

Research Article

Effect of Removal of Woody Biomass after Clearcutting and Intercropping Switchgrass (*Panicum virgatum*) with Loblolly Pine (*Pinus taeda*) on Rodent Diversity and Populations

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Plant-based feedstocks have long been considered viable, potential sources for biofuels. However, concerns regarding production effects may outweigh gains like carbon savings. Additional information is needed to understand environmental effects of growing feedstocks, including effects on wildlife communities and populations. We used a randomized and replicated experimental design to examine initial effects of biofuel feedstock treatment options, including removal of woody biomass after clearcutting and intercropping switchgrass (*Panicum virgatum*), on rodents to 2 years post-treatment in regenerating pine plantations in North Carolina, USA. Rodent community composition did not change with switchgrass production or residual biomass removal treatments. Further, residual biomass removal had no influence on rodent population abundances. However, *Peromyscus leucopus* was found in the greatest abundance and had the greatest survival in treatments without switchgrass. In contrast, abundance of invasive *Mus musculus* was greatest in switchgrass treatments. Other native species, such as *Sigmodon hispidus*, were not influenced by the presence of switchgrass. Our results suggest that planting of switchgrass, but not biomass removal, had species-specific effects on rodents at least 2 years post-planting in an intensively managed southern pine system. Determining ecological mechanisms underlying our observed species associations with switchgrass will be integral for understanding long-term sustainability of biofuels production in southern pine forest.

1. Introduction

Plant-based feedstocks have long been considered viable, potential sources for biofuels, which are defined as liquid fuels derived from biological materials [1] that could displace fossil fuel consumption. However, concerns with production effects on prices of food crops, the carbon footprint of large-scale production, potential land-use change, and effects on biodiversity may outweigh gains in carbon savings or other benefits [1, 2]. Therefore, additional information is needed related to sustainability of biofuels production and to understand environmental impacts of growing biofuel feedstocks, including impacts on biodiversity [3–5].

Source landscapes for biofuel feedstocks have expanded from agricultural to forested as technologies have developed to transform cellulosic portions of woody plants into liquid fuels at a production scale. In managed forests, biofuels typically have been produced by harvesting biomass including forest residues (i.e., tops, stumps, limbs, or unmerchantable trees that remain after harvesting and removing crop trees) or by planting short rotation woody species, such as aspen (*Populus* spp.) or eucalyptus (*Eucalyptus* spp.) [6, 7]. However, more recent efforts also have examined whether perennial grasses such as switchgrass (*Panicum virgatum*), a species native to eastern North America, could be intercropped between rows of crop trees in pine (*Pinus* spp.) plantations

so that both traditional forest products and cellulosic energy crops could be produced on the same land base. Intercropping biofuels on managed forests may avoid concerns related to land-use change or the food versus fuel debate while still producing a carbon-neutral source of energy [1, 8]. Recently, there has been increasing interest in intercropping switchgrass within intensively managed pine stands within the southeastern United States with pilot projects being conducted to assess feasibility. For example, in 2008, Catchlight Energy LLC, a Chevron/Weyerhaeuser Joint Venture (<http://www.catchlightenergy.com/>), was formed in part to assess the large-scale viability of such a management system. However, environmental effects of producing biofuels from forests, either from harvesting residual woody biomass or by intercropping perennial grasses, have not been well studied [1, 6, 9].

Rodents are model species for understanding ecological effects of anthropogenic activities for a multitude of reasons [10–12]. They contribute to species and functional diversity of the animal community [13] and are ecosystem engineers that create, modify, and maintain habitat structures (such as burrows) that influence nutrient cycling, soil aeration and habitat use by other animal species [14]. Rodents directly influence distribution and abundance of many species through their use of a variety of food sources (e.g., plants, lichen, fungi, and invertebrates) and as prey for many vertebrate predators [13].

Abundances of small mammals are influenced by changes in habitat structure, including removal of woody debris [15, 16] and cover of grasses [17, 18], so biofuels production may affect population dynamics and habitat relationships of rodents. Population abundance can be influenced by changes in habitat structure and quality, and a range of demographic parameters determines population abundance. The primary influences on population abundance are apparent survival and recruitment, which together indicate reproductive success, mortality, immigration to and emigration from a population. Additionally, adult sex ratio is important because it helps determine the effective population size. Thus, population abundance can be influenced by habitat structure, and with supporting demographic data, can be used to infer habitat quality [19]. Community composition and diversity of small mammals also vary with habitat structure [20, 21] suggesting that changes to habitat associated with biofuels production could also affect rodent diversity and abundance.

To better understand effects of producing biofuels from forested landscapes on animal communities and populations, we examined initial outcomes of a range of biofuels treatment options, including a biomass removal harvest and intercropping of switchgrass, on rodent community structure and population demographics. Population demographic parameters we examined included population abundance, recruitment, survival, and sex ratio for two years following treatment application. We hypothesized that rodent population abundance and community diversity would be greater in treatments where habitat heterogeneity was the greatest. We also hypothesized that population metrics related to habitat quality, including recruitment, survival, and male:female ratios would reflect greater habitat quality in treatments

