The Condor 114(4):689–697

YEAR-ROUND BIRD USE OF MONOTYPIC STANDS OF THE CHINESE TALLOW TREE, *TRIADICA SEBIFERA*, IN SOUTHEAST TEXAS

KATHERINE L. GIFFORD[1] AND JAMES W. ARMACOST JR.

Biology Department, Lamar University, Beaumont, TX 77710

Abstract. Invasive species pose the second greatest threat to biodiversity after habitat loss. Although invasive plant species negatively affect invaded ecosystems and diminish native biodiversity, they may provide food and other resources for some native birds. As monotypic stands of the invasive Chinese tallow tree (*Triadica sebifera*) become more common in the southeastern United States, it is important to assess their suitability as habitat for native forest birds. We used point counts to compare habitat use of forest birds in stands of native mixed-species forest, of mature tallow trees, and of young tallow trees on the coast of Texas during 2009 and 2010. The composition of trees in stands of mature and young tallow was more homogeneous than in stands of mixed native species, but mature tallow stands had an understory more complex than that of other habitat types. Mature tallow stands supported significantly fewer species of forest birds than did native forest only during the spring, and birds' population densities were similar in mature tallow and native forest throughout the year. Young tallow stands supported significantly fewer species of forest birds than did native forest in all seasons except for fall and significantly lower population densities during the breeding season (spring and summer). While monotypic stands of Chinese tallow trees provide suitable habitat for some forest birds, especially in winter, to preserve the widest diversity of forest birds we recommend the preservation of native mixed-species forest.

Key words: *avian habitat selection, avian species richness, Chinese tallow tree, invasive species, monotypic stand, Yellow-rumped Warbler.*

Uso de Rodales Monotípicos de *Triadica sebifera* Durante Todo el Año por Aves en el Sureste de Texas

Resumen. Las especies invasivas plantean la segunda mayor amenaza a la biodiversidad luego de la pérdida de hábitat. Aunque las especies de plantas invasivas afectan negativamente los ecosistemas invadidos y disminuyen la biodiversidad nativa, pueden brindar alimento y otros recursos a algunas aves nativas. Dado que los rodales monotípicos del árbol invasivo *Triadica sebifera* se volvieron más comunes en el sudeste de Estados Unidos, es importante evaluar su valor como hábitat para las aves de bosque nativas. Empleamos conteos por punto para comparar el uso de hábitat de las aves del bosque en rodales de especies mixtas nativas, de árboles maduros de *T. sebifera* y de árboles jóvenes de *T. sebifera* en la costa de Texas durante 2009 y 2010. La composición de los árboles en los rodales maduros y jóvenes de *T. sebifera* fue más homogénea que en los rodales de especies mixtas nativas, pero los rodales maduros de *T. sebifera* tuvieron un sotobosque más complejo que el de otros tipos de hábitat. Los rodales maduros de *T. sebifera* albergaron significativamente menos especies de aves de bosque que los bosques nativos solo durante la primavera y las densidades de las poblaciones de aves fueron similares en bosques maduros de *T. sebifera* y en nativos a lo largo del año. Los rodales jóvenes de *T. sebifera* albergaron significativamente menos especies de aves de bosque que los bosques nativos en todas las estaciones excepto en el otoño y densidades poblacionales significativamente más bajas durante la estación reproductiva (primavera y verano). Mientras que los rodales monotípicos de *T. sebifera* brindan hábitat adecuado para algunas aves de bosque, especialmente en invierno, para preservar la mayor diversidad de aves de bosque recomendamos la preservación de bosques de especies mixtas nativas.

Manuscript received 19 January 2012; accepted 28 March 2012.
[1]Current address: 2320A Lower Mission Valley Rd., Victoria, TX 77905. E-mail: klgifford@ymail.com

The Condor, Vol. 114, Number 4, pages 689–697. ISSN 0010-5422, electronic ISSN 1938-5422. DOI: 10.1525/cond.2012.120010

INTRODUCTION

Invasive plants have tremendous negative ecological and economic effects (Mack et al. 2000, Allendorf and Lundquist 2003, Lodge and Shrader-Frechette 2003). Among their ecological effects, invasive plants alter the physical structure of habitats, changing the suitability of the habitats for other species (Mack et al. 2000). Birds interact with invasive plants in many ways (reviewed in Richardson et al. 2000, Reichard et al. 2001). Many studies have documented the role of birds as agents of dispersal for invasive plants (e.g., Stiles 1982, Willson 1986, White and Stiles 1992, Witmer 1996, Renne et al. 2002, Bartuszevige 2004), but there has been less research on the effect of invasive plants on suitability of habitat for birds, via changes in the vegetative structure of the habitat (but see Cohan et al. 1978, Beissinger and Osborne 1982, Breytenbach 1986, Green et al. 1989, Griffin et al. 1989, Whelan and Dilger 1992, Rottenborn 1999, Schmidt and Whelan 1999).

Vegetation structure affects the species richness and composition of bird communities, the density of bird populations, and the foraging and nesting success of individual birds (Holmes and Robinson 1981, Cody 1985, Whelan 2001), typically through its influence on the availability and diversity of resources (Ellis 1995, Rumble et al. 2001, Brown et al. 2002). Vegetative communities differ in floristic and structural diversity, with monotypic stands representing one extreme. Monotypic stands, regardless of whether they are composed of a native, domesticated, or invasive plant, often have lower richness and a different composition of bird species than does more diverse vegetation, because monotypic stands have a more homogeneous vegetative structure and less diversity in resources (Knopf and Olsen 1984, Hunter et al. 1988, Rotenberg 2007). While richness of bird species is typically lower in monotypic stands, the population density of a given bird species in a monotypic stand may be lower than, similar to, or higher than its population density in more diverse vegetation, depending upon the resource the species requires. The population density of a bird species can be quite high in monotypic stands, even those of an invasive plant, if that plant provides an abundance of resources that the native vegetation does not (Wells et al. 1979, Ellis 1995, Brown et al. 2002). Monotypic stands play an important role in the annual cycles of some birds in winter and during migration (Sieg 1991, Ellis 1995, Brown et al. 2002).

