the relationship between the local-level parameters. Abundances ( $\lambda$ ) at the local level are often modeled with the Poisson distribution such that

$$N_i|\lambda_i \sim \text{Poi}(\lambda_i). \quad (2)$$

Heterogeneity in abundance among populations due to habitat

covariates ( $x_i$ ) can be modeled using a Poisson-regression formulation of local mean abundances given by  $\log(\lambda_i) = \beta_0 + \beta_1 x_i$ .

Per-individual detection probability ( $p$ ) follows a binomial distribution and is modeled according to:

$$c_{ij}|N_i \sim \text{Bin}(N_i, p_{ij}). \quad (3)$$

Sources of heterogeneity in detection can also be identified by modeling associations between sampling covariates and  $p_i$  such that  $\text{logit}(p_{ij}) = \alpha_0 + \alpha_1 x_{ij}$ .

To conduct our analysis, we first separated turtle count data by sex for each species. Because we were interested in turtle abundance between golf, farm, and residential ponds, we considered turtle abundance to be specified by:

$$\begin{aligned} N_i|\lambda_i &\sim \text{Poi}(\lambda_i) \\ \log(\lambda_i) &= \beta_0 + \beta_1 * \text{Golf Pond} + \beta_2 * \text{Rural Pond} + \beta_3 * \text{Pond Area}, \end{aligned} \quad (4)$$

where *Golf Pond* and *Farm Pond* were vectors of 1 or 0 dependent on the location of the pond. The covariate *Pond Area* (ha) was also added to this model, as pond area differed among our sites and may affect local abundances (Ficetola et al., 2004; Failey et al., 2007). Abundance at residential ponds was calculated via the intercept and *Pond Area* covariate.

Because we sampled some ponds in spring (i.e., May and June), whereas other ponds were sampled in summer months (i.e., July and August), we assumed the per-individual detection rate of turtles may differ among visits due to daily maximum air temperature ( $^{\circ}\text{C}$ ). Parmenter (1980) indicates that turtle feeding behavior, which may affect trap response, is strongly influenced by water temperature; however, these data were not available from all of our study sites. Thus, we used air temperature as a surrogate measure. We modeled heterogeneity in detection rate as:

$$\begin{aligned} c_{ij}|N_i &\sim \text{Bin}(N_i, p_{ij}) \\ \text{logit}(p_{ij}) &= \alpha_0 + \alpha_1 * \text{Temperature}. \end{aligned} \quad (5)$$

We standardized both the *Pond Area* and *Temperature* covariates so the mean of the population was 0 and the standard deviation was 1.

To estimate population abundances and model parameters for males and females of each species inhabiting golf course ponds and ponds located off golf courses, we used WinBUGS Version 2.10 (Spiegelhalter et al., 2003) with data handling in R (Ihaka and Gentleman, 1996). Our models used uninformative priors; specifically, we assumed  $\beta_0, \beta_1, \beta_2,$  and  $\beta_3, \sim N(0, 10^2)$ ,  $\alpha_0 \sim N(0, 1.6^2)$ , and  $\alpha_1 \sim N(0, 10^2)$ . The  $\alpha_0$  prior approximates a  $U(0, 1)$  prior for  $\text{expit}(\alpha_0)$ , where  $\text{expit}$  represents the inverse logit function (i.e.  $\exp(\alpha)/(1 + \exp(\alpha))$ ). Posterior summaries for each parameter were based on 100,000 Markov chain Monte Carlo iterations with a 20,000-sample burn-in and a thinning rate of 5. The mean and standard deviation of the model coefficients were calculated along with the 2.5 and 97.5 percentiles of the distribution, which represent 95% Bayesian credible intervals. Site-specific abundance estimates for each species were derived using the log transformation (i.e.  $\exp(\beta_0) * \exp(\beta_1 * \text{Golf Pond}) * \exp(\beta_2 * \text{Farm Pond}) * \exp(\beta_3 * \text{Pond Area})$ ).