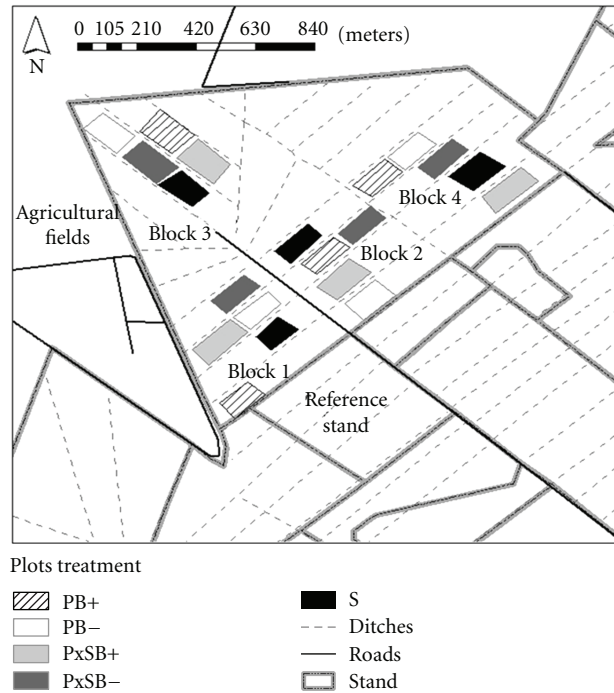


FIGURE 1: Map of study area, established and maintained by Catchlight Energy LLC, located in timberlands owned and managed by Weyerhaeuser Company in Lenoir Co., NC. Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P × SB+ = Pine × Switchgrass Biomass+; P × SB- = Pine × Switchgrass Biomass-; S = Switchgrass.

where more nest sites were available, such as treatments without a biomass removal harvest. This study is the first that we are aware of to experimentally examine a range of biofuels options and their effects on rodent communities and populations.

2. Methods

2.1. Study Area and Experimental Design. Our study was conducted in the southeastern coastal plain on the Lenoir 1 Sustainability Study Site, established and maintained by Catchlight Energy LLC on land owned and managed by Weyerhaeuser Company in Lenoir County, North Carolina, USA. The region is sandy bottomland forest dominated by agriculture and intensively managed loblolly pine (*Pinus taeda*) forests. The study site was a 72-ha loblolly pine plantation established in 1974 with a site index of 70 for loblolly pine at 25 years (by 25 years of growth, loblolly pine trees can be expected to reach 70-feet (21.3 m) in height due to site potential). Ground water levels were maintained via linear ditches that occurred along forest edges and paralleled each other through interiors of study site blocks (Figure 1).

Within the study site, a 33.6 ha research area was divided into 20 experimental plots that were clear-cut harvested, site prepared, and replanted with loblolly pine seedlings and/or switchgrass in 2008-09 (Figure 1). The experimental design was a randomized complete block with plots as the

experimental unit (4 replicates) and study sites as blocks (4 study blocks and 5 treatments per block). Plots averaged 1.11 ha in size (range: 0.76–1.39 ha; Figure 1). Within a block, experimental plots were randomly assigned one of the following five treatments:

2.1.1. Pine with Residual Woody Debris in Place, (Pine Biomass+, PB+). Treatment plots had standard mechanical site preparation (V-shearing of stumps and roots, subsoiling, and bedding at 6.1 m between rows). Loblolly pines planted in December 2008 were centered on raised beds and spaced 1–2 m apart. Residual woody debris was left on site.

2.1.2. Pine with Residual Woody Debris Removed, (Pine Biomass–, PB–). Standard mechanical site preparation occurred (V-shearing of stumps and roots, subsoiling, and bedding at 6.1 m between rows) and pines were planted on beds 1–2 m apart. Residual woody debris was removed from the plot with an excavator to simulate a biomass harvest.

2.1.3. Pine and Switchgrass Intercropped with Residual Woody Debris in Place (Pine × Switchgrass, Biomass+, P × SB+). Site preparation and planting of pines followed the pine biomass + treatments. Switchgrass was machine planted in June 2009 in the alleys between rows of planted pines.

2.1.4. Pine and Switchgrass Intercropped with Residual Woody Debris Removed, (Pine × Switchgrass, Biomass–, P × SB–). Site preparation and planting of pines followed the pine biomass treatments. Residual woody debris was removed with an excavator to simulate a biomass harvest. Switchgrass was machine planted in June 2009 in the alleys between rows of planted pines.

2.1.5. Switchgrass Only, (S). Site preparation included excavation to remove residual woody debris, V-shearing, and root raking to establish a suitable planting surface. Switchgrass was machine-planted May–July 2009 across the entire plot.

Habitat structure of each treatment was assessed using two 30 m transect lines on each study plot (Figure 2) in October 2010. To estimate canopy coverage, vegetation, or woody debris that crossed transect lines and covered a minimum length of 10 cm along the line were measured using point interception methods [22, 23]. Percent coverage was calculated as the linear distance a habitat feature intersected the transect line divided by transect line length (30 m) and averaged across two transect lines per plot. Mean height of each habitat feature that intersected a transect line was estimated to the nearest 10 cm. Height was calculated as mean height of each habitat variable on a transect line, averaged across two transect lines on each study plot.

2.2. Rodent Live Trapping. We established trapping grids (30 m × 60 m) approximately 20 m from the edge of each study plot using four parallel trap lines with 10 m between traps. Each trap line was composed of six Sherman live traps (H.B. Sherman Traps Inc., Tallahassee, Florida, USA) and

one randomly assigned Longworth (Rogers Manufacturing Co., Peachland, British Columbia, Canada).

We set traps at sunset (1700–2030) and checked them at sunrise (0600–0830) for three consecutive nights (henceforth referred to as a trapping period). We baited with a mixture of sunflower seeds and rolled oats. We conducted 8 trapping periods from 15 July to 9 December, 2009, and 6 trapping periods from 19 July to 14 November, 2010. We marked all rodents with a unique numbered ear-tag (Monel Numeric size 1005-1; National Band and Tag Co, Newport, Kentucky, USA). Upon capture of each rodent, we recorded ear-tag number, species, sex, age-class, reproductive condition, and mass (g). We considered *P. leucopus* to be adults if they had completed their post-juvenile molt [24] and *S. hispidus* to be adults if they weighed >80 g [25]. We based age classes of other species on a combination of body mass and pelage characteristics. All rodent trapping, marking, and handling techniques were approved by the North Carolina Wildlife Resources Commission (Permit Numbers: 09-SC00162 and 10-SC00162) and the UNCG Institutional Animal Care and Use Committee (Protocol Numbers: 09-09 and 10-04).