The Chinese tallow tree, *Triadica sebifera* (Euphorbiaceae) is an invasive species of moist soils along the coast of the southern United States from Texas to North Carolina (Jubinsky and Anderson 1996). It often forms monotypic stands on abandoned agricultural lands, but it is generally unable to outcompete natives in well-established forests (Siemann and Rogers 2006). Its success is due to broad ecological tolerance (Jubinsky and Anderson 1996), high fecundity (Siemann and Rogers 2001, 2006), high growth rate (Scheld and Cowles 1981), and lack of predatory insects (Lankau et al. 2004). Birds facilitate the spread and growth of the tallow tree by dispersing its seeds (Renne et al. 2000) and by increasing the frequency and rate of seed germination by ingesting the seeds and passing them through the gut (Renne et al. 2001). The Chinese tallow tree is considered a serious threat to coastal prairie in Texas (Bruce et al. 1995, Siemann and Rogers 2006). It can alter the depth of the water table and availability of soil nutrients, change the vertical structure of the vegetation, and compete with and limit recruitment of native species (Gordon 1998). Controlled fires can reduce the probability of seed germination (Burns and Miller 2004) and can be used to remove young tallow trees, but once tallow trees become established, they are very difficult to eradicate (Grace et al. 2005).

Unfortunately, little of the abandoned agricultural land on the coastal plain of Texas is likely to be restored to coastal prairie, so much of it will become dominated by monotypic stands of the Chinese tallow tree. As they mature, the monotypic stands of the tallow tree become more similar structurally to forest than to grassland, so they are unlikely to support many native grassland birds, but little is known about their potential value as habitat for native forest birds. To that end, we compared vegetative structure, bird species richness, and bird population densities in three vegetation types: young monotypic stands of the tallow tree, mature monotypic stands of the tallow tree, and native mixed-species forest (which served as a control) on the coastal plain of southeast Texas. We compared the bird communities of these habitats throughout the year because previous studies of birds in tallow tree stands focused on the nonbreeding season (Conway et al. 2002, Baldwin 2005). We hypothesized that monotypic stands of the tallow tree (young and mature) should have a less complex, more homogeneous vegetative structure than does native mixed-species forest, that bird species richness in monotypic tallow tree stands (young and mature) should be lower than in native mixed-species forest, and that at all seasons birds' population densities should be lower in monotypic tallow tree stands (young and mature), except for some frugivorous species that might feed on tallow fruit and thus have higher densities in the tallow stands during the winter.

METHODS

We selected two locations with each vegetation type. Sabine Woods in Jefferson County (29.70° N, 93.95° W) and Dujay Sanctuary in Hardin County (30.33° N, 94.40 ° W) were control sites dominated by native mixed-species forest. Ninth Avenue (29.93° N, 93.95° W) and Dunigan Pasture (30.01° N, 94.04° W), both in Jefferson County, were sites dominated by mature tallow trees. Pipkin Ranch (29.74° N, 94.21° W and Taylor Ranch (29.77° N, 94.23° W), both in Jefferson County, were sites dominated by young tallow trees (Fig. 1).

Throughout the study we monitored 27 count points: nine in native mixed-species habitats, nine in mature tallow stands, and nine in young tallow stands. Bird surveys consisted of 50-m fixed-radius point counts for 10 min periods at each point. Wherever possible, we located points 100 m from habitat edges.

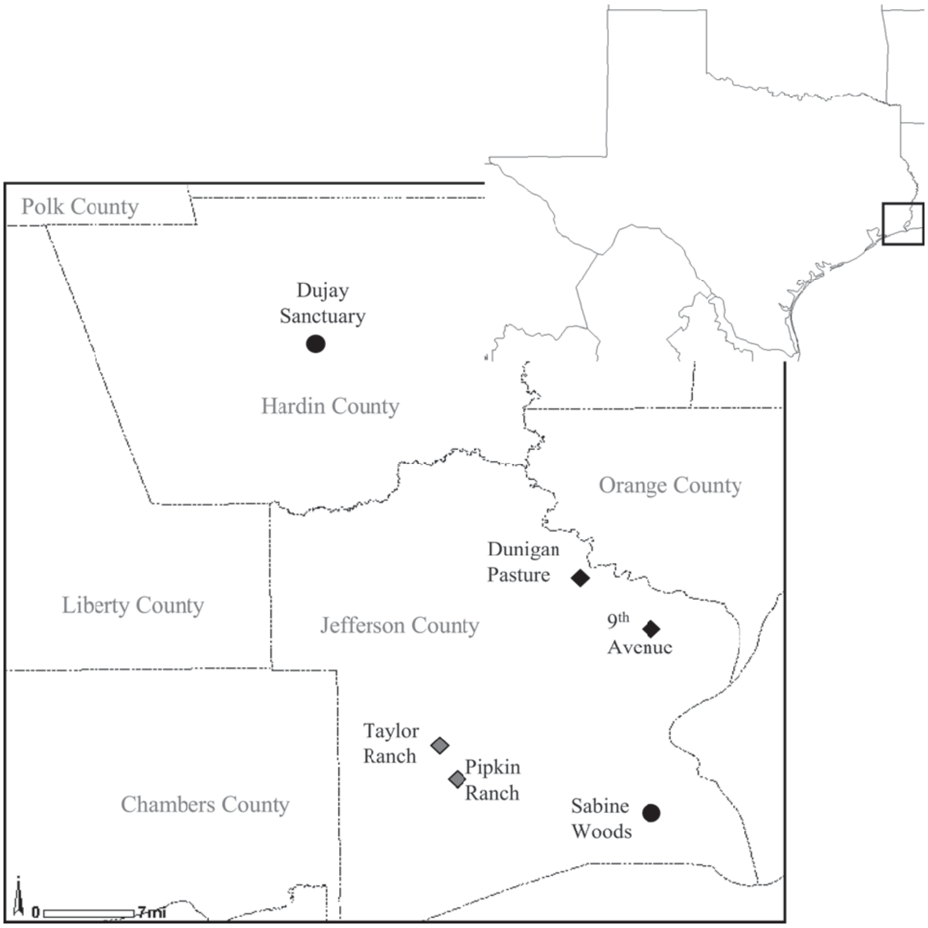

FIGURE 1. Location of the six study sites for assessment of the effect on birds of the Chinese tallow tree in southeast Texas. Gray diamonds, sites dominated by young Chinese tallow trees; black diamonds, sites dominated by mature Chinese tallow trees; black circles, control sites dominated by native mixed-species forest.

