



Management and Conservation Note

Regional-Scale Assessment of Deer Impacts on Vegetation Within Western Connecticut, USA

ANGELA C. RUTHERFORD, *School of Forestry and Environmental Studies, Yale University, 370 Prospect Street, New Haven, CT 06511, USA*

OSWALD J. SCHMITZ,¹ *School of Forestry and Environmental Studies, Yale University, 370 Prospect Street, New Haven, CT 06511, USA*

ABSTRACT High densities of white-tailed deer (*Odocoileus virginianus*) are believed to cause broad-scale forest regeneration failure and loss of plant diversity. But, the empirical basis for such presumptions is limited. We, therefore, conducted a survey in western Connecticut, USA, woodlots to examine how spatial variation in deer densities influences variation in impacts on plant species abundance, identity and diversity, and tree regeneration. We also used a Geographic Information System to quantify trends between land-cover type and deer density. Deer density was not correlated with any vegetation or land use variable. This suggests that deer density is not a leading factor determining variation in vegetation impacts across western Connecticut.

KEY WORDS Connecticut, deer damage, deer–human interactions, deer overabundance, deer population control, habitat fragmentation, *Odocoileus virginianus*, plant diversity, white-tailed deer, woodlot regeneration.

White-tailed deer (*Odocoileus virginianus*) populations in the northeastern United States have increased from near-extirpation around the turn of the 20th century to >25 deer/km² today (DeCalesta and Stout 1997, Russell et al. 2001, Horsley et al. 2003). Such dense populations, and inevitable interactions between deer and humans, have precipitated numerous calls for management to address what is now widely viewed as a problem of deer overabundance (McShea and Rappole 2000, Russell et al. 2001, Cote et al. 2004, Levy 2006).

Calls to manage the overabundance problem stem from observations that where deer are highly abundant there are also high incidences of several environmental impacts such as forest regeneration failure, loss of biodiversity, invasion by exotic species, Lyme disease risk, and deer–vehicle collisions (DeCalesta 1994, McShea and Rappole 2000, Levy 2006, Kilpatrick and LaBonte 2007). Yet, such observations of environmental impact may be conflated by contemporaneous human alteration of landscapes because rising human populations have changed local land-use regimes throughout a region (Bashore et al. 1985, Hubbard et al. 2000, Allan et al. 2003, Brownstein et al. 2005). Thus, conclusions about overabundance are often based on human values in relation to preferences for particular environmental states (Sinclair 1997, Russell et al. 2001).

Because local values and deer impacts may be conflated, a general policy to institute density reductions can only be scientifically justified if deer abundance is shown to be the leading factor determining environmental impacts among localities within a region (Schmitz and Sinclair 1997, Russell et al. 2001, Cote et al. 2004). Such regional-scale determinations of deer impact have, however, rarely been done (Russell et al. 2001, Cote et al. 2004). We, therefore, conducted an assessment of deer impacts across western Connecticut, USA, a geographic region that encompassed a broad range of deer densities, land-use types, and human

settlements (Fig. 1) and in which there was a perceived deer overabundance problem (Kilpatrick and LeBonte 2007).

We used 2 complementary approaches to assess whether deer abundance is a leading factor determining environmental impacts across this region. We tested for potential deer impacts on woodlot plant diversity and regeneration by conducting an intensive field survey across western Connecticut (Fig. 1d). We also evaluated whether deer densities varied with the nature of land use, especially in relation to important determinants of deer presence like proportion of developed habitat and the degree of habitat fragmentation (Cote et al. 2004, Brownstein et al. 2005).

If deer density is the leading factor driving environmental impacts, then across western Connecticut, variation in deer density should explain a high amount of variation in the magnitude of environmental impacts. Nevertheless, deer abundances in this region (Fig. 1b) are below minimum levels (on the order of 14–15 deer/km²) for which significant damaging impacts on plant species abundance and diversity have been reported (Russell et al. 2001). We, therefore, did not expect to see a strong relationship between variation in deer density and variation in environmental impact across western Connecticut. We hypothesized further that if transformation of the landscape by human land-use patterns determines deer density, then across this geographic region, variation in land use should explain variation in deer densities.

STUDY AREA

Our study focused on an 8,261-km² area of western Connecticut, USA, between Long Island Sound in the south and the state of Massachusetts in the north (Fig. 1). Developed land was concentrated within an urban belt along the south and eastern side of the region and became more rural and forested toward the northwest corner of the state (Fig. 1a). Deer population densities varied from an average of 11.5 deer/km² in urbanized southeastern areas to 1.4 deer/km² in rural parts (Fig. 1b). The degree of land

¹ E-mail: oswald.schmitz@yale.edu

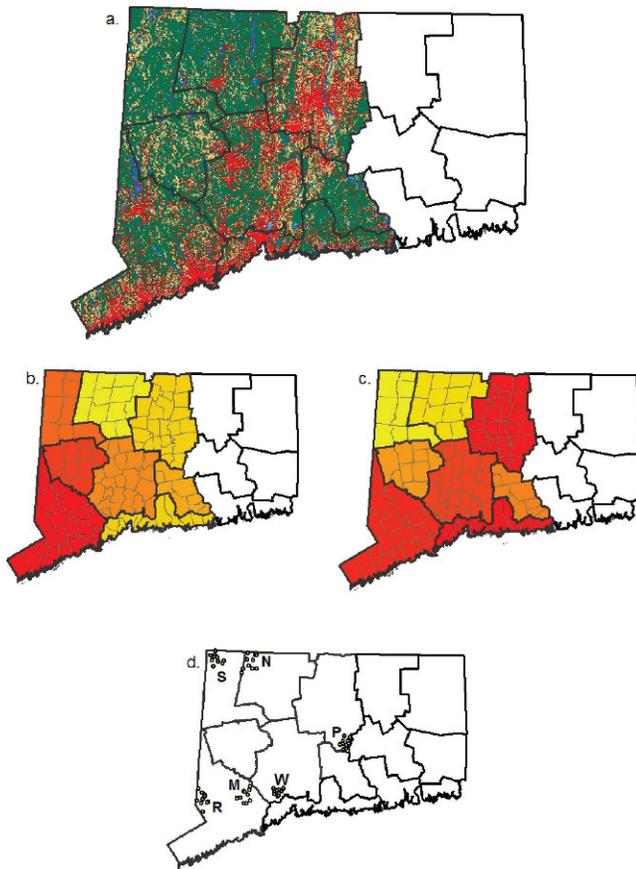


Figure 1. Study region of western Connecticut, USA, separated by Connecticut Department of Environmental Protection Hunting Management Zone. The region presents spatial variability in land use, deer density, and human population size to assess regional deer impacts on vegetation. (a) land use patterns (red = developed, brown = field, green = forest, blue = water); (b) deer density (deer/km²) ■ = 11.5, ■ = 8.8, ■ = 7.9, ■ = 5.6, ■ = 2.8, ■ = 2.0, ■ = 1.4, ■ = 1.1; (c) human population density (persons/km²) ■ = 60.9, ■ = 59.1, ■ = 54.5, ■ = 54.1, ■ = 53.0, ■ = 36.0, ■ = 19.3, ■ = 9.3; (d) sampling sites: Woodbridge (W), Monroe (M), Salisbury (S), Ridgefield (R), Norfolk (N), and Portland (P). Maps are based on data current to 2007.

development corresponded to human population density, which was highest within the urban belt at approximately 23 persons/km² and decreased toward the northwestern corner to <10 persons/km² (Fig. 1c). Western Connecticut thus offered a range of environmental conditions needed to assess how variation in deer density, human land-use patterns, and environmental impacts might be related.

METHODS

We sampled in 6 replicate towns across the western Connecticut study area (Woodbridge, Monroe, Ridgefield, Salisbury, Norfolk, and Portland), which ranged from low to high deer densities based on the Connecticut Department of Environmental Protection (CT DEP) aerial survey data (Fig. 1b). Within each of these 6 towns, we randomly identified 20 potential sampling locations using Geographic Information Systems (GIS) and then sampled a subset of 10, giving 60 sampling plots statewide (Fig. 1d). Given our focus on woodlot plant diversity and regeneration, we deemed a sampling site as suitable if it had sufficient

nonlandscaped woodlot, defined as a small (≤ 5 ha) forested patch of land within an urban or agricultural land matrix, to accommodate a 20 × 20-m sampling plot. Our final choice of 10 sampling locations was determined by property owner permission to access the location.

We conducted vegetation surveys within each of the 60 plot locations between 7 June and 11 July 2007. At each sampling location we flagged a 20 × 20-m plot. This plot size was small enough to fit into neighborhood study locations, yet large enough to include a representative number of plant species (Condit et al. 1996, Healy 1997, Ladd et al. 2005, Goslee 2006). We then located 1 × 1-m subplots in each corner and in the center of the 20 × 20-m plot. In each of the 5 subplots, we identified each plant to species, designated it as woody or herbaceous and native, invasive, or ornamental. We estimated the percent of the 1-m² subplot area covered by each herbaceous and woody seedling species that was within 1 m above ground, and we counted the number of trees (saplings and canopy trees) taller than 1 m. We estimated plant species diversity within 1 m above the ground in each subplot using richness and evenness metrics because deer can affect both the number of plant species in a community (richness) and the relative dominance of particular plant species (evenness) via selective foraging (Russell et al. 2001, Cote et al. 2004). We calculated evenness using the standard Shannon index (Krebs 1998). An evenness value of zero would indicate that one or a few species dominate the plot; an evenness value approaching 1 would indicate that all species in a plot have similar relative abundance. We averaged the subplot abundance and diversity data to obtain an independent estimate of each variable for each 20 × 20-m sampling plot.