We calculated community diversity at the plot scale with multiple metrics, including species richness (number of species captured in a plot/year), the Shannon Diversity Index, and the Fisher's α Diversity Index. We calculated Shannon Diversity Index as H' using number of unique individuals, for each species, captured in a year [26] using Ecosim7 [27]. We calculated Fisher's α Diversity Index for study plots using number of unique individuals, for all species, captured in a year as $S = a + \ln(I + (n/a))$ where S = species richness, n = number of unique individuals, and a = Fisher's α . We used Estimate S 8.2 [28] with 1000 runs, strong hash encryption, and randomized samples without replacement to determine Fisher's α Diversity Index.

We calculated population abundance and apparent survival for all adults in the populations using Program MARK [29]. We used Pollock's robust design model (Huggins closed capture estimator—see [30, 31]) to estimate population abundance during each trapping period and population survival during each intertrapping interval [32]. We used the Huggins close capture estimator because it is conditional on only animals captured and is, therefore, a more stable estimator for small sample sizes [30]. Data were grouped by species and year for analysis; the best model for each grouping was chosen based on AICc values.

In addition to examining community diversity and population abundance, we also examined sex ratio and recruitment because these demographic parameters can be influenced by habitat quality [19]. We determined sex ratio and recruitment using Program MARK. Adult sex ratio was calculated as number of adult males divided by number of adults in each population as estimated by Pollock's robust design models (Huggins closed capture estimator) with sex as a group variable. Recruitment (births and immigration) was calculated using a robust design Pradel survival and recruitment model, with Huggins closed captures estimator [33]. Due to small sample sizes when data were separated by species and plots, model selection was limited to those models with constant recruitment over time. Therefore, reported

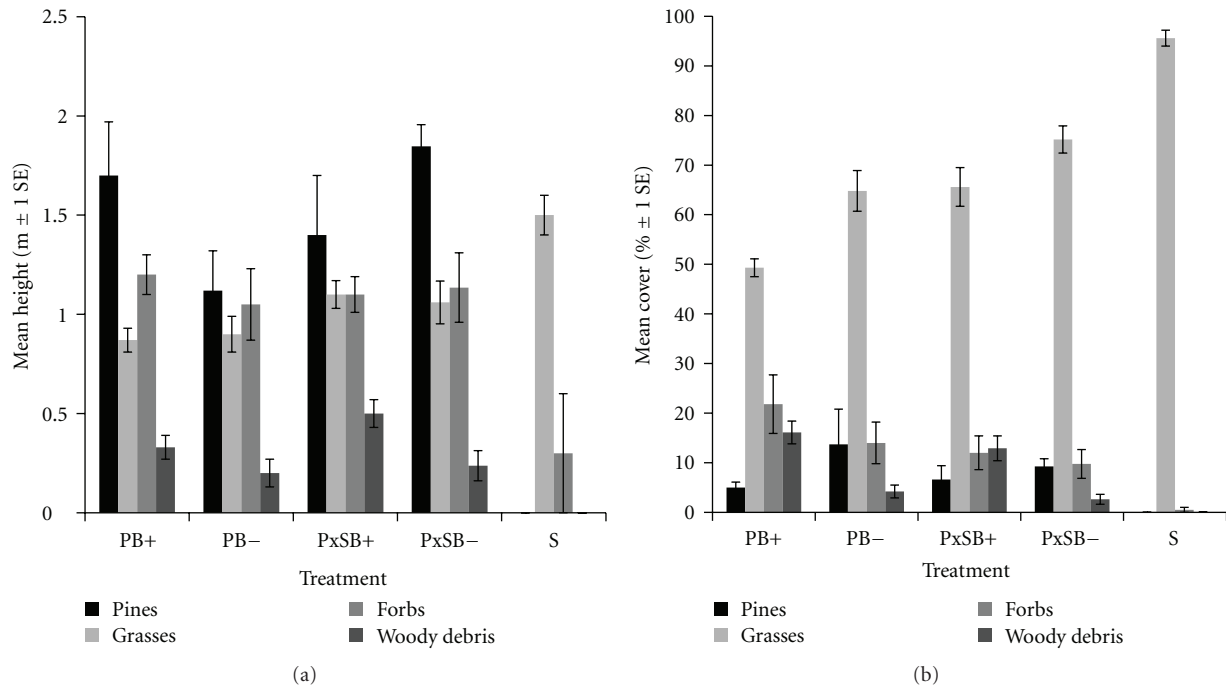


FIGURE 2: Mean (a) height (m; ± 1 standard error) and (b) cover (%; ± 1 standard error) of pines, grasses, forbs, and woody debris in treatment plots at the Lenoir 1 Sustainability Study Site, in Lenoir Co, NC. Mean percent cover and mean height represent the mean value of four replicate plots based on measurements taken in October 2010. Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass.

recruitment estimates represent recruitment per existing member for each plot. Parameterization of encounter probabilities (p and c) for each year-species combination matched the best fit model from the adult population abundance models.

2.3. Statistical Analyses. All dependent variables were calculated for each plot that represented an area of approximately 1,800 m². All data are presented as mean \pm 1 standard error unless otherwise noted. We tested variables for normality using a Kolmogorov-Smirnov test and homogeneity of variance using a Levene's test. Where appropriate, we transformed variables that violated tests for normality or homogeneity of variance with log (+1), rank, square root, or square root-arcsin transformations.

We tested the hypothesis of no differences among treatments in community metrics (i.e., richness, Shannon Diversity Index, Fisher's α Index), abundance, recruitment, survival, and sex ratio among treatments using analysis of variance (ANOVA) or a Kruskal-Wallis test when parametric assumptions were not met. We examined relationships among population abundance and treatments by rank transforming population abundance and using an ANOVA because this facilitated a repeated measures approach [34]. For the three rodent species for which we could calculate abundance, we used repeated measures ANOVA with treatment as the between-subject factor and trapping period as the within-subject factor. In cases where the repeated measures ANOVA tests did not meet assumptions of sphericity,

we used a Greenhouse-Geisser correction factor [35]. For posthoc comparisons among treatment groups, we used Tukey's analyses for ANOVA procedures and Mann-Whitney U pairwise comparisons for Kruskal-Wallis procedures. We used an alpha level of 0.05 for all statistical tests and we conducted analyses with SPSS 16.0 (SPSS 2007, Chicago, Ill, USA).

3. Results

In 2009, we trapped small mammals on 77 nights (15,366 trap nights), capturing 648 unique individual rodents (267 *Peromyscus leucopus*, 248 *Mus musculus*, 122 *Sigmodon hispidus*, and 11 *Reithrodontomys humulis*) 1,806 times. In 2010, we trapped on 54 nights (11,044 trap nights) capturing 1,634 unique individual rodents (1,030 *S. hispidus*, 310 *M. musculus*, 297 *P. leucopus*, 15 *R. humulis*, and 6 *Oryzomys palustris*) 3,594 times. In 2009 and 2010, treatment did not influence species richness or diversity (Table 1). Sample sizes were too small for analyses for *R. humulis* and *O. palustris*.

In 2009, there was a significant effect of trapping period on *M. musculus* population abundance ($F_{7,105} = 7.81$, $P < 0.001$) whereby a relatively small population of *M. musculus* in early trapping rounds increased through trapping periods 4 and 5 (Figure 3). In addition, there was a treatment effect on *M. musculus* where population abundance was less in PB+ plots than P \times SB- plots ($F_{4,15} = 4.29$, $P = 0.02$; Figure 4). *M. musculus* population abundance was greater in treatments that contained switchgrass than treatments

TABLE 1: Mean (± 1 SE) values of species richness, Shannon Diversity Index, and Fisher's α by treatment in 2009 and 2010. Rodents were captured and released on site at the Lenoir 1 Sustainability Study Site, in Lenoir Co., NC, during 15 July–9 December, 2009 and 19 July–14 November, 2010. Statistical results are from comparisons using a 1-way ANOVA or Kruskal-Wallis test when parametric assumptions were not met. Kruskal-Wallis test statistics are reported as χ^2 estimates. Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass.

Year	Treatment	Species Richness	Shannon Diversity Index	Fisher's α
2009	PB+	3.25 \pm 0.25	0.75 \pm 0.10	0.75 \pm 0.07
	PB-	2.75 \pm 0.25	0.62 \pm 0.19	0.63 \pm 0.08
	P \times SB+	3.25 \pm 0.48	0.92 \pm 0.11	0.72 \pm 0.13
	P \times SB-	3.25 \pm 0.25	0.99 \pm 0.09	0.72 \pm 0.09
	S	3.50 \pm 0.09	0.84 \pm 0.15	0.76 \pm 0.09
		$\chi^2 = 3.20, df = 4, P = 0.53$	$F_{4,19} = 1.20, P = 0.34$	$F_{4,19} = 0.31, P = 0.87$
2010	PB+	3.00 \pm 0.0	0.78 \pm 0.09	0.55 \pm 0.01
	PB-	3.50 \pm 0.29	0.91 \pm 0.07	0.66 \pm 0.07
	P \times SB+	3.50 \pm 0.29	0.86 \pm 0.14	0.65 \pm 0.08
	P \times SB-	3.75 \pm 0.25	0.76 \pm 0.15	0.69 \pm 0.07
	S	3.50 \pm 0.29	0.89 \pm 0.08	0.67 \pm 0.07
		$\chi^2 = 4.61, df = 4, P = 0.33$	$F_{4,19} = 0.40, P = 0.80$	$\chi^2 = 1.02, df = 4, P = 0.91$