Counts were confined to the morning between 06:00 and 12:00 Central Standard Time and not conducted during rain or high wind (Ralph et al. 1993). Once every 2–3 weeks from August 2009 to August 2010, we recorded all birds detected within the circular plots, along with their distance from the center. We omitted from the analyses birds detected outside or flying over the plots. We recorded birds by the dependent double-observer method because it yields a detection probability and precision higher than the single-observer and independent double-observer methods, especially in forests (Forcey et al. 2006).

Our vegetation analysis, in June 2010, was based on the field protocol of the Breeding Biology Research and Monitoring Database (Martin et al. 1997). One vegetation plot was centered on the count point and consisted of a plot of radius 11.3 m in which we counted tree stems and a nested plot of radius 5 m in which we counted shrub stems and calculated ground cover. We divided the 5-m plot into quadrats with 5 m ropes at 90° from each other and extended in the cardinal directions. At the center of each plot, we measured the average top canopy height with a clinometer, the total canopy and high canopy cover (above 5 m) with a densiometer, and we recorded the dominant plant species (at least 40%) in the high canopy and, if present, a co-dominant plant species (at least 40%). In the 5-m plot, we counted shrubs taller than 50 cm. Depth of litter on the ground was averaged from 12 measurements recorded at 2-m intervals along the quadrat lines. In each of the quadrats we estimated the percent cover below a height of 50 cm of the following categories: green vegetation (all live vegetation including grass and marsh vegetation), grass, brush (dead woody vegetation), fern, moss, leaf litter, downed logs, marsh vegetation, and bare ground. In the 11.3-m plot, we recorded the species and diameter at breast height (DBH) of live trees and of snags (standing, dead trees) and the percentage of tallow trees (number of tallow trees/number of all trees). To determine if landscape variables affected the species richness or population density of forest birds, we measured the area of the study site (ha), distance to the nearest forest patch (km) of the same size as the study site or larger, and percent forest cover within a 1-km radius of each study site (Baldwin 2005) with Google Earth.

We compared the mean species richness of forest birds within each of the three habitats with an analysis of variance (ANOVA). We calculated population densities of the most common forest species with Distance 6.0, version 2 (Thomas et al. 2010), which corrects for detectability, then compared these densities by habitat with a *Z*-test (Buckland et al. 2001). Although we analyzed species separately, resulting in multiple comparisons among habitat types, to avoid type II errors we used an uncorrected a of 0.05 as the criterion of significance. Even with this less conservative approach, we detected few significant differences, mainly because of high variances. We reduced the number of vegetation variables with a principal component analysis (PCA) and used an ANOVA to compare the habitat types along the principal component axes and to compare the various vegetation variables by habitat. We used equivalent nonparametric tests when the data failed to meet the assumptions of normality or equality of variances. We used linear regression to determine if species richness and abundance of forest birds were related to the landscape variables, analyzing each season (summer, fall, winter, and spring) separately. For the PCA we used SPSS student version 15.0, for all other analyses, SigmaPlot 11.0.

RESULTS

During the 542 point counts, we observed 137 bird species, including 3243 individuals of 69 species of forest birds, retained in the subsequent analyses.

Species richness of forest birds was similar in mature tallow stands and native, mixed-species forest during the summer, fall, and winter (Tables 1, 2, and 3). In the spring, we found significantly fewer species of forest birds in the mature tallow stands than in the native, mixed-species forest ($P < 0.05$; Table 4), when migrants such as the Yellow-billed Cuckoo (*Coccyzus americanus*), Eastern Wood-Pewee (*Contopus virens*), Acadian Flycatcher (*Empidonax virescens*), Red-eyed Vireo (*Vireo olivaceus*), House Wren (*Troglodytes aedon*), Swainson's Thrush (*Catharus ustulatus*), Wood Thrush (*Hylocichla mustelina*), Black-and-white Warbler (*Mniotilta varia*), Yellow-throated Warbler (*Setophaga dominica*), Orchard Oriole (*Icterus spurius*), and Baltimore Oriole (*Icterus galbula*) were present in the native forest but absent from the mature tallow stands.

The total population density of all species of forest birds combined was similar in mature tallow stands and in native, forest during all seasons (Tables 1–4). The density of the Northern Cardinal (*Cardinalis cardinalis*) was significantly higher in mature tallow stands than in native forest during the fall ($P < 0.001$; Table 2), which may indicate dispersal of young birds from native forest into the tallow stands. The densities of the Carolina Chickadee (*Poecile carolinensis*) and Ruby-crowned Kinglet (*Regulus calendula*) were significantly lower in mature tallow stands than in native forest during the winter (both $P = 0.01$; Table 3). The density of the Yellow-rumped Warbler (*Setophaga coronata*) was higher in mature tallow stands than in native forest during the winter (no statistical test, as the species was too rare in the native forest for its density there to be estimated).

Species richness of forest birds was significantly lower in young tallow stands than in native forest during the winter, spring, and summer (all $P < 0.05$; Tables 1, 3, and 4). During the fall, species richness of forest birds in young tallow stands and in native forest was similar (Table 2).

The total population density of all forest bird species combined was significantly lower in young tallow stands than in native forest during the breeding season (spring and summer) ($P = 0.03$ and $P = 0.001$, respectively; Tables 1 and 4), but the difference was not significant during the nonbreeding season (fall and winter; Tables 2 and 3). The density of Northern Cardinals was significantly lower in young tallow stands than in native forest during the spring and summer (both $P = 0.001$; Tables 1 and 4). The density of Ruby-crowned Kinglets and Northern Cardinals was lower in young tallow than in native forest during the winter ($P = 0.003$ and $P = 0.03$, respectively; Table 3), but the density of Eastern Phoebes (*Sayornis phoebe*) was higher in young tallow stands in the winter ($P < 0.001$; Table 3). The density of Yellow-rumped Warblers was higher in young tallow stands than in native forest during the winter (no statistical test, as the Yellow-rumped Warbler was too rare in the native forest for its density there to be estimated).