Deer impacts vary with plant species composition and palatability among sites (Russell et al. 2001, Cote et al. 2004). We categorized all plant species as palatable or unpalatable based on lists assembled by official wildlife extension services (Burroughs and Dudek 2008, Bridgen 2009, Morton Arboretum 2009, Rutgers University 2009). If a species did not make any list, we categorized it as unknown. We calculated the proportion of species within each plot that was palatable. We also estimated the degree to which plant species composition was common among all town sites using pair-wise comparisons. We divided the number of species that 2 towns shared by the total number of species in each of the 2 towns.

We assessed regeneration by measuring woody vegetation (tree seedling and sapling) density (% cover) up a 2-m vertical profile within each subplot location. We estimated percent cover at 4 heights (0–0.5 m, 0.5–1.0 m, 1.0–1.5 m, 1.5–2.0 m) using a vertical cover-board technique (Nudds 1977, McShea and Rappole 2000). We quantified an index of woody vegetation regeneration in a plot as foliage height diversity (FHD) where $FHD = -\sum p_i \ln(p_i)$, and p_i is average vegetation density (% cover) of the 5 subplots at height i , and the summation was over the 4 height categories. Our rationale was that if an area is regenerating in all age classes uniformly, then woody vegetation density should be equal at all heights along the vertical cover-board

gradient. Lack of consistent regeneration would be indicated by little or no woody vegetation density along the gradient (McShea and Rappole 2000). Accordingly, a high FHD value indicates that habitat was regenerating throughout the 2-m understory; a low FHD indicates that regeneration was uneven. We averaged the subplot FHD data to obtain an independent estimate for each 20 × 20-m sampling plot.

We estimated relative deer density at each sampling location along 4 2-m-wide transects beginning at the center of each 20 × 20-m plot and radiating 50 m outward from the 4 sides of the plot. We measured 3 indices of deer density. We counted the number of woody stems browsed by deer relative to available stems up to 2 m high within 1 × 1-m plots spaced every 10 m along the transect. Stems could be browsed either by rabbits (*Sylvilagus floridanus*) or by deer. We attributed browsing to rabbits wherever there was an obvious 45° sharp angular clip. We attributed it to deer wherever there was an obvious sign of twig removal (no apical bud and remnants of bark strips) along an erect stem. We counted deer pellet groups (fresh and old) comprised of ≥13 pellets and normalized counts by the sampling area they covered (400 m² over all 4 transects) to yield an estimate of pellet group density (Grund et al. 2002, Thompson and Sharpe 2005). We counted the number of independent deer tracks crossing each transect. Because the ground in the woodlots was typically moist, we could readily observe hoof impressions on the soil surface. We conducted correlation analysis among the 3 indices of density for each plot to determine whether those indices were congruent.

We also verified whether our original designation of a south-to-north deer density gradient across Connecticut, based on CT DEP aerial survey data, was indeed reflected in our sampling by correlating our measures with the CT DEP data (Gregonis 2007). The CT DEP data provide one estimate for each management zone (Fig. 1b), whereas our sampling provided several estimates within each zone (Fig. 1d). We, therefore, used a randomization procedure to generate single, independent town-level data for comparison with management zone data. We took 5 random deer density estimates from the 10 density estimates within each of the 6 town sites and averaged them for each town. We then conducted a correlation analysis between our average town-level estimate and the respective CT DEP hunting management zone estimate. We repeated this algorithm 10 times for each of the 3 indices of deer density (browse, tracks, and pellets).

We used a general linear model procedure to examine relationships between our sampled indices of deer density (independent variable) and vegetation variables (i.e., average vegetative cover, plant species richness and evenness, FHD, abundance of ornamental and native species). Broad-scale geographic analyses run the risk of obscuring important local scale trends if qualitative differences in the strength of the relationship between deer density and impact exist among locations. We, therefore, conducted our analyses of deer impact at 2 scales. We examined relationships between plot-level deer density and plot-level impact for each of the 6 towns individually. We then examined the overall geo-

Table 1. Mean percentage similarity in woody and herbaceous plant species among 6 towns across a range of deer densities in western Connecticut, USA. Estimates are based on plant species sampling within 10 plots in each town during summer 2007.