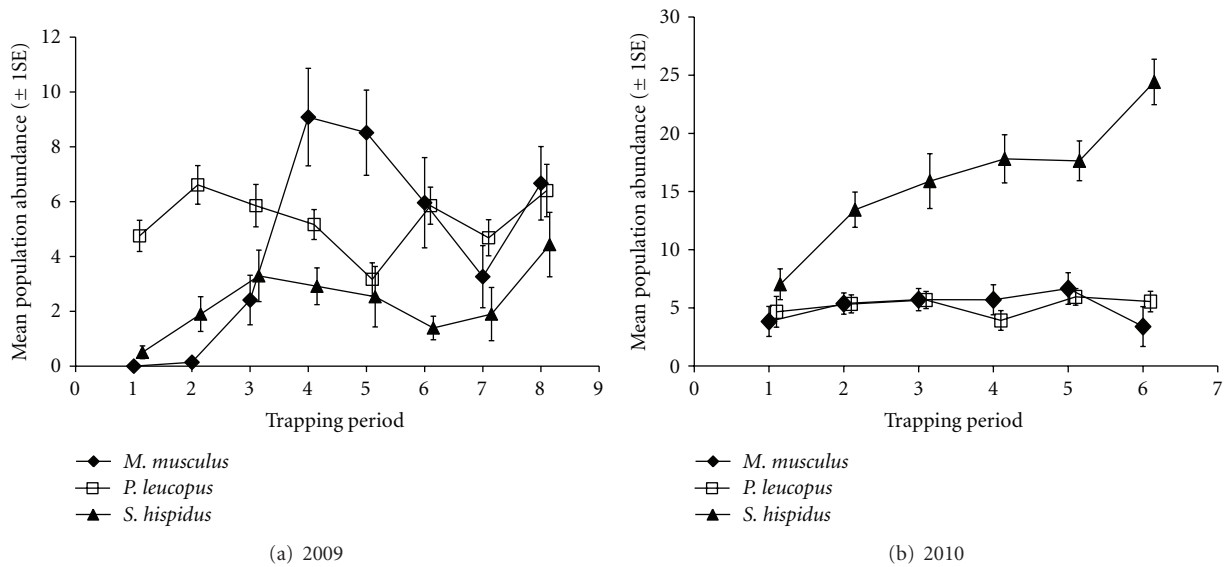


FIGURE 3: Population abundance (mean abundance ± 1 standard error; $n = 4$) for *Mus musculus*, *Peromyscus leucopus*, and *Sigmodon hispidus* by trapping period in (a) 2009 and (b) 2010 at the Lenoir 1 Sustainability Study Site, in Lenoir Co., NC. Axes vary between plot (a) and (b) to improve readability. Although not shown, 18 *Reithrodontomys humulius* were captured in 2009: 1 in period 2 (P2), 3 in P3, 2 in P4, 2 in P5, 5 in P6, 3 in P7, and 2 in P8. Sixteen *R. humulius* were captured in 2010 (3 in P3, 3 in P4, 3 in P5, and 7 in P6). Additionally, 6 *Oryzomys palustris* were captured in the last trapping periods of 2010. Rodents were captured and released during 15 July–9 December, 2009 and 19 July–14 November, 2010.

that did not, although not all pairwise comparisons were significant (Figure 4). *P. leucopus* population abundance did not differ among trapping periods ($F_{7,105} = 0.13, P = 1.00$; Figure 3) nor treatments ($F_{4,15} = 0.88, P = 0.50$; Figure 4). *S. hispidus* population abundance did not differ among

trapping periods ($F_{3,49,52.38} = 1.21, P = 0.32$; Figure 3) nor treatments ($F_{4,15} = 0.53, P = 0.72$; Figure 4).

Population abundances were not influenced by trapping period in 2010 (*M. musculus*: $F_{5,75} = 0.21, P = 0.96$; *P. leucopus*: $F_{2,73,40.90} = 0.03, P = 0.99$; *S. hispidus*: $F_{1,72,40.89} = 0.11,$

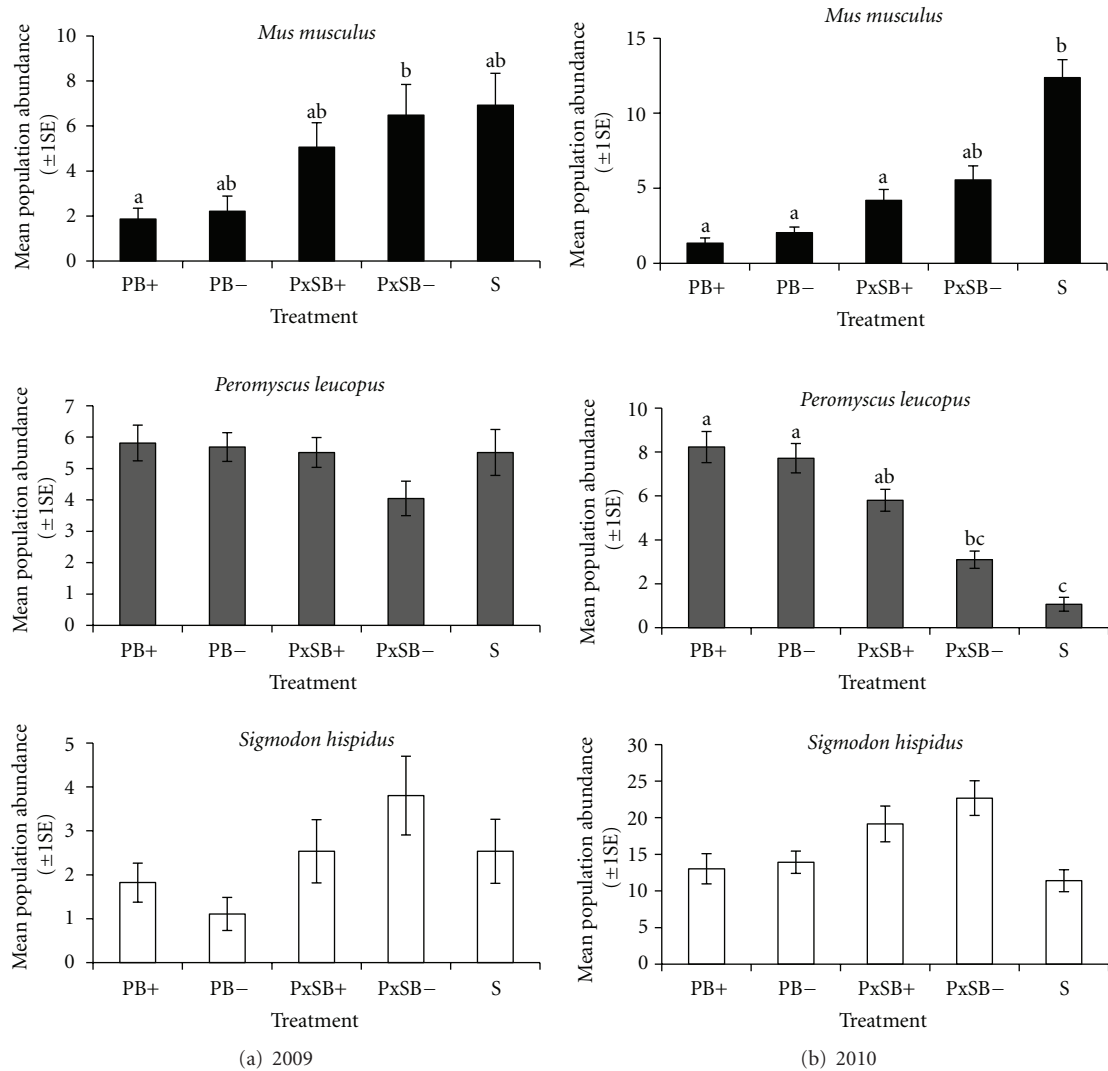


FIGURE 4: Population abundance estimates from Program MARK (mean number of adult individuals/plot, ± 1 standard error; $n = 4$) for *Mus musculus*, *Peromyscus leucopus*, and *Sigmodon hispidus* by treatment in (a) 2009 and (b) 2010 at the Lenoir 1 Sustainability Study Site, in Lenoir Co, NC. y -axes vary between plot (a) and (b) to improve readability. Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass. Treatments with different letters are significantly different ($P < 0.05$; Tukey's post hoc tests). Rodents were captured and released during 15 July–9 December, 2009 and 19 July–14 November, 2010.

$P = 0.94$; Figure 3). *M. musculus* population abundance was greatest in switchgrass only plots ($F_{4,15} = 8.25$, $P < 0.001$; Figure 4). In contrast, population abundance was the lowest for *P. leucopus* in switchgrass only plots and the greatest in plots without switchgrass (PB+ and PB-), although not all pairwise comparisons showed significant differences ($F_{4,15} = 13.61$, $P < 0.001$; Figure 4). *S. hispidus* population abundance did not differ among treatments ($F_{4,15} = 1.40$, $P = 0.28$; Figure 4).

There was no influence of treatment on recruitment of *M. musculus*, *P. leucopus*, or *S. hispidus* in either year (Table 2). Survival of *P. leucopus* was less in the pure switchgrass treatment than in treatments without switchgrass in 2010 (PB-, PB+; Table 3). Treatments also influenced sex ratio where there were fewer *M. musculus* males in PB+ than PB- treatments in 2010 (Table 4).

4. Discussion

Our results suggest that community composition of rodents did not change due to switchgrass intercropping or biomass removal treatments. However, there were species level responses to switchgrass whereby more *P. leucopus* were in plots without switchgrass and more *M. musculus* in plots with switchgrass. Additionally, decreased apparent survival of *P. leucopus* in switchgrass suggests that individual *P. leucopus* that did move into switchgrass was either less likely to survive or more likely to permanently emigrate from those treatments. This finding contrasts with higher apparent survival of *P. leucopus* in PB- and PB+ treatments (treatments without switchgrass), where they experienced either decreased mortality or decreased likelihood of permanent emigration from those treatments.

TABLE 2: Mean (\pm 1SE) recruitment estimates (new individual via birth or immigration/existing member/plot) by treatment (calculated in Program MARK). Rodents were captured and released on site at the Lenoir 1 Sustainability Study Site, in Lenoir Co., NC, during 15 July–9 December, 2009 and 19 July–14 November, 2010. Statistical results are from comparisons using a 1-way ANOVA or Kruskal-Wallis test when parametric assumptions were not met. Kruskal-Wallis test statistics are reported as χ^2 estimates. Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass.

Year	Treatment	<i>M. musculus</i>	<i>P. leucopus</i>	<i>S. hispidus</i>
2009	PB+	0.24 \pm 0.14	0.03 \pm 0.01	0.10 \pm 0.07
	PB-	0.44 \pm 0.15	0.03 \pm 0.00	0.73 \pm 0.25
	P \times SB+	0.27 \pm 0.19	0.03 \pm 0.00	0.06 \pm 0.23
	P \times SB-	0.10 \pm 0.01	0.03 \pm 0.01	0.44 \pm 0.23
	S	0.20 \pm 0.14	0.03 \pm 0.01	0.79 \pm 0.77
		$F_{4,19} = 0.80, P = 0.54$	$F_{4,19} = 0.19, P = 0.94$	$\chi^2 = 4.14, df = 4, P = 0.39$
2010	PB+	0.22 \pm 0.16	0.02 \pm 0.00	0.05 \pm 0.01
	PB-	0.22 \pm 0.12	0.03 \pm 0.00	0.04 \pm 0.00
	P \times SB+	0.07 \pm 0.02	0.03 \pm 0.00	0.04 \pm 0.01
	P \times SB-	0.20 \pm 0.11	0.21 \pm 0.18	0.03 \pm 0.00
	S	0.06 \pm 0.01	0.24 \pm 0.19	0.05 \pm 0.01
		$\chi^2 = 3.64, df = 4, P = 0.46$	$\chi^2 = 5.73, df = 4, P = 0.22$	$\chi^2 = 3.96, df = 4, P = 0.41$

TABLE 3: Mean (\pm 1SE) survival estimates for intertrapping period intervals by treatment (averaged across time and plots within treatment; calculated in Program MARK). Rodents were captured and released on site at the Lenoir 1 Sustainability Study Site, in Lenoir Co., NC, during 15 July–9 December, 2009 and 19 July–14 November, 2010. Statistical results are from comparisons using a 1-way ANOVA or Kruskal-Wallis test when parametric assumptions were not met. Kruskal-Wallis test statistics are reported as χ^2 estimates. Within a year-species pairing, means with different letters are significantly different from one another (Tukey's post hoc tests). Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass.

Year	Treatment	<i>M. musculus</i>	<i>P. leucopus</i>	<i>S. hispidus</i>
2009	PB+	0.46 \pm 0.03	0.93 \pm 0.04	0.57 \pm 0.06
	PB-	0.51 \pm 0.04	0.92 \pm 0.04	0.39 \pm 0.03
	P \times SB+	0.56 \pm 0.05	0.92 \pm 0.04	0.53 \pm 0.06
	P \times SB-	0.57 \pm 0.05	0.93 \pm 0.03	0.53 \pm 0.05
	S	0.66 \pm 0.05	0.87 \pm 0.05	0.60 \pm 0.06
		$F_{4,15} = 0.85, P = 0.51$	$F_{4,15} = 0.14, P = 0.97$	$F_{4,12} = 1.29, P = 0.33$
2010	PB+	0.49 \pm 0.07	0.98 \pm 0.00 ^a	0.93 \pm 0.04
	PB-	0.50 \pm 0.06	0.99 \pm 0.00 ^a	0.95 \pm 0.03
	P \times SB+	0.67 \pm 0.07	0.93 \pm 0.04 ^a	0.96 \pm 0.03
	P \times SB-	0.59 \pm 0.07	0.75 \pm 0.07 ^{ab}	0.99 \pm 0.00
	S	0.81 \pm 0.06	0.56 \pm 0.06 ^b	0.97 \pm 0.01
		$F_{4,14} = 1.02, P = 0.43$	$F_{4,15} = 5.30, P = 0.007$	$F_{4,15} = 1.28, P = 0.32$

Rodent species captured in this study were expected to be found inhabiting pine forests in the southeast [15, 17, 36, 37]. *Peromyscus leucopus*, *S. hispidus*, *R. humulis*, and *M. musculus* were the only species captured in study plots in the first year of study and were >99% of the individuals captured in the second year of study (6 *O. palustris* were captured during November 2010).

The most abundant species captured shifted from *P. leucopus* in 2009 to *S. hispidus* in 2010. Both years of the study experienced similar weather, and there were no discrete weather events that would explain the patterns of population abundance we saw between years. Rather, shifts in species relative population abundance between years is likely due to successional changes, and has been observed in other

studies of young pine plantations in the southeastern USA, and in secondary succession of both forest and grassland habitat types [36, 38, 39]. *Sigmodon hispidus* prefers grassy understory [40, 41], which was not available on study plots until 2010. This shift from *P. leucopus* to *S. hispidus* being the most abundant species suggests the rodent community responded to changes that occurred in habitat structure between site preparation at the beginning of year one and treatment establishment.

Our treatments varied along two resource axes: residual pine biomass (i.e., coarse woody debris; CWD) and switchgrass presence. Our results suggest that retention of biomass did not influence rodents within the first two years postsite preparation in this system because, overall, treatments with

TABLE 4: Mean (± 1 SE) values of sex ratio (proportion males) from adult population abundance estimates calculated in Program MARK. Rodents were captured and released on site at the Lenoir 1 Sustainability Study Site, in Lenoir Co., NC, during 15 July–9 December, 2009 and 19 July–14 November, 2010. Statistical results are from comparisons using a 1-way ANOVA or Kruskal-Wallis test when parametric assumptions were not met. Kruskal-Wallis test statistics are reported as χ^2 estimates. Within a year-species pairing, means with different letters are significantly different from one another (Mann-Whitney U pairwise comparisons with Bonferroni correction). Treatments: PB+ = Pine Biomass+; PB- = Pine Biomass-; P \times SB+ = Pine \times Switchgrass Biomass+; P \times SB- = Pine \times Switchgrass Biomass-; S = Switchgrass.

Year	Treatment	<i>M. musculus</i>	<i>P. leucopus</i>	<i>S. hispidus</i>
2009	PB+	0.72 \pm 0.09	0.65 \pm 0.04	0.49 \pm 0.12
	PB-	0.57 \pm 0.11	0.57 \pm 0.05	0.44 \pm 0.12
	P \times SB+	0.65 \pm 0.08	0.60 \pm 0.04	0.58 \pm 0.09
	P \times SB-	0.61 \pm 0.07	0.61 \pm 0.06	0.38 \pm 0.19
	S	0.52 \pm 0.08	0.62 \pm 0.04	0.74 \pm 0.09
		$F_{4,19} = 0.35, P = 0.84$	$\chi^2 = 0.27, df = 4, P = 0.99$	$F_{4,19} = 0.63, P = 0.65$
2010	PB+	0.44 \pm 0.12 ^a	0.62 \pm 0.04	0.63 \pm 0.04
	PB-	0.81 \pm 0.08 ^b	0.55 \pm 0.03	0.55 \pm 0.04
	P \times SB+	0.40 \pm 0.09 ^{ab}	0.79 \pm 0.04	0.43 \pm 0.04
	P \times SB-	0.60 \pm 0.06 ^{ab}	0.60 \pm 0.07	0.43 \pm 0.05
	S	0.64 \pm 0.04 ^{ab}	0.42 \pm 0.13	0.61 \pm 0.05
		$\chi^2 = 11.03, df = 4, P = 0.026$	$F_{4,19} = 0.96, P = 0.46$	$F_{4,19} = 2.02, P = 0.14$

biomass removal had similar rodent abundances compared to treatments without biomass removal. A recent meta-analysis indicated little or no consistent response of mammal diversity to CWD and suggests that small mammal response to CWD is very context dependent. We were not able to find a relationship between small mammal metrics and removal of biomass in this study, which is consistent with other studies in the southeastern USA. At this time, it is unclear if a greater range of removal levels or a greater range in CWD amounts among treatments would have caused a stronger small mammal response [42]. Although biomass provides important habitat structure (i.e., nest sites, foraging substrate, and cover from predators) for rodents, it is likely that other factors such as presence of switchgrass (i.e., a potential source of food and cover) exerted a greater influence during early succession in this managed pine forest.

Presence of switchgrass changed habitat structure and resources available to the rodent community in young managed pine stands during the first two years of growth [43]. Differences in habitat structure among treatments with and without switchgrass were reflected in rodent population responses within the first year of study and became more pronounced in the second year of the study. While there was no effect of treatment on sex ratio or recruitment, population abundance and survival data suggest that *P. leucopus* was negatively affected by switchgrass presence. During the second year of study, *P. leucopus* adults were found in greatest abundance in PB+ and PB- treatments (nonswitchgrass treatments) and had greater apparent survival (mortality and permanent emigration) in these plots. Our results suggest that individual *P. leucopus* in plots with switchgrass (P \times SB- and S) were either less likely to survive or more likely to permanently emigrate from the habitat. In contrast, *P. leucopus* in PB+ and PB- habitat experienced either decreased mortality or decreased likelihood of permanent emigration from the habitat. *Peromyscus leucopus* is typically

an early successional species that is more abundant in open habitat types than areas with greater understory foliage cover [36, 44]. In our study, plots with switchgrass contained dense switchgrass cover. Our results suggest that switchgrass-dominated understories may not provide sufficient resources to support *P. leucopus* and/or that *P. leucopus* may be outcompeted by *M. musculus* in these treatments. However, additional research is needed to determine if this effect is temporary, if it changes with further stand development, and the causal mechanisms of this effect.

In contrast to our findings for *P. leucopus*, the population abundance of *M. musculus* was positively affected by switchgrass. During the first year of study, there were more *M. musculus* in the P \times SB- and S plots than the other plots, and by the second year of study, there were over twice as many adults captured in switchgrass only plots than any other treatment. Additionally, *M. musculus* abundance in the intercropping treatments was intermediate to either S or PB+/PB- treatments. However, there was no effect of treatment on the apparent survival or recruitment (*per capita* births and immigration) of *M. musculus*. Therefore, while more *M. musculus* individuals were found in switchgrass treatments, the proportional population dynamics were similar among treatments—*M. musculus* were not more or less likely to survive, reproduce, immigrate or emigrate from any treatment. Yet in the second year, the adult populations of *M. musculus* in PB- were male dominated, the populations in PB+ had even sex ratios and the switchgrass treatments were intermediate between the two pine treatments. These findings suggest that female *M. musculus* may have been differentially affected by the presence of biomass in their less preferred habitat.

Mus musculus is an introduced, invasive species that can increase in abundance rapidly when required resources become available [44]. In the first year of our study, *M. musculus* abundance increased across trapping periods from

July–December 2009. Appearance of *M. musculus* in study plots in late August 2009 occurred simultaneously with harvest of corn (*Zea mays*) fields located adjacent to the north and east borders of our study area (Figure 1). Based on initial captures, abundance of *M. musculus* adults increased, suggesting an initial immigration event occurred on the study site from adjacent agricultural fields. Additional support for the immigration of *M. musculus* onto the study site from agricultural fields comes from our trapping data that shows individuals were first captured at trap stations close to the adjacent corn fields.

There was no treatment effect on population abundance of *S. hispidus* in either year of our study. Additionally, there was no treatment effect on *S. hispidus* recruitment, apparent survival, or sex ratio. However, population abundances in the second year of our study were approximately four to ten times greater than the population abundances in the same treatments in the first year of our study, and while not significant, *S. hispidus* population abundances were numerically greatest in P × SB– plots during both years. *S. hispidus* is associated with early successional or disturbed habitat types that have greater understory cover [17]. Furthermore, *S. hispidus* can out-compete other rodent species for resources in habitat types with greater understory foliage cover [36, 45, 46]. Our results suggest that in more complex habitat, and as habitat heterogeneity increased from the first to second year of the study, *S. hispidus* was able to out-compete other species and become dominant. However, understanding effects of competition versus successional change in community structure would require additional studies.

During both years of our study, few *R. humulis* were captured, which was unsurprising given that it is a relatively rare species in pine forests [36]. Nevertheless, we captured *R. humulis* individuals in all treatment options at low numbers suggesting that for at least two years postpreparation, presence of switchgrass does not influence abundance of this species relative to nonswitchgrass plots. This result was surprising because we expected *R. humulis* to increase in presence of switchgrass due to increased availability of seeds during autumn and granivorous habits of *R. humulis*. Lack of response of *R. humulis* abundance suggests this species does not respond rapidly to changes in resource abundance or may be locally rare in managed pine forests in our study area. Data from other studies in the coastal plain suggest the former; *R. humulis* are among the most commonly captured species in the southeastern coastal plain [17, 37, 46] but are captured at relatively low rates after initial stand establishment [37].

Our results suggest either that *P. leucopus* is more competitive in plots with pine and *M. musculus* is more competitive in plots with switchgrass or that switchgrass does not provide suitable resources for *P. leucopus* and plots with pine do not provide suitable resources for *M. musculus*. Evidence from studies of mammalian diversity in pine plantations suggests that, although *M. musculus* is present, it is not usually the most abundant small mammal species present (e.g., [36, 45]). However, the potential mechanisms underlying our observed patterns, especially the clearly opposite responses of *P. leucopus* and *M. musculus* to switchgrass treatments,

require further investigation. Field experiments involving competitor exclusion or limited-resource provision could be used to determine ecological mechanisms underlying the responses of *P. leucopus* and *M. musculus* to the presence of switchgrass in the future.

5. Implications

(1) Our results suggest residual biomass removal has no influence on rodent community structure or population demographics in an intensively managed, coastal plain southern pine system.

(2) In our study, switchgrass has a positive influence on the invasive *M. musculus* and a negative influence on the native *P. leucopus*. Moreover, intercropping switchgrass with pine does not appear to completely offset effects of switchgrass, at least for *P. leucopus*. However, other native species, such as *S. hispidus*, were not influenced by presence of switchgrass. In the short term, switchgrass can affect abundance of native and invasive rodent species. Determining ecological effects underlying associations is important for long-term implications.

Acknowledgments

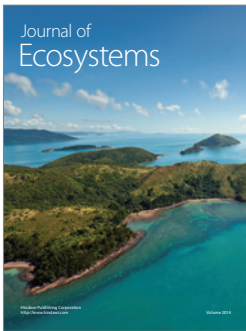
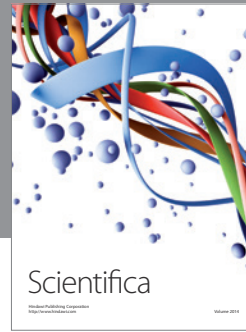
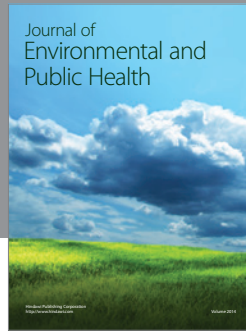
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