To estimate the factors affecting species-specific heterogeneity in abundances among golf courses, we assumed that turtle abundance may differ due to golf course-specific factors, particularly the amount of residential land cover within the course boundary. Using a geographic information system (ArcGIS Desktop: Release 10, 2011; Environmental Systems

Research Institute [ESRI], Redlands, CA), we delimited the golf course boundary by creating a polygon to outline the outer border of the course. Secondly, using 2010 georeferenced 1':200'' digital orthophotos, we classified the land cover on each course, focusing on aquatic cover (ponds, lakes, rivers, etc.), green spaces (forest, recreational grassland, etc.), and urban and suburban areas (buildings, roads, manicured yards, etc.). We used the PLAND index in FRAGSTATS v.3.3 (McGarigal et al., 2002) to determine the amount of residential cover within the golf course boundary; this metric was calculated by taking the sum of the areas (m<sup>2</sup>) of all residential cover and dividing by the total area of the golf course. We considered abundances of turtle species on the golf course to be specified by:

$$\begin{aligned} N_i | \lambda_i &\sim \text{Poi}(\lambda_i) \\ \log(\lambda_i) &= \beta_0 + \beta_1 * \text{Pond Area} + \beta_2 * \% \text{Residential}. \end{aligned} \quad (6)$$

We standardized these covariates so the mean of the population was 0 and the standard deviation was 1. Similar to our first analysis, we assumed the per-individual detection rate of turtles may differ among visits due to daily maximum air temperature (°C) (Equation 5).

To estimate the influence of covariates for each species on golf courses, we used WinBUGS Version 2.10 (Spiegelhalter et al., 2003) with data handling in R (Ihaka and Gentleman, 1996). Our models used uninformative priors; specifically, we assumed priors for all  $\beta \sim N(0, 10^2)$ ,  $\alpha_0 \sim N(0, 1.6^2)$ , and  $\alpha_1 \sim N(0, 10^2)$ . Posterior summaries for each parameter were based on 100,000 Markov chain Monte Carlo iterations with a 20,000-sample burn-in and a thinning rate of 5. The mean and standard deviation of the model coefficients were calculated along with 95% Bayesian credible intervals.

## RESULTS

Trapping for turtles during the spring–summer of 2010 resulted in counts of 777 Painted Turtles (*Chrysemys picta*), 504 Sliders (*Trachemys scripta*), 85 Common Snapping Turtles (*Chelydra serpentina*), 48 Eastern Mud Turtles (*Kinosternon subrubrum*), 27 Musk Turtles (*Sternotherus odoratus*), and 3 River Cooters (*Pseudemys concinna*). Because of the low sample sizes of most species, we focused on abundances of male and female Painted Turtles and Sliders. Estimated abundances of male and female Painted Turtles were greater in larger ponds ( $\beta_{3(\text{males})} = 0.5076$ ; 95% CI = 0.412, 0.6047;  $\beta_{3(\text{females})} = 0.3708$ ; 95% CI = 0.2531, 0.4854), and estimated abundances were greater in golf course and farm ponds than in ponds located in residential areas (Fig. 1). Estimated abundances of Sliders were also greater in larger ponds ( $\beta_{3(\text{males})} = 0.8092$ ; 95% CI = 0.707, 0.9116;  $\beta_{3(\text{females})} = 0.6595$ ; 95% CI = 0.5321, 0.7926; Fig. 2). We found that abundances of male Sliders were similar in golf and farm ponds but lower in residential ponds (Fig. 2). Female Sliders were the most abundant in farm ponds, followed by golf course ponds and then residential ponds (Fig. 2). Per-individual detection probabilities of male Painted Turtles were positively correlated with daily high air temperature ( $\alpha_{1(\text{male})} = 0.1715$ ; 95% CI = 0.002, 0.3442) and averaged 0.26 (95% CI = 0.20, 0.33). Detection probabilities of female Painted Turtles,  $p = 0.32$  (95% CI = 0.25, 0.39) and male and female Sliders,  $p_{(\text{males})} = 0.09$  (95% CI = 0.05, 0.15);  $p_{(\text{females})} = 0.12$  (95% CI = 0.06, 0.20) were not significantly influenced by daily high air temperature.

The percent of course composed of residential cover varied among the 10 golf courses. The amount of developed land

within golf course boundaries ranged from 1.29% to 54.89% (mean = 24.77%, SE = 6.22%). Six of the golf courses contained >30% residential cover within the course boundaries. Similar to our first analysis, we found that the estimated abundance of all species and sexes was greater in larger ponds (Fig. 3). Painted Turtle ( $\beta_{2(\text{males})} = -0.3381$ ; 95% CI = -0.4978, -0.1834;  $\beta_{2(\text{females})} = -0.5183$ ; 95% CI = -0.7101, -0.3304; Fig. 3) and Slider ( $\beta_{3(\text{males})} = -0.4277$ ; 95% CI = -0.592, -0.2734;  $\beta_{3(\text{females})} = -0.2188$ ; 95% CI = -0.4263, -0.0231; Fig. 3) abundances were affected negatively by percent of golf course in residential cover.

## DISCUSSION

Our study showed that semi-aquatic turtle abundances on golf course ponds were equal to abundances in farm ponds; the one exception was abundances of female Sliders, which had greater abundance in farm ponds than in golf course ponds. However, both Sliders and Painted Turtles had the lowest abundances in ponds surrounded by residential development. In a recent review, Colding et al. (2006) state the majority of studies (64%) indicate that golf courses can play an important role in supporting regional biodiversity and, thus, hold high ecological value. For wetland-dependent fauna, such as amphibians, ponds on golf courses have been shown to provide important habitats, especially in urban areas with high road densities and little green space (Hodgkison et al., 2007; Colding et al., 2009). Our results suggest that golf courses can also offer habitat for semi-aquatic turtle populations in urbanized regions.

Painted Turtles and Sliders are among the most common and ubiquitous turtles in slow-moving, shallow aquatic environments in the Piedmont of North Carolina (Palmer and Braswell, 1995; Lindsay and Dorcas, 2001; Failey et al., 2007), yet these species, and many semi-aquatic turtles in general, share similarities in terms of sensitivity to residential development. Thus, it was not surprising that we found the lowest abundances in ponds surrounded by residential development. The observed differences between pond types in our study may be due to several factors including differences in pond attributes (i.e., food resources; Knight and Gibbons, 1968); higher nest predation in residential areas than on farmsteads and golf courses (Marchand and Litvaitis, 2004); increased mortality of adult turtles in suburban areas (Aresco, 2005); greater amounts of open canopy areas required for turtles to nest (Congdon and Gatten, 1989; Bodie and Semlitsch, 2000), or all of these. Differences in pond attributes, specifically nutrient levels, may be responsible for differences in abundance because this leads to greater aquatic plant growth, which is a food source for both Sliders and Painted Turtles (Knight and Gibbons, 1968; Gibbons, 2008). Although we did not measure nutrient loads, all of the ponds in our study were likely enriched through fertilization of lawns, fairways, or by cattle. Pond size is another attribute that may be responsible for differences in turtle abundance among pond types, as larger ponds often have greater semi-aquatic turtle abundance (Ingold et al., 1986; Lindeman, 1990; Failey et al., 2007). Our results corroborate this relationship, as we show that pond size is an important predictor of local abundances.

Residential areas are expected to have high abundances of human-subsidized predators that frequently consume turtle eggs (Marchand and Litvaitis, 2004). Although predation may be common in some regions, a simultaneous investigation at our study sites found predation rates on simulated turtle nests to be similar between golf, farm, and residential ponds (Foley et al., 2012). Perhaps a greater threat than predation at our study sites



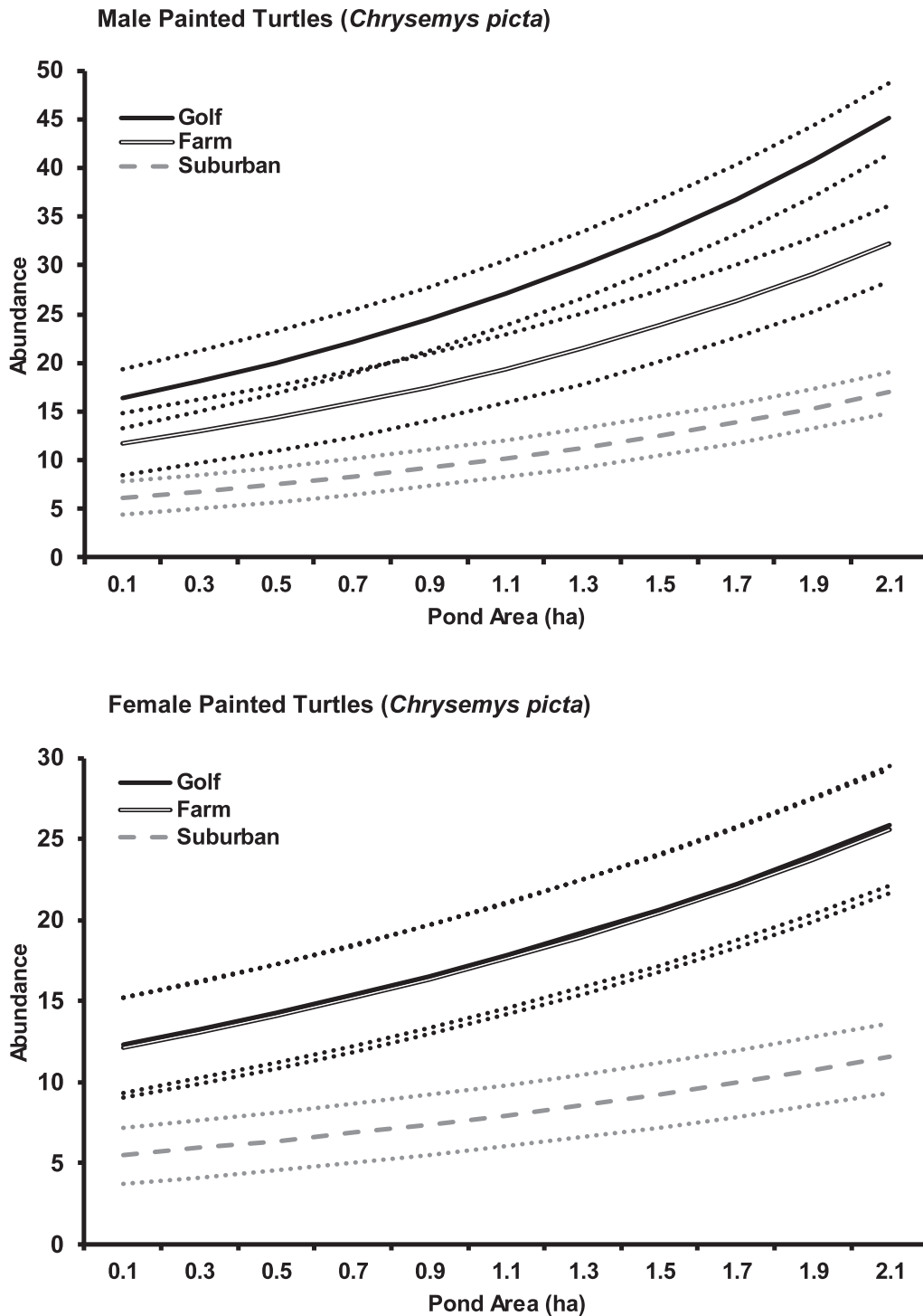


FIG. 1. Estimated mean abundances and 95% credible intervals (dotted lines) of male and female Painted Turtles (*Chrysemys picta*) as influenced by pond size (ha) and pond type (i.e., ponds located on golf courses, rural areas [farms], and residential areas) near Charlotte, North Carolina, USA.

is road mortality induced by movements associated with nesting and dispersal (Aresco, 2005; Steen et al., 2006). Female Sliders and Painted Turtles typically migrate >100 m to open-canopy nesting sites (Congdon and Gatten, 1989; Bodie and Semlitsch, 2000; Foley et al. 2012), and individuals may disperse up to 3,300 m among ponds in response to local environmental conditions (Bowne, 2008). Thus, traversing suburban environments to find appropriate habitat would likely increase

mortality risk for turtles and is a likely explanation for lower abundances in residential ponds.

Several factors of golf course and farm ponds may promote semi-aquatic turtle abundance. Characteristics include large expanses of open-canopy habitat for nesting (Congdon and Gatten, 1989; Bodie and Semlitsch, 2000), connectivity of habitat patches which may lead to successful dispersal among ponds (Bowne, 2008) and, in the case of golf courses, numerous ponds to inhabit. In a previous study, Foley et al. (2012) found that

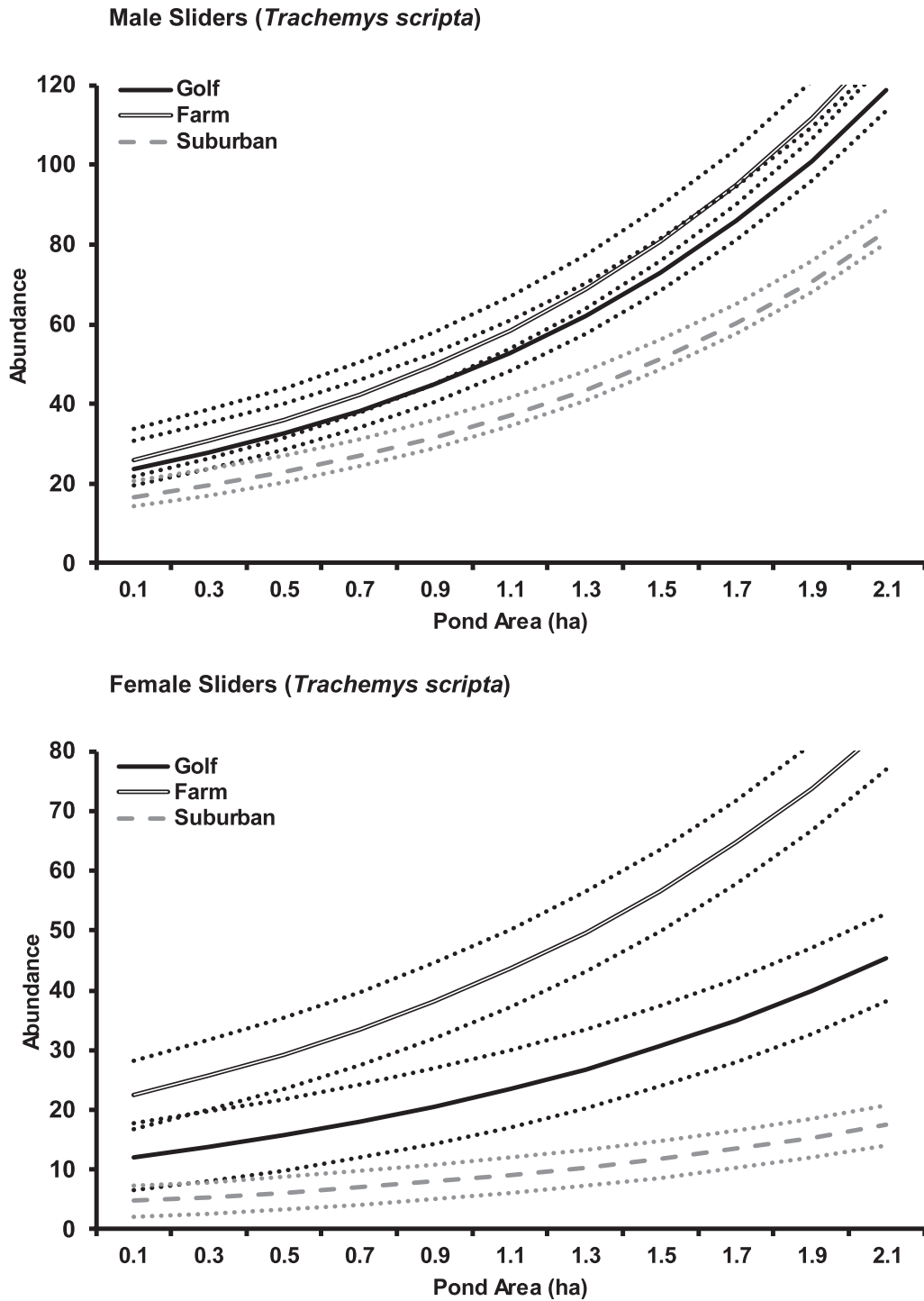


FIG. 2. Estimated mean abundances and 95% credible intervals (dotted lines) of male and female Sliders (*Trachemys scripta*) as influenced by pond size (ha) and pond type (i.e., ponds located on golf courses, rural areas [farms] and residential areas) near Charlotte, North Carolina, USA.

Painted Turtles inhabiting a golf course pond nested in close proximity to the pond (range 1–115 m and average 36 m) and selected areas covered in mulch or mowed grasses in which to nest. Given the low predation rates at our study sites (Foley et al., 2012) and the ubiquity of landscaped and mowed areas on golf courses, golf courses may benefit turtle abundances by contributing to high juvenile recruitment and by reducing adult mortality risk during overland movements.

Farm ponds share several characteristics with golf course ponds, including large amounts of open-canopy habitat for

nesting, which may have contributed to the similarities in estimated turtle abundance between farm and golf course ponds. One difference, however, between farm and golf course ponds is that several of the golf course ponds in our study had a significant amount of residential development (>30%) within the course boundaries. We found that turtle abundance was associated negatively with increasing residential land-cover within golf course boundaries, and residential areas within golf courses may explain why female Sliders, which are highly susceptible to road mortality during nesting forays due in part

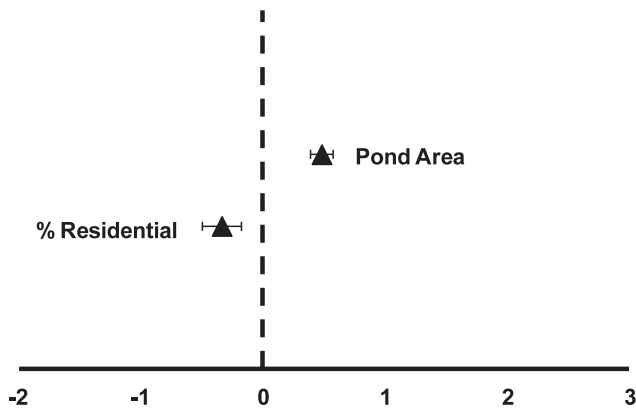
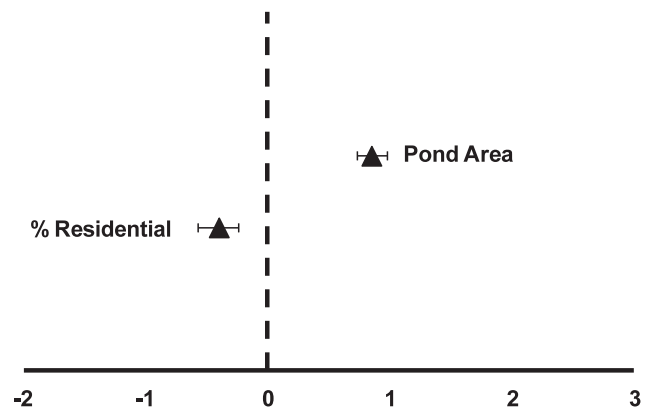
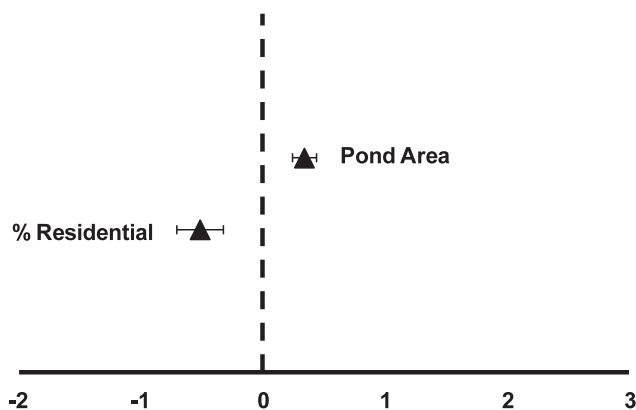
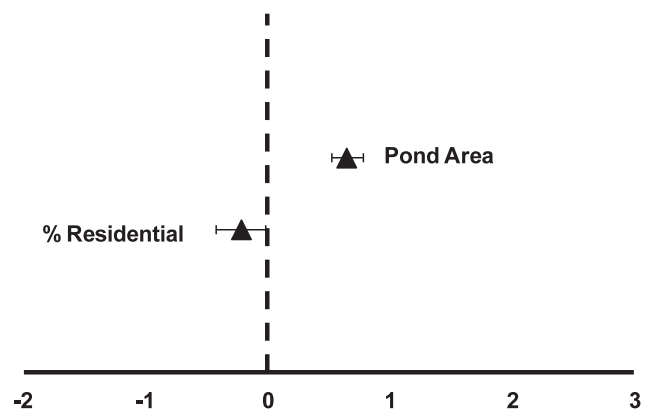
A. Male Painted Turtles (*Chrysemys picta*)C. Male Sliders (*Trachemys scripta*)B. Female Painted Turtles (*Chrysemys picta*)D. Female Sliders (*Trachemys scripta*)

FIG. 3. Estimates of  $\beta_1$  (pond size [ha]) and  $\beta_2$  (percent of the golf course covered in residential development) on abundances of (A) male Painted Turtles (*Chrysemys picta*), (B) female Painted Turtles (*C. picta*), (C) male Sliders (*Trachemys scripta*), and (D) female Sliders (*T. scripta*) counted at 10 golf course ponds near Charlotte, North Carolina, USA. Error bars represent 95% credible intervals. Species and sexes with parameter estimates (including 95% credible intervals) below or above zero represent a positive and negative effect, respectively, of the covariate.

to a larger size than male Sliders and Painted Turtles (Steen et al., 2006), had lower abundances in golf course ponds relative to farm ponds.

Researchers have suggested that golf courses should be included in residential designs in order to promote biodiversity conservation in urban areas (Colding et al., 2006). Yet, development trends indicate that a significant proportion of golf courses constructed in recent years are associated with real estate projects (Mulvihill, 2001). We found that golf courses that contained greater amounts of residential development within the course boundary negatively affected abundances of semi-aquatic turtles. Thus, it appears that golf courses containing substantial amounts of residential development may not provide the same benefits to semi-aquatic turtle populations as golf courses that have minimal development within course boundaries, suggesting that residential development within golf courses may significantly influence the potential a golf course holds as wildlife habitat.

Living adjacent to or within golf courses is generally viewed as a preferable place for human habitation (Mulvihill, 2001). One of the main reasons why humans choose to live near golf courses is the aesthetic qualities of green space and a country-like atmosphere (Nicholls and Crompton, 2007). These aesthetic

qualities also benefit a wide variety of wetland-dependent wildlife, including semi-aquatic turtles, and our analysis showed that golf courses had higher semi-aquatic turtle abundances than did ponds surrounded by residential development. However, our study suggests that some aspects, particularly the wildlife populations, that contribute to the aesthetics of golf courses appear to be compromised as residential areas are incorporated into golf course design. If golf courses are to be considered reserves to local biodiversity, promotion of naturalistic golf courses (i.e., those with minimal design and low housing density (Termon, 1997) should be considered as a more preferable option than courses associated with real estate development.

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