

**TRACKING ITS ROOTS: EXPANSION BY JAPANESE STILTGRASS
(*MICROSTEGIUM VIMINEUM* (TRIN.) A. CAMPUS) IN THE
FORESTLANDS OF TENNESSEE**

An Undergraduate Research Scholars Thesis

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ABSTRACT

Tracking its Roots: Expansion by Japanese stiltgrass (*Microstegium vimineum* (Trin.) A. Campus) in the Forestlands of Tennessee

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Invasions by non-natives contribute to the loss of ecosystem biodiversity and productivity, modification of biogeochemical cycles, and inhibit natural regeneration of native species. Japanese stiltgrass (*Microstegium vimineum*) is one of the most prevalent invasive grasses in the forestlands of Tennessee. Hence, we aim to identify potential determinants of invasion and quantify the relative importance of each factor. We analyzed extensive field data collected Forest Inventory and Analysis Program of the U.S. Forest Service to quantify the range expansion of Japanese stiltgrass from 2000 to 2011. We then identified potential factors influencing the likelihood of its presence using boosted regression trees. Our results indicated that the presence of Japanese stiltgrass on sampled plots almost doubled during this period (from 269 to 404 plots), spreading extensively, geographically. The probability of invasion was positively correlated with landscape features, forest features, and disturbance factors. Our results suggest that range expansion by Japanese stiltgrass will continue to expand in Tennessee. The efficacy of management practices can be guided by identifying these factors of invasions and reduce the likelihood of invasion.

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CHAPTER I

INTRODUCTION

Forest ecosystems provide innumerable goods and services every year. From wood products to clean water supply and carbon sequestration, Costanza et al. (1997) has estimated the economic returns, on a global scale, to be \$4.7 trillion annually. While the net loss of global forest area has significantly declined since the 1990s, Keenan et al. (2015) points out naturally regenerating forests have decreased by 6% from 1990 to 2015. The decline in native forests parallels with the loss of native pollinators (Aizen and Feinsinger 1994) and a rise in unknown impacts of herbicides (Miller and Miller 2004). Plantations, on the other hand, have experienced a dramatic increase as the world's population expands and demands for more forest products (Fox 2000). Consequently, the understory of non-native plantations has been associated with the regeneration of native fauna. For example, in tree plantations of Costa Rica, Carnevale et al. (2002) reported the litter depth and diverse shading created by plantations yielded higher germination of native seedlings than naturally regenerated stands. Similarly, Feyera et al. (2002) argues the use of exotic species such as *Eucalyptus* in tropical plantations creates microclimates that improves the forest floor structure and favors conditions for native woody species.

Despite innovations in forestry practices to produce more wood, faster (Harfouche et al. 2011), forest plantations are often established by single, introduced species (FAO 2006). Introduced species are likely to become invasive in habitats where the intensity of a disturbance results in altered resource availability, low competition, and limited environmental stress (Alpert et al. 2000). Ehrenfeld et al. (2008) showed that areas invaded by non-natives exhibit dynamic changes in nutrient cycling processes (i.e. increased nitrification) and net primary productivity

(i.e. higher standing biomass). A prime example of this is in case of the pervasive shrub, *Tamarix ramosissima*, in the Mojave Desert where its large leaf area increased transpiration and changed the water budget (Sala et al. 1996). In the absence of anthropogenic disturbances, natives are particularly resilient to invasions. In a restored hardwood forest in southern Wisconsin, Harrington et al. (1989) found the native shrub increased its leaf area under low canopy light and hypothesized that it may have attributed to its growth. However, human activities such as intensive logging and land conversion to agriculture fields have fragmented and degraded many forest areas making them vulnerable to invasions (FAO 2006). Invasive species in the United States alone, account for \$120 billion/year in environmental damage; this estimate excludes the economic loss of species extinction, aesthetics, and other ecosystem services (Pimentel et al. 2005).