In the vegetation analysis, we considered 16 habitat variables (Table 5). Of these, only four (percent live vegetation cover, percent log cover, number of snags, and mean snag DBH) did not differ significantly by site. Canopy height decreased, and the number of trees other than tallow decreased significantly from the control to the two tallow-dominated habitats ($P < 0.001$) and from a few in mature tallow to none in young tallow sites ($P = 0.04$). Litter depth ($P < 0.001$) and percent cover of dead leaves ($P < 0.05$) were significantly higher at control sites than at mature or young tallow sites but did not differ significantly between mature and young tallow sites. Canopy cover, number of shrubs, and mean DBH of trees other than tallow were significantly higher at control and mature tallow sites than young tallow sites ($P < 0.05$) but did not differ significantly between control and mature tallow sites. Percent bare ground ($P < 0.001$), number of tallow trees, mean DBH of tallow trees, and percent tallow (all $P < 0.05$) were significantly higher at mature and young tallow sites than at control sites but did not differ significantly between mature and young tallow sites. The percent grass cover was significantly higher at young tallow sites than at control sites ($P = 0.01$), but mature tallow sites did not differ significantly from either young tallow or control sites.

In the PCA of these 16 habitat variables, the first three principal components explained 89% of the variance among the sites. The first component, which explained 54% of the variance among sites, represented a gradient from a low canopy and high percentage of tallow trees (i.e., young tallow stands) to a tall canopy and low percentage of tallow trees (i.e., native forest). All three sites differed significantly along the first principal component axis ($P < 0.05$). The second principal component, which

TABLE 1. Comparison of the species richness and population density of common forest birds[a] in young and mature stands of the Chinese tallow tree and in native forest in southeast Texas during the summer.

Calculation	Control		Mature		Young		P^b C vs. M	P^b C vs. Y	P^b M vs. Y
Mean (± SE) no. species point^{-1}	4.1 ± 0.2		3.2 ± 0.2		1.4 ± 0.1		n.s.[c]	<0.05*	<0.05*
Density (total birds ha^{-1}, ± SE)	8.7 ± 2.8	315	7.3 ± 3.0	237	2.0 ± 0.6	109	0.371	0.001*	0.040*
Red-bellied Woodpecker (*Melanerpes carolinus*)	0.5 ± 0.2	24	0.2 ± 0.1	11	—[d]	1	0.082	—	—
Downy Woodpecker (*Picoides pubescens*)	0.6 ± 0.2	20	0.7 ± 0.3	19	—[d]	5	0.367	—	—
White-eyed Vireo (*Vireo griseus*)	1.2 ± 0.5	46	1.0 ± 0.3	42	—	0	0.417	—	—
Carolina Chickadee (*Poecile carolinensis*)	0.6 ± 0.2	27	0.5 ± 0.2	15	—	0	0.417	—	—
Carolina Wren (*Thryothorus ludovicianus*)	0.7 ± 0.2	20	0.7 ± 0.3	23	—[d]	1	0.433	—	—
Northern Mockingbird (*Mimus polyglottos*)	—[d]	6	—[d]	6	1.2 ± 0.3	60	—	—	—
Common Yellowthroat (*Geothlypis trichas*)	—[d]	1	1.0 ± 0.8	39	—	0	—	—	—
Northern Cardinal (*Cardinalis cardinalis*)	3.5 ± 0.7	103	2.7 ± 0.7	75	0.5 ± 0.2	16	0.218	<0.001*	0.002*

[a]Considered common if $n \geq 30$ in all habitats combined.
[b]Asterisks specify value significant at $\alpha = 0.05$.
[c]Not a significant difference.
[d]$n < 10$, too small for reliable estimate of density.

explained 20% of the variance among sites, represented a gradient from lower to greater density of shrubs. The mature tallow and control sites and the mature tallow and young tallow sites differed significantly along this second axis ($P < 0.05$), but the control and young tallow sites did not differ significantly from each other. A graph of the first two principal components shows the sites in the three habitat types clustered together (Fig. 2), indicating clear differences in vegetative structure by habitat type. The third principal component explained 15% of the variance among sites but was most highly correlated with percent live vegetation, which did not differ significantly by site.

Regression analysis of patch area, distance to nearest patch, and percent forest cover within 1 km of the sites with species richness and abundance of forest birds revealed no significant correlations during summer, winter, or spring. During the fall, there was a positive correlation between species

TABLE 2. Comparison of the species richness and population density of common forest birds[a] in young and mature stands of the Chinese tallow tree and in native forest in southeast Texas during the fall.

Calculation	Control		Mature		Young		P^b C vs. M	P^b C vs. Y	P^b M vs. Y
Mean (± SE) no. species point^{-1}	4.6 ± 0.6		3.6 ± 0.3		3.2 ± 0.4		n.s.[c]	n.s.	n.s.
Density (total birds ha^{-1}, ± SE)	14.3 ± 8.9	147	9.9 ± 3.9	171	11.3 ± 12.3	67	0.326	0.421	0.460
Downy Woodpecker (*Picoides pubescens*)	0.6 ± 0.2	16	0.7 ± 0.3	14	—	0	0.401	—	—
Eastern Phoebe (*Sayornis phoebe*)	—[d]	7	0.7 ± 0.3	14	—[e]	16	—	—	—
Carolina Wren (*Thryothorus ludovicianus*)	3.0 ± 1.3	34	2.2 ± 0.5	39	4.7 ± 3.9	16	0.281	0.341	0.261
Gray Catbird (*Dumetella carolinensis*)	0.4 ± 0.2	10	1.3 ± 0.6	13	—[d]	9	0.087	—	—
Northern Cardinal (*Cardinalis cardinalis*)	1.8 ± 0.4	28	5.9 ± 1.1	55	—[e]	10	<0.001*	—	—

[a]Considered common if $n \geq 30$ in all habitats combined.
[b]Asterisks specify value significant at $\alpha = 0.05$.
[c]Not a significant difference.
[d]$n < 10$, too small for reliable estimate of density.
[e]Standard error too high for reliable estimate of density.

TABLE 3. Comparison of the species richness and population density of common forest birds[a] in young and mature stands of the Chinese tallow tree and in native forest in southeast Texas during the winter.

Calculation	Control		Mature		Young		P^b C vs. M	P^b C vs. Y	P^b M vs. Y
Mean (± SE) no. species point^{-1}	5.7 ± 0.4		4.7 ± 0.3		3.3 ± 0.2		n.s.[c]	<.05*	<.05*
Density (total birds ha^{-1}, ± SE)	27.4 ± 15.2	496	33.5 ± 7.5	450	17.7 ± 6.3	233	0.359	0.278	0.053
Density without the Yellow-rumped Warbler	22.3 ± 8.9	420	11.8 ± 4.5	216	9.9 ± 3.2	143	0.149	0.095	0.363
Red-bellied Woodpecker (*Melanerpes carolinus*)	1.0 ± 0.3	34	1.0 ± 0.4	17	—[d]	2	0.500	—	—
Downy Woodpecker (*Picoides pubescens*)	1.5 ± 0.5	34	0.9 ± 0.4	28	—[d]	2	0.161	—	—
Eastern Phoebe (*Sayornis phoebe*)	0.4 ± 0.2	11	—	0	4.2 ± 0.8	38	—	<0.001*	—
Carolina Chickadee (*Poecile carolinensis*)	1.9 ± 0.8	25	0.3 ± 0.2	10	—	0	0.017*	—	—
Carolina Wren (*Thryothorus ludovicianus*)	1.0 ± 0.4	24	1.0 ± 0.4	20	0.8 ± 0.4	16	0.500	0.413	0.413
Ruby-crowned Kinglet (*Regulus calendula*)	3.9 ± 0.9	39	1.3 ± 0.7	17	1.0 ± 0.4	18	0.014*	0.003*	0.356
American Robin (*Turdus migratorius*)	4.1 ± 2.6	78	0.8 ± 0.5	14	0.8 ± 0.5	14	0.106	0.104	0.492
Gray Catbird (*Dumetella carolinensis*)	0.5 ± 0.3	17	0.7 ± 0.3	17	—[d]	7	—	—	—
Northern Mockingbird (*Mimus polyglottos*)	—[d]	8	—[d]	3	1.6 ± 0.6	34	—	—	—
Cedar Waxwing (*Bombycilla cedrorum*)	0.8 ± 0.4	14	0.5 ± 0.2	19	—[d]	3	0.264	—	—
Yellow-rumped Warbler (*Setophaga coronata*)	—[e]	76	21.7 ± 3.0	234	7.8 ± 3.2	90	—	—	0.001*
Pine Warbler (*Setophaga pinus*)	1.2 ± 0.4	34	1.7 ± 0.5	20	—[d]	3	0.230	—	—
Northern Cardinal (*Cardinalis cardinalis*)	3.5 ± 1.0	72	3.7 ± 0.9	54	1.4 ± 0.4	23	0.436	0.031*	0.011*

[a]Considered common if $n \geq 30$ in all habitats combined.
[b]Asterisks specify value significant at $\alpha = 0.05$.
[c]Not a significant difference.
[d]$n < 10$, too small for reliable estimate of density.
[e]Standard error too high for reliable estimate of density.

TABLE 4. Comparison of the species richness and population density of common forest birds[a] in young and mature stands of the Chinese tallow tree and in native forest in southeast Texas during the spring.

Calculation	Control		Mature		Young		P^b C vs. M	P^b C vs. Y	P^b M vs. Y
Mean (± SE) no. species point^{-1}	5.3 ± 0.3		3.7 ± 0.3		1.6 ± 0.2		<0.05*	<0.05*	<0.05*
Density (total birds ha^{-1}, ± SE)	12.1 ± 4.6	278	10.6 ± 4.0	179	2.8 ± 1.4	78	0.401	0.026*	0.034*
White-eyed Vireo (*Vireo griseus*)	0.8 ± 0.3	27	1.1 ± 0.4	27	—	0	0.278	—	—
Ruby-crowned Kinglet (*Regulus calendula*)	2.7 ± 1.0	40	—[c]	2	—[c]	4	—	—	—
Yellow-rumped Warbler (*Setophaga coronata*)	1.4 ± 0.8	24	1.6 ± 0.6	20	1.0 ± 0.6	21	0.397	0.378	0.248
Northern Cardinal (*Cardinalis cardinalis*)	3.6 ± 0.7	87	5.1 ± 1.3	75	0.6 ± 0.2	21	0.149	<0.001*	<0.001*

[a]Considered common if $n \geq 30$ in all site types combined.
[b]Asterisks specify value significant at $\alpha = 0.05$.
[c]$n < 10$, too small for reliable estimate of density.

TABLE 5. Comparison of the vegetation variables in young and mature stands of the Chinese tallow tree and in native forest in southeast Texas.

	Mean ± SE			P[a]		
Habitat variable	Control	Mature	Young	C vs. M	C vs. Y	M vs. Y
Top canopy height (m)	17.6 ± 1.2	7.5 ± 1.1	3.8 ± 0.7	<0.001*	<0.001*	0.036*
Canopy cover	88.9 ± 1.3	83.0 ± 5.8	20.3 ± 6.7	n.s.[b]	<0.05*	<0.05*
Depth of litter on ground (mm)	28.6 ± 3.5	6.3 ± 2.8	0.4 ± 0.4	<0.001*	<0.001*	0.223
Grass cover (%)	7.8 ± 4.5	27.2 ± 11.1	52.2 ± 13.2	0.348	0.010*	0.184
Cover of other live vegetation (%)	32.2 ± 10.0	56.1 ± 12.2	38.9 ± 12.5	n.s.	n.s.	n.s.
Cover of leaf litter (%)	93.3 ± 2.0	26.1 ± 7.0	4.4 ± 1.6	<0.05*	<0.05*	n.s.
Log cover (%)	3.9 ± 0.8	9.4 ± 2.9	17.2 ± 5.7	n.s.	n.s.	n.s.
Bare ground (%)	1.7 ± 0.9	64.4 ± 7.6	78.3 ± 5.9	<0.001*	<0.001*	0.170
Number of shrubs	27.6 ± 8.3	44.7 ± 12.2	4.2 ± 1.9	n.s.	<0.05*	<0.05*
Number of snags	1.6 ± 1.4	0.7 ± 0.3	1.0 ± 0.6	n.s.	n.s.	n.s.
Mean snag DBH (cm)	5.5 ± 3.1	3.1 ± 2.8	5.9 ± 3.5	n.s.	n.s.	n.s.
Number of trees other than tallow	13.0 ± 1.5	4.2 ± 1.4	0.0 ± 0.0	<0.001*	<0.001*	0.032*
Mean DBH (cm) of trees other than tallow	23.8 ± 3.2	23.5 ± 6.0	0.0 ± 0.0	n.s.	<0.05*	<0.05*
Number of tallow trees	0.0 ± 0.0	20.4 ± 4.8	11.1 ± 4.2	<0.05*	<0.05*	n.s.
Mean DBH (cm) of tallow	0.0 ± 0.0	12.7 ± 0.4	13.7 ± 0.8	<0.05*	<0.05*	n.s.
Percent of tallow	0.0 ± 0.0	80.0 ± 10.0	100.0 ± 0.0	<0.05*	<0.05*	n.s.

[a]Asterisks specify value significant at $\alpha = 0.05$.
[b]Not a significant difference.

richness and distance to nearest patch ($R^2 = 0.754$; $P = 0.03$) and abundance and distance to nearest patch ($R^2 = 0.688$; $P = 0.04$).

DISCUSSION

Mature tallow stands had a shorter canopy and a higher percentage of tallow trees but a denser shrub layer than did native, mixed-forest; canopy closure was similar. Species richness and population density of forest birds were similar in mature tallow and native forest in all seasons except spring, when species richness of forest birds was lower in mature tallow stands, due to the absence of some migrants. Despite their low percentage of native trees, mature tallow stands, with a closed canopy and dense shrub layer, provided suitable habitat for many forest bird. Other studies have also shown that habitat use by birds is more dependent upon vegetation structure than on species composition (Rotenberry and Wiens 1980, Rotenberry 1985). Native birds may use stands of invasive plants; for example, saltcedar (*Tamarix* spp.), although less suitable than native vegetation, is an important habitat for many riparian birds in the southwestern United States (Hunter et al. 1988, Ellis 1995, van Riper et al. 2008).

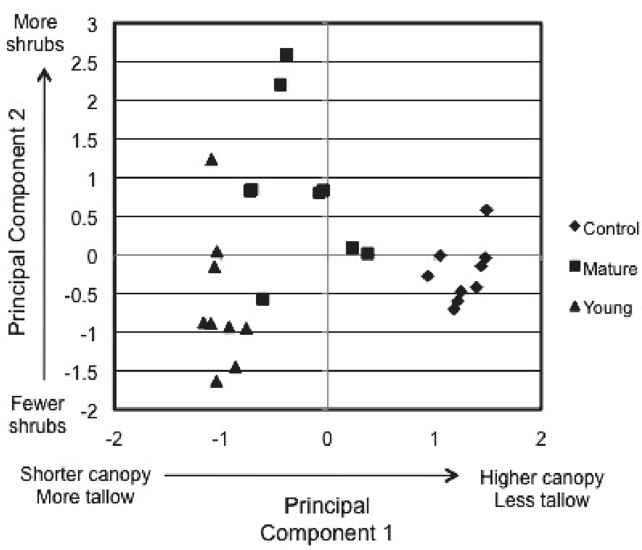

FIGURE 2. Distribution of characteristics of young and mature stands of the Chinese tallow tree and of native forest in southeast Texas along the axes of the first two principal components.

Young tallow stands had a lower, more open canopy, a higher percentage of tallow trees, and a sparser shrub layer than did native forest. Species richness of forest birds was lower in young tallow stands than in native forest in all seasons except fall, when forest birds may have dispersed into young tallow stands after breeding. Other studies have documented the dispersal of young birds into early successional habitats not typically used by forest species during the breeding season (Anders et al. 1998, Rivera et al. 1998). The population density of forest birds was lower in young tallow stands than in native forest during the breeding season but not during the nonbreeding season, when forest birds such as the Yellow-rumped Warbler foraged in young tallow stands for tallow fruit and other resources. Although forest birds used young tallow stands during the nonbreeding season, these stands, with their low, open canopies and sparse shrub layers, did not provide suitable habitat for breeding forest birds. Conway et al. (2002) documented foraging by

24 species of birds in tallow stands on the Texas coast during the fall. Baldwin (2005) found lower species richness of birds in tallow stands than in hardwood forest on the Louisiana coast during the winter, but relative population densities of birds in tallow versus hardwood forest varied by species, with three species being more common in tallow stands, six more common in hardwood forest, and six showing no difference.

Landscape features, such as patch size and isolation, generally had little effect on either species richness or population density of forest birds, with the exception of distance to nearest patch during the fall, when we found more birds at Sabine Woods, an important stopover site for migratory birds despite its isolation. Other studies have demonstrated that some of the decrease in species richness in monotypic stands of vegetation is often attributable to the small size and isolation of the monotypic patches within a matrix of differing vegetation (Sieg 1991, Johns 1993, Turchi et al. 1995, Grant and Berkey 1999, Rotenberg 2007). The near lack of effects of area or isolation indicates that forest birds are capable of finding and using tallow stands if and when they offer suitable resources.

Chinese tallow has many detrimental effects on invaded habitats, particularly coastal prairies (Bruce et al. 1995, Gordon 1998, Siemann and Rogers 2006), but monotypic stands of Chinese tallow are likely to become an increasingly prominent feature of the coast of the Gulf of Mexico, so it is useful to assess their suitability as habitat for native forest birds. We found that mature tallow stands provide habitat for forest birds throughout the year, but young stands are useful mainly during the nonbreeding season, and few forest birds use the young tallow stands during the breeding season. Although our conclusions are based on a single year of data, they generally agree with the conclusions of other studies of bird use of tallow tree stands (Conway et al. 2002, Baldwin 2005). Stands of the Chinese tallow tree can provide food and other resources for native birds, especially during the nonbreeding season, as can other invasive plants (Ellis 1995, Reichard et al. 2001, Stoleson and Finch 2001), but management that encourages the preservation of native forest will best ensure the maintenance of overall avian biodiversity, especially during the breeding season.

ACKNOWLEDGMENTS

A large portion of this research was funded by the Dujay Sanctuary Fund, granted to Lamar University by the late Eva Dujay. We thank the following landowners for granting us access to their properties: Billy Doornbos (C. Doornbos, Inc.), Joe Dunnigan, Grayson Pipkin, Joe Taylor, Lamar University, and the Texas Ornithological Society.

We thank Richard Harrel, Randall Terry, and Robert Corbett for providing helpful advice and comments throughout this project. We thank two anonymous reviewers for additional useful comments on the manuscript. Finally, we thank Cody Conway for his assistance in the field with the point counts.

LITERATURE CITED

Allendorf, F. W., and L. L. Lundquist. 2003. Introduction: population biology, evolution, and control of invasive species. Conservation Biology 17:24–30.

Anders, A. D., J. Faaborg, and F. R. Thompson III. 1998. Postfledging dispersal, habitat use, and home-range size of juvenile Wood Thrushes. Auk 115:349–358.

Baldwin, M. J. 2005. Winter bird use of the Chinese tallow tree in Louisiana. M.Sc. thesis, Louisiana State University, Baton Rouge, LA.

Bartuszevige, A. M. 2004. The influence of habitat suitability, landscape structure, and seed dispersers on invasion of an exotic plant species, *Lonicera maackii* (Rupr.) Herder, Amur honeysuckle. Ph.D. dissertation, Miami University, Oxford, OH.

Beissinger, S. R., and D. R. Osborne. 1982. Effects of urbanization on avian community organization. Condor 84:75–83.

Breytonbach, G. J. 1986. Impacts of alien organisms on terrestrial communities with emphasis on communities of the south-western Cape, p. 229–238. *In* I. A. W. MacDonald, F. J. Kruger, and A. A. Ferrar [eds.], The ecology and management of biological invasions in southern Africa. Oxford University Press, Cape Town.

Brown, D. R., C. M. Strong, and P. C. Stouffer. 2002. Demographic effects of habitat selection by Hermit Thrushes wintering in a pine plantation landscape. Journal of Wildlife Management 66:407–416.

Bruce, K. A., G. N. Cameron, and P. A. Harcombe. 1995. Initiation of a new woodland type on the Texas coastal prairie by the Chinese tallow tree (*Sapium sebiferum* (L.) Roxb.). Bulletin of the Torrey Botanical Club 122:215–225.

Buckland, S. T., D. Anderson, K. Burnham, J. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling. Oxford University Press, Oxford, UK.

Burns, J. H., and T. E. Miller. 2004. Invasion of Chinese tallow (*Sapium sebiferum*) in the Lake Jackson area, northern Florida. American Midland Naturalist 152:410–417.

Cody, M. L. 1985. An introduction to habitat selection in birds, p. 3–56. *In* M. L. Cody [ed.], Habitat selection in birds. Academic Press, San Diego.

Cohan, D. R., B. W. Anderson, and R. D. Ohmart. 1978. Avian population response to salt cedar along the lower Colorado River, p. 371–382. *In* Strategies for protection and management of flood plain wetlands and other riparian ecosystems: Proceedings of the symposium December 11–13, 1978. USDA Forest Service General Technical Report WO-12.

Conway, W. C., L. M. Smith, and J. F. Bergan. 2002. Avian use of Chinese tallow seeds in coastal Texas. Southwestern Naturalist 47:550–556.

Ellis, L. M. 1995. Bird use of saltcedar and cottonwood vegetation in the middle Rio Grande Valley of New Mexico, U.S.A. Journal of Arid Environments 30:339–349.

Forcey, G. M., J. T. Anderson, F. K. Ammer, and R. C. Whitmore. 2006. Comparison of two double-observer point-count approaches for estimating breeding bird abundance. Journal of Wildlife Management 70:1674–1681.

Gordon, D. R. 1998. Effects of invasive, non-indigenous plant species on ecosystem processes: lessons from Florida. Ecological Applications 8:975–989.

Grace, J. B., L. K. Allain, H. Q. Baldwin, A. G. Billock, W. R. Eddleman, A. M. Given, C. W. Jeske, and R. Moss. 2005. Effects of prescribed fire in the coastal prairies of Texas. U.S. Geological Survey Open File Report 2005–1287.

Grant, T. A., and G. B. Berkey. 1999. Forest area and avian diversity in fragmented aspen woodland of North Dakota. Wildlife Society Bulletin 27:904–914.

Green, R. J., Catterall, C. P., and D. N. Jones. 1989. Foraging and other behavior of birds in subtropical and temperate suburban habitats. Emu 89:216–222.

Griffin, G. F., Stafford-Smith, D. M., Morton, S. R., Allan, G. E., and K. A. Masters. 1989. Status and implications of the invasion of tamarisk (*Tamarix aphylla*) on the Finke River, Northern Territory, Australia. Journal of Environmental Management 29:297–315.

Holmes, R. T., and S. K. Robinson. 1981. Tree species preference of foraging insectivorous birds in a northern hardwood forest. Oecologia 48:31–35.

Hunter, W. C., Ohmart, R. D., and B. W. Anderson. 1988. Use of exotic saltcedar (*Tamarix chinensis*) by birds in arid riparian systems. Condor 90:113–123.

Johns, B. W. 1993. The influence of grove size on bird species richness in aspen parklands. Wilson Bulletin 105:256–264.

Jubinsky, G., and L. C. Anderson. 1996. The invasive potential of Chinese tallow-tree (*Sapium sebiferum* Roxb.) in the southeast. Castanea 61:226–231.

Knopf, F. L., and T. E. Olsen. 1984. Naturalization of Russian olive: implications to Rocky Mountain wildlife. Wildlife Society Bulletin 12:289–298.

Lankau, R. A., W. E. Rogers, and E. Siemann. 2004. Constraints on the utilisation of the invasive Chinese tallow tree *Sapium sebiferum* by generalist native herbivores in coastal prairies. Ecological Entomology 29:66–75.

Lodge, D. M., and K. Shrader-Frechette. 2003. Nonindigenous species: ecological explanation, environmental ethics, and public policy. Conservation Biology 17:31–37.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M., Clout, and F. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecological Applications 10:689–710.

Martin, T. E., C. R. Paine, C. J. Conway, W. M. Hochachka, P. Allen, and W. Jenkins. 1997. BBIRD field protocol. Montana Cooperative Wildlife Research Unit, Missoula, MT.

Ralph, C. J., G. R. Geupel, P. Pyle, T. E. Martin, and D. F. DeSante. 1993. Handbook of field methods for monitoring landbirds. General Technical Report PSW-GTR-144, Pacific Southwest Research Station, Albany, CA.

Reichard, S. H., Chalker-Scott, L., and S. Buchanan. 2001. Interactions among non-native plants and birds, p. 179–223. *In* J. M. Marzluff, R. Bowman, and R. Donnelly [eds.], Avian ecology and conservation in an urbanizing world. Kluwer Academic, Boston.

Renne, I. J., W. C. Barrow Jr., L. A. Johnson-Randall, and W. C. Bridges Jr. 2002. Generalized avian dispersal syndrome contributes to Chinese tallow tree (*Sapium sebiferum*, Euphorbiaceae) invasiveness. Diversity and Distributions 8:285–295.

Renne, I. J., S. A. Gauthreaux Jr., and C. A. Gresham. 2000. Seed dispersal of the Chinese tallow tree (*Sapium sebiferum* (L.) Roxb.) by birds in coastal South Carolina. American Midland Naturalist 144:202–215.

Renne, I. J., T. P. Spira, and W. C. Bridges. 2001. Effects of habitat, burial, age and passage through birds on germination and establishment of Chinese tallow tree in coastal South Carolina. Journal of the Torrey Botanical Society 128:109–119.

Richardson, D. M., N. Allsopp, C. M. D'Antonio, S. J. Milton, and M. Rejmanck. 2000. Plant invasions—the role of mutualisms. Biological Review 75:65–93.

Rivera, J. H., J. H. Rappole, W. J. McShea, and C. A. Haas. 1998. Wood Thrush postfledging movements and habitat use in northern Virginia. Condor 100:69–78.

Rotenberg, J. A. 2007. Ecological role of a tree (*Gmelina arborea*) plantation in Guatemala: an assessment of an alternative land use for tropical avian conservation. Auk 124:316–330.

Rotenberry, J. T. 1985. The role of habitat in avian community composition: physiognomy or floristics? Oecologia 67:213–217.

Rotenberry, J. T., and J. A. Wiens. 1980. Habitat structure, patchiness and avian communities in North American steppe vegetation: a multivariate analysis. Ecology 61:1228–1250.

Rottenborn, S. C. 1999. Predicting the impacts of urbanization on riparian bird communities. Biological Conservation 88:289–299.

Rumble, M. A., L. D. Flake, T. R. Mills, and B. L. Dykstra. 2001. Do pine trees in aspen stands increase bird diversity? U.S. Department of Agriculture Forest Service Proceedings RMRS-P-18:185–192.

Scheld, H. W., and J. R. Cowles. 1981. Woody biomass potential of the Chinese tallow tree. Economic Botany 35:391–397.

Schmidt, K. A., and C. J. Whelan. 1999. Effects of exotic *Lonicera* and *Rhamnus* on songbird nest predation. Conservation Biology 13:1502–1506.

Sieg, C.H. 1991. Rocky Mountain juniper woodlands: year-round avian habitat. U.S. Department of Agriculture Forest Service Research Paper RM-296:1–7.

Siemann, E., and W. E. Rogers. 2001. Genetic differences in growth of an invasive tree species. Ecology Letters 4:514–518.

Siemann, E., and W. E. Rogers. 2006. Recruitment limitation, seedling performance and persistence of exotic tree monocultures. Biological Invasions 8:979–991.

Stiles, E. W. 1982. Expansions of mockingbird and multiflora rose in the northeastern United States and Canada. American Birds 36:358–364.

Stoleson, S. H., and D. M. Finch. 2001. Breeding bird use of and nesting success in exotic Russian olive in New Mexico. Wilson Bulletin 113:452–455.

Thomas, L., S. T. Buckland, E. A. Rexstad, J. L. Laake, S. Strindberg, S. L. Hedley, J. R. B. Bishop, T. A. Marques, and K. P. Burnham. 2010. Distance software: design and analysis of distance sampling surveys for estimating population size. Journal of Applied Ecology 47:5–14.

Turchi, G. M., P. L. Kennedy, D. Urban, and D. Hein. 1995. Bird species richness in relation to isolation of aspen habitats. Wilson Bulletin 107:463–474.

Van Riper, C. III, K. L. Paxton, C. O'Brien, P. B. Shafroth, and L. J. McGrath. 2008. Rethinking avian response to *Tamarix* on the lower Colorado River: a threshold hypothesis. Restoration Ecology 16:155–167.

Wells, D., B. W. Anderson, and R. D. Ohmart. 1979. Comparative avian use of southwestern citrus orchards and riparian communities. Journal of the Arizona–Nevada Academy of Science 14:53–58.

Whelan, C. J. 2001. Foliage structure influences foraging of insectivorous forest birds: an experimental study. Ecology 82:219–231.

Whelan, C. J., and M. L. Digler. 1992. Invasive, exotic shrubs: a paradox for natural area managers. Natural Areas Journal 12:109–110.

White, D. W., and E. W. Stiles. 1992. Bird dispersal of fruits of species introduced into eastern North America. Canadian Journal of Botany 70:1689–1696.

Willson, M. F. 1986. Avian frugivory and seed dispersal in eastern North America, p. 223–279. *In* R. F. Johnston [ed.], Current Ornithology, vol. 3. Plenum, New York.

Witmer, M. C. 1996. Consequences of an alien shrub on the plumage coloration and ecology of Cedar Waxwings. Auk 113:735–743.