Town	Town				
	Monroe	Ridgefield	Salisbury	Norfolk	Portland
Woodbridge	38.1	37.8	25	24	27
Monroe		38.4	30.5	31.1	41.3
Ridgefield			34.1	27.5	37.8
Salisbury				39.6	34.4
Norfolk					33.7

graphic-scale relationship for all towns combined. At this scale, the appropriate regression model includes all the deer density data as an independent variable, but also treats town as a random effect (block) to account for potential correlation structure of the town-level sampling data owing to multiple plots in each town. We also conducted a correlation analysis between the proportion of palatable species within a plot and vegetation variables in a plot using data for all 60 plots. We normalized all proportion data using a square-root transformation.

Using GIS, we quantified land-use attributes within a 1-km-radius buffered circle around each sampling location. We used 2008 land cover data from the University of Connecticut's Center for Land use Education And Research (CLEAR 2008) land-use cover ArcGIS polygon shapefile. Because deer thrive in fragmented landscapes, we estimated degree of habitat fragmentation around each sampling location by calculating the perimeter to area ratio of each polygon within the 1-km-radius buffered circle (Cote et al. 2004, Brownstein et al. 2005). We also calculated the proportion of land within the buffered-circle classified by CLEAR as developed, field, and forested. Here, the developed class represents the urban built environment; field includes barren lands, nonforested wetlands, turf, grass, and agriculture; and forested includes coniferous forest, deciduous forest, and forested wetland. We conducted regression analyses to examine effects of local habitat fragmentation and proportion of land use of developed, field, or forest on our estimates of local deer density.

RESULTS

We sampled 157 plant species (119 herbaceous, 38 woody) across the entire geographic region. In any one town, there were 15–28 woody species (40–74% of total woody species) and 34–51 herbaceous species (28–43% of total herbaceous species). Typically, there was 33% overlap in species composition among towns (Table 1). Density of saplings and trees that were part of the canopy (dbh ≥ 10 cm) and seedlings (woody stems growing within 1 m above ground) were similar among towns except for Salisbury, which had double the typical sapling density (Table 2).

We found significant correlations between our sampled indices of deer density among the 60 sampling sites (all $P < 0.01$). Deer track counts correlated poorly with the pellet group index and browse index (both $r < 0.45$). Pellet counts and browse indices were better correlated ($r = 0.68$).

Table 2. Density (stems/400-m² plot) of trees (saplings and canopy trees >10 cm dbh) and woody seedlings within sampling sites in 6 towns across a range of deer densities in western Connecticut, USA, in summer 2007.

Town	Trees		Woody seedlings		n
	\bar{x}	SE	\bar{x}	SE	
Woodbridge	24.9	6.6	46.1	8.6	10
Monroe	28.7	2.8	44.4	6.8	10
Ridgefield	24.2	3.9	49.1	15.4	10
Salisbury	23.5	4.8	93.1	23.4	10
Norfolk	24.8	2.2	42.9	6.9	10
Portland	20.4	2.3	58.5	11.8	10

(Rabbit browsing accounted for <1% of impact on tree seedlings and saplings.) Randomization test comparisons of our sampled density indices with CT DEP density estimates revealed that each of the 10 replications produced statistically significant ($P < 0.05$) results, indicating that statistical power of our conclusion was high. Our conclusion was that there was a correlation between each of our indices of deer density (based on browse impact, pellet groups, and track counts) and those reported by the CT DEP (all $r \geq 0.659$), thus affirming that we did indeed sample along a deer density gradient. We found the highest average correlation between browse impact data and CT DEP data (mean $r = 0.776$, mean $P < 0.05$). We, therefore, used average browse impact for each sampling plot as an independent measure of deer density in our subsequent analyses.

Our index of deer density and values of plant variables varied considerably within towns and across the study region (Table 3). For example, all towns had similar FHD (index of forest regeneration) values (0.2–0.6) but had highly variable indices of deer abundance (Fig. 2). In general, we found no significant town (block) effect on any of the regressions (all $P > 0.35$, $df = 5, 53$). There was also no correlation among deer density and any of the plant variables within specific locations and across the entire geographic region (Table 3).

The proportion of palatable plants ranged from an average 5% in Woodbridge to 8% in Monroe. Because of small within- and among-site variation in palatability there was no correlation between plant palatability and deer density among locations ($r = 0.07$, $P > 0.60$). Accordingly, there was no correlation (all $r < 0.02$, $P > 0.18$) between proportion of palatable plants among sampling locations and either of plant percent cover, plant species richness, plant species evenness, FHD, percentage ornamental plants, or percentage native plants. Given our sampling effort and error structure of the data, we estimated that statistical power of our results ranged from 75% to 90% (i.e., the likelihood of falsely concluding that there is no relationship between any 2 variables ranged between 10% and 25%). Neither deer density and degree of habitat fragmentation, nor deer density and land use allocation, were correlated (Table 3).

DISCUSSION

Our analysis of the relationship between deer abundance and impact across western Connecticut tested whether deer

Table 3. Relationship between measured index of deer density and measured vegetative variables or land use type. Variables were measured at 10 sampling sites within each of 6 towns across a range of deer densities in western Connecticut, USA, in summer 2007.

Deer density vs.	All sites ^a		Salisbury		Norfolk		Portland		Ridgefield		Monroe		Woodbridge				
	R ²	P-value	R ²	P-value	n	R ²	P-value	n	R ²	P-value	n	R ²	P-value	n			
% vegetative cover	0.05	0.72	0.30	0.11	10	0.11	0.35	10	0.01	0.76	10	0.12	0.34	10	0.01	0.79	10
Species richness	0.03	0.81	0.03	0.28	10	0.28	0.11	10	0.14	0.28	10	0.11	0.34	10	0.06	0.48	10
Species evenness	0.28	0.89	0.15	0.27	10	0.27	0.13	10	0.12	0.33	10	0.01	0.78	10	0.45	0.03	10
Ornamentals (%)	0.02	0.69	0.05	0.55	10	0.11	0.36	10	0.01	0.77	10	0.14	0.29	10	0.01	0.91	10
No. native plants	0.02	0.33	0.12	0.32	10	0.04	0.58	10	0.08	0.44	10	0.06	0.51	10	0.02	0.71	10
% native plants	0.00	0.99	0.22	0.17	10	0.05	0.53	10	0.18	0.23	10	0.09	0.39	10	0.02	0.70	10
Tree density	0.00	0.88	0.23	0.17	10	0.19	0.21	10	0.01	0.79	10	0.00	0.96	10	0.00	0.99	10
Stem density	0.03	0.37	0.02	0.67	10	0.09	0.39	10	0.01	0.81	10	0.20	0.20	10	0.01	0.91	10
% field	0.01	0.82	0.00	0.99	10	0.16	0.25	10	0.14	0.30	10	0.09	0.40	10	0.04	0.56	10
% forest	0.02	0.71	0.15	0.28	10	0.23	0.16	10	0.09	0.41	10	0.01	0.86	10	0.12	0.34	10
Fragmentation	0.01	0.54	0.02	0.69	10	0.12	0.33	10	0.05	0.54	10	0.03	0.66	10	0.02	0.73	10

^a Based on general linear model with deer density and town as random effects.

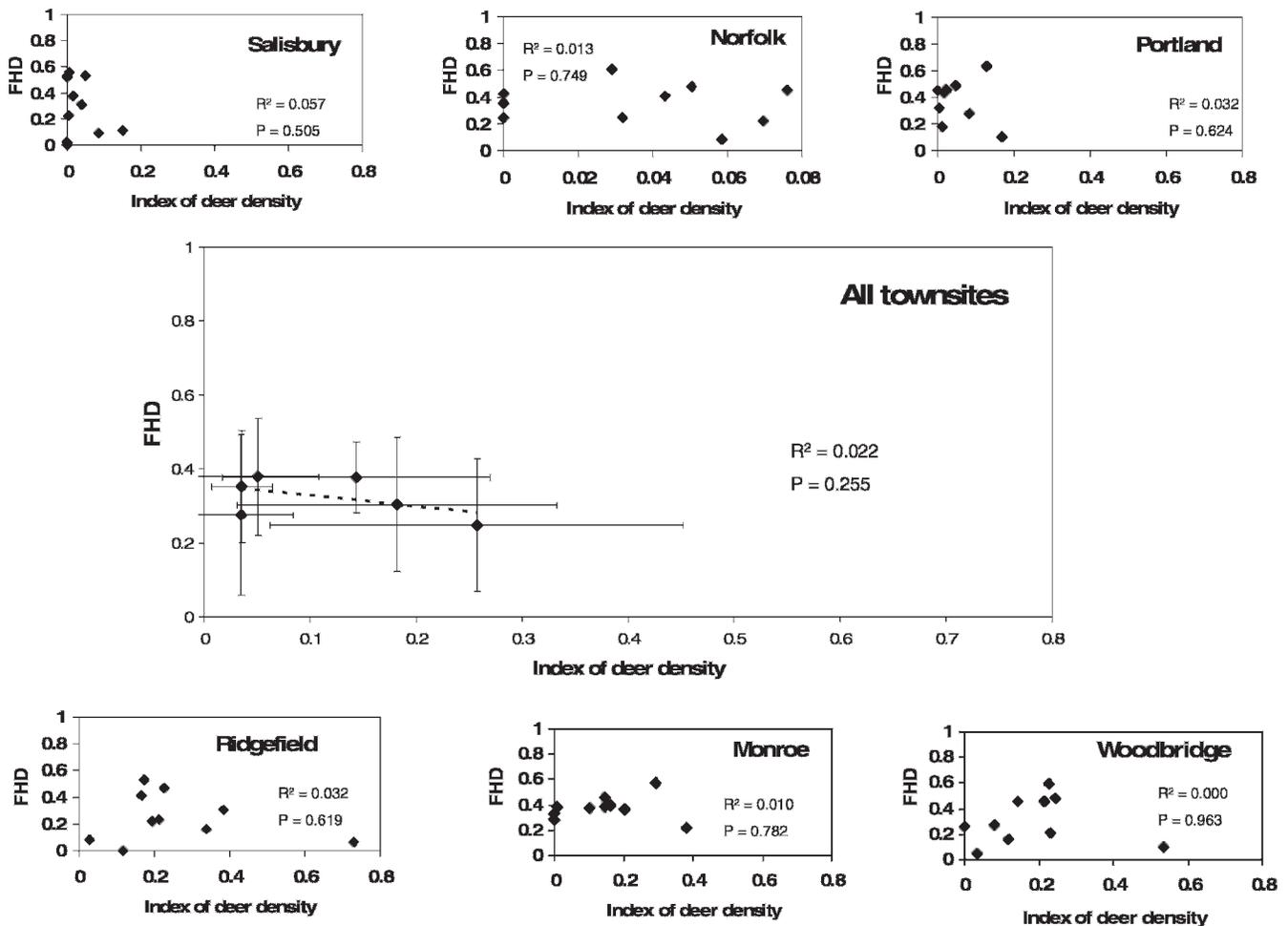


Figure 2. Correlation of foliage height diversity (FHD; an indicator of forest regeneration) and deer density index for 6 town sites in western Connecticut, USA, in which deer impacts were sampled during summer 2007. High FHD values indicate high forest regeneration and high deer index values indicate high deer densities. The data are presented for each town individually and for all town sites collectively. Values for individual town sites are the mean subplot value for each of 10 sampling plots. Values for all town sites are mean of the plot values \pm 1 standard error. The patterns observed in this figure are representative of all other associations of deer density index and vegetation variables measured in our study.

density was a leading factor determining loss of plant identity, plant diversity, and plant regeneration across this geographic region. Our index of deer abundance was unrelated to any measure of plant species abundance or diversity, and deer abundance did not seem to impact woodlot tree regeneration, measured either as FHD or total abundance of canopy trees (Table 2; Fig. 2). We believe that a lack of deer impact arises from 2 factors. First, only a minor fraction of plant species within each of the sites was palatable. Second, although data indicate deer in this region have erupted in abundance over the last 50 years, deer abundance still tends to range between low and modest levels compared to deer densities in other locations throughout the United States (Russell et al. 2001, Cote et al. 2004, Kilpatrick and LeBonte 2007). Moreover the highest average densities reported for our study region (11.5 deer/km²) remain below minimum levels (approx. 14–15 deer/km²) for which significant damaging impacts on plant species abundance and diversity have been reported (Russell et al. 2001). Indeed, deer enclosure treatments in unthinned forests for which deer densities varied over the same range of

comparatively low abundance as we observed in western Connecticut corroborate an outcome of no correlation between deer density and impact on seedling density (Tilghman 1989).

By treating deer as an independent variable in our statistical analyses, we explicitly assumed that deer are the direct determinant of environmental impacts. However, it may be that deer impacts are an indirect consequence of other factors that encourage local deer effects (Russell et al. 2001, Cote et al. 2004). For example, the nature and extent of land management and land development such as creation of edge via habitat fragmentation can create habitat that is attractive to deer, leading to local impacts (Chang et al. 1995, Russell et al. 2001, Kie et al. 2002, Cote et al. 2004, Tremblay et al. 2007). That is, deer would be a proximate cause of damage, not the ultimate cause, which is attractive habitat created by human land use. However, we did not find a correlation between deer density and degree of habitat fragmentation or between deer density and land use allocation (Table 3). We do not deny that deer impacts can be quite strong within each of the towns throughout our

study region (Fig. 2). But, these large impacts cannot generally be attributed to locally high deer density. Instead, in these localized sites impacts may be strongly affected by microsite conditions, such as soil conditions, light regimes, and microclimate, and by plant species composition, which differed locally, given only about 33% average similarity between town sites (Table 1). Implications of these variables on deer impacts are not normally investigated but perhaps they should be.

We are not trying to imply that deer cannot have significant environmental impacts. However, the evidence from our study in comparison with other published studies suggests that deer densities anywhere on the western Connecticut landscape may be below levels needed to cause strong impacts on the environment (Russell et al. 2001). Accordingly reducing deer density within much of this region likely will not alter levels of plant diversity or woodlot regeneration.

Our survey also speaks to management decision-making that often relies on scientific insights from local experimental studies to justify regional-scale deer population reductions to alleviate deer damage (e.g., Russell et al. 2001, Kilpatrick and LeBonte 2007). Basing decisions on data from experiments makes intuitive sense given the cause-effect insights that may be obtained. Indeed, experimentation has shown that deer density is often positively related to the magnitude of deer impacts on vegetation (Russell et al. 2001, Cote et al. 2004).

There is, however, an important limitation to using scientific insights from local experiments to make regional management prescriptions. Because of their local nature, experiments control for or homogenize extraneous variation to focus specifically on deer density and impacts. These experiments, therefore, do not include the kinds of landscape-scale variables that may confound or conflate deer density effects across broader geographic regions (Russell et al. 2001, Cote et al. 2004). Recognition of this limitation and the corresponding preponderance of such local case examples in the scientific literature have resulted in a call for more regional assessments across broader geographic contexts to quantify variation between deer abundance and impact (Russell et al. 2001, Cote et al. 2004).

Management Implications

Understanding how deer density is linked to environmental impacts is vital to making effective deer management decisions. Much of past management has been based on inferring links between high deer densities and detrimental environmental effects based on studies from other geographic regions, as opposed to measuring the potential for such links within a management area. We argue that drawing such a link may not be as reliable as widely presumed. Managers may want to decrease deer populations for other reasons, but the rationale that decreasing deer populations will, in and of itself, decrease damage to vegetation needs to be re-evaluated across a range of geographical scales and a range of deer densities. Managers would benefit from understanding the regional deer densities at which detrimental impacts begin to affect local

woodlots and forests. Additionally, if deer are not the primary factor causing impacts such as decreased tree regeneration and loss of plant diversity, it would increase management effectiveness to understand the causal factors, and their interactions, of such impacts.

Acknowledgments

We thank M. Cornish and the Doris Duke Charitable Foundation for their generous financial support. We appreciated a donation by Buzz Off Insect Repellent Clothing. We thank J. Daigneault for invaluable field assistance, T. Siccama for help with plant identification, and B. Barton, G. P. Hall, D. Hawlena, H. Jones, D. Skelly, and 2 anonymous reviewers for comments and discussion.

LITERATURE CITED

- Allan, B. F., F. Keesing, and R. S. Ostfeld. 2003. Effect of forest fragmentation on Lyme disease risk. *Conservation Biology* 17:267–272.
- Bashore, T. L., W. M. Tzilkowski, and E. D. Bellis. 1985. Analysis of deer-vehicle collision sites in Pennsylvania. *Journal of Wildlife Management* 49:769–774.
- Bridgen, M. 2009. Dr. Bridgen's list of plants that deer do not like to eat. Cornell University Long Island Horticultural Research and Extension Center. <<http://www.gardening.cornell.edu/factsheets/deerdef/>>. Accessed 15 Aug 2009.
- Brownstein, J. S., D. K. Skelly, T. R. Holford, and D. Fish. 2005. Forest fragmentation predicts local scale heterogeneity of lyme disease risk. *Oecologia* 146:469–475.
- Burroughs, J. P., and T. A. Dudek. 2008. "Deer-resistant" plants for homeowners. Michigan State University, Extension Bulletin E-3042, East Lansing, USA.
- Center for Land use Education And Research [CLEAR]. 2008. Connecticut landcover. <<http://clear.uconn.edu/data/interactive.htm>>. Accessed 28 Jan 2008.
- Chang, K. T., D. L. Verbyla, and J. J. Yeo. 1995. Spatial-analysis of habitat selection by Sitka black-tailed deer in Southeast Alaska, USA. *Environmental Management* 19:579–589.
- Condit, R., S. P. Hubbell, J. V. Lafrankie, R. Sukumar, N. Manokaran, R. B. Foster, and P. S. Ashton. 1996. Species-area and species-individual relationships for tropical trees: a comparison of three 50-ha plots. *Journal of Ecology* 84:549–562.
- Cote, S. D., T. P. Rooney, J. P. Tremblay, C. Dussault, and D. M. Waller. 2004. Ecological impacts of deer overabundance. *Annual Review of Ecology Evolution and Systematics* 35:113–147.
- DeCalesta, D. S. 1994. Effect of white-tailed deer on songbirds within managed forests in Pennsylvania. *Journal of Wildlife Management* 58:711–718.
- DeCalesta, D. S., and S. L. Stout. 1997. Relative deer density and sustainability: conceptual framework for integrating deer management with ecosystem management. *Wildlife Society Bulletin* 25:252–258.
- Goslee, S. C. 2006. Behavior of vegetation sampling methods in the presence of spatial autocorrelation. *Plant Ecology* 187:203–212.
- Gregonis, M. 2007. 2006/2007 aerial deer survey indicates stable population. *Connecticut Wildlife Magazine* 27(3):3–3.
- Grund, M. D., J. B. McAninch, and E. P. Wiggers. 2002. Seasonal movements and habitat use of female white-tailed deer associated with an urban park. *Journal of Wildlife Management* 66:123–130.
- Healy, W. M. 1997. Influence of deer on the structure and composition of oak forests in central Massachusetts. Pages 249–266 in W. J. McShea, H. B. Underwood, and J. H. Rappole, editors. *The science of overabundance: deer ecology and population management*. Smithsonian Institution Press, Washington, D.C., USA.
- Horsley, S. B., S. L. Stout, and D. S. DeCalesta. 2003. White-tailed deer impact on the vegetation dynamics of a northern hardwood forest. *Ecological Applications* 13:98–118.
- Hubbard, M. W., B. J. Danielson, and R. A. Schmitz. 2000. Factors influencing the location of deer-vehicle accidents in Iowa. *Journal of Wildlife Management* 64:707–713.

- Kie, J. G., R. T. Bowyer, M. C. Nicholson, B. B. Boroski, and E. R. Loft. 2002. Landscape heterogeneity at differing scales: effects on spatial distribution of mule deer. *Ecology* 83:530–544.
- Kilpatrick, H. J., and A. M. LeBonte. 2007. *Managing urban deer in Connecticut: a guide for residents and communities*. Second edition. Connecticut Department of Environmental Protection, Hartford, USA.
- Krebs, C. J. 1998. *Ecological methodology*. Second edition. Benjamin Cummings, San Francisco, California, USA.
- Ladd, P. G., R. Crosti, and S. Pignatti. 2005. Vegetative and seedling regeneration after fire in planted Sardinian pinewood compared with that in other areas of Mediterranean-type climate. *Journal of Biogeography* 32:85–98.
- Levy, S. 2006. A plague of deer. *Bioscience* 56:718–721.
- McShea, W. J., and J. H. Rappole. 2000. Managing the abundance and diversity of breeding bird populations through manipulation of deer populations. *Conservation Biology* 14:1161–1170.
- Morton Arboretum. 2009. Plants not favored by deer. <http://www.mortonarb.org/index.php?option=com_content&view=article&id=856&Itemid=6>. Accessed 15 Aug 2009.
- Nudds, T. D. 1977. Quantifying the vegetation structure of wildlife cover. *Wildlife Society Bulletin* 5:113–117.
- Russell, F. L., D. B. Zippin, and N. L. Fowler. 2001. Effects of white-tailed deer (*Odocoileus virginianus*) on plants, plant populations and communities: a review. *American Midland Naturalist* 146:1–26.
- Rutgers University. 2009. Landscape plants rated by deer resistance. New Jersey Agricultural Experiment Station. <<http://njaes.rutgers.edu/deerresistance/>>. Accessed 15 Aug 2009.
- Schmitz, O. J., and A. R. E. Sinclair. 1997. Rethinking the role of deer in forest ecosystem dynamics. Pages 201–223 in W. J. McShea, H. B. Underwood, and J. H. Rappole, editors. *The science of overabundance: deer ecology and population management*. Smithsonian Institution Scholarly Press, Washington, D.C., USA.
- Sinclair, A. E. 1997. Carrying capacity and the overabundance of deer: a framework for management. Pages 380–394 in W. J. McShea, H. B. Underwood, and J. H. Rappole, editors. *The science of overabundance: deer ecology and population management*. Smithsonian Institution Scholarly Press, Washington, D.C., USA.
- Thompson, J. A., and W. E. Sharpe. 2005. Soil fertility, white-tailed deer, and three trillium species: a field study. *Northeastern Naturalist* 12:379–390.
- Tilghman, N.G. 1989. Impacts of white-tailed deer on forest regeneration in northwestern Pennsylvania. *Journal of Wildlife Management* 53:524–532.
- Tremblay, J. P., J. Huot, and F. Potvin. 2007. Density-related effects of deer browsing on the regeneration dynamics of boreal forests. *Journal of Applied Ecology* 44:552–562.

Associate Editor: Hall.