Among the most aggressive invasive species, grasses have proven to be significantly effective at invading and altering ecosystem processes in native communities (D'Antonio and Vitousek 1992). Experiments by Rhodes (1968) suggest an advantageous root system of certain grasses could explain this invasion phenomenon. One study conducted in California grasslands attests to these experiments, by finding the roots of the introduced annual grasses depleted the soil moisture and induced physiological strain on the growth of the oak seedlings (Gordon 1989). In addition to competing for water, the accumulation of litter by grass invasions can modify the availability of nutrient and light and impede on the development of native species. For example, areas invaded by a prevalent wetland grass, *Phalaris arundinaceae*, had higher litter density, which encouraged a positive-feedback in low nutrient conditions favorable to invader dominance (Eppinga and Molofsky 2013). Lastly, the flammability of many invasive grasses can modify fire

regimes. Fuentes-Ramirez et al. (2016) did an extensive study in the Sonoran and Mojave deserts that determined the flammability of the exotic grasses *Bromus madritensis* and *Schismus arabicus* initiated faster, shorter and cooler fires in a landscape poorly adapted to frequent fire. As biological invasions, continue to intensify and become a widespread problem, restoration of these novel habitats are complicated by human influences and global climate change, thereby making eradication in some cases impractical and management ineffective (Guo and Norman 2013). Although complete the removal of invasive species is rare (West 2002), there is an increasing number of studies using the understanding of plant invasions to improve management strategies (Blossey and Schroeder 1995; Luken 1997; Lesica and Marin 2003; Ellis-Felege et al 2013).

One such example is the introduction of Japanese stiltgrass (*Microstegium vimineum*), which has become a noxious weed in eastern United States. A native grass to Asia, Japanese stiltgrass was first discovered in Tennessee in 1919 and originally used as package filling for Chinese porcelain (Barden 1987). Invasion of Japanese stiltgrass can reduce growth and flowering of native species, suppress native plant communities, alter and suppress insect communities, slow plant succession, and alter nutrient cycling (Emery et al. 2013). Managing Japanese stiltgrass is challenged by persistent seedbanks in the soil (Gibson et al 2002) and invaded areas are often too vast for practical measures. Thus, few cost-effective methods exist that minimize harmful effects on native plant populations. Although there is currently no biological agent used to control infestations, mechanical and chemical methods have advanced in recent years. Judge et al. (2005) experimentally treated Japanese stiltgrass with grass herbicides, originally used for controlling *Digitaria sanguinalis* (large crabgrass), and found the herbicides

had a successful kill rate of 87% or greater with repeated application. After a 2-year experiment in southern Indiana, Flory (2010) noted that hand weeding and two herbicides (POST and PRE) were effective at reducing re-establishments of Japanese stiltgrass but did little to assist in the recovery of the native species. While there is no silver bullet for solving invasive plant problems, spatial distribution models can guide management decisions and become an invaluable tool for determining the vulnerability of habitats to invasion (Stohlgren 2002). Modelling spatial patterns can also address underlying causes of invasions. For example, Chong et al. (2001) determined that the invasive species were more abundant where native species richness was highest and positively correlated to lower elevations. Similarly, in the southern Blue Ridge Mountains, Anderson et al. (2013) suggested that the introduction of Japanese stiltgrass depends on variation in topography, degree of human activity (i.e. roads), and on biotic, abiotic conditions. However, due to seed dispersal limitations the development of a holistic spatial distribution of Japanese stiltgrass is difficult (Cole and Weltzin 2004). Hence, the estimation of recent range expansions is needed in order to enact preventative measures for controlling the spread of Japanese stiltgrass.

In this paper, I seek to identify trends in the range expansion Japanese stiltgrass within the forestlands of Tennessee. Based on an extensive set of field data collected by the U.S. Forest Service from 2002 to 2012, I will calculate the percent coverage change between two survey years and compare the spatial distributions. I will then identify potential factors influencing the likelihood of its presence using boosted regression trees. Since introductions and detrimental effects of insidious species like Japanese stiltgrass are hard to predict, understanding spatial

patterns is a way for management to be proactive and assist in identifying regions vulnerable to invasions.

CHAPTER II

METHODS

2.1. Study area

The primary focus for our investigation was in the state of Tennessee, where the first collection of Japanese stiltgrass was recorded in Knoxville, Tennessee (35.9606° N, 83.9207° W) alongside a creek bank (Fairbrothers and Gray 1972). Since then Japanese stiltgrass has expanded its geographical range into 27 states and invaded a diversity of habitats (USDA 2014). Little is known about whether temperature and precipitation patterns limit the range of the species.

2.2. Focal Species

Japanese stiltgrass germinates in early spring and slowly grows to a height of 2 m until mid-summer (Nees 2016). Like many other invasive exotic species, Japanese stiltgrass exhibit superior competitive ability in novel ecosystems. For example, in a single season it can produce thousands of seeds that persist for several years (Gibson et al. 2002). Moist soil conditions are conducive to rapid invasions of Japanese stiltgrass and often invades roadsides, stream corridors, and trails. A unique characteristic is its ability tolerate low-light environments and swiftly forms monocultures within the forest understories (Oswalt et al. 2007). Following a disturbance activity either by a natural (i.e. flooding) or anthropogenic (i.e. timber harvest, mowing) source, Japanese stiltgrass can rapidly invade and replace native plant communities (Nees 2016).

Table 1. Descriptions, possible values or units of measure, and means or counts of landscape conditions, forest features, disturbance factors, and forest management activities evaluated as potential factors of site invasion by Japanese stiltgrass in forest plots of Tennessee.

Variable	Value or Unit of Measure	Mean (Range) for Continuous Data/Count for Categorical Data
Landscape conditions		
Elevation	m	213.04 (-28.04 ~ 1809.9)
Slope	Degree	10.88 (0.00 ~ 57.5)
Adjacency to water bodies	No	2138
	Yes	665
Forest features		
Stand age	years	55.42 (2 ~ 137)
Site productivity	L1: 0-1.39 m ³ · ha ⁻¹ · year ⁻¹	0
	L2: 1.40-3.39	256
	L3: 3.50-5.94	1840
	L4: 5.95-8.39	802
	L5: 8.40-11.54	290
	L6: 11.55-15.74	78
	L7: >15.74	11
Species diversity	Shannon's species diversity	1.88 (0 ~ 3.02)
Basal area	m ²	98.15 (0.00 ~ 1447.67)
Natural regeneration ^a	No	3483
	Yes	64
Disturbance factors		
Distance to the nearest road	D1: < 30m	237
	D2: 30-91	390
	D3: 92-152	354
	D4: 153-305	578
	D5: 306-805	777
	D6: 806-1609	333
	D7: 1610-4828	116
	D8: 4829-8047	11
	D9: > 8047	7
Fire disturbance	No	3511
	Yes	36
Animal disturbance	No	3478
	Yes	69
Disease disturbance	No	3537
	Yes	10
Insect disturbance	No	3421

Table 1. Cont.

Variable	Value or Unit of Measure	Mean (Range) for Continuous Data/Count for Categorical Data
Human-caused disturbance	No	3482
	Yes	65
Weather disturbance	No	3470
	Yes	77
Forest management activities		
Site preparation	No	3508
	Yes	39
Artificial regeneration	No	3389
	Yes	158
Cutting	No	3270
	Yes	277
Ownership		
Forestland ownership	Public	486
	Private	3061

^a Normally within the past five years

2.3. Data Analysis

I associated the data on presence or absence of Japanese stilgrass (SNIPET) with the data on landscape conditions, forest features, disturbance factors, and forest management activities (FIA Data and Tools) using the FIA plot identification numbers. I also used the same dataset to compute Shannon's index of tree species diversity, H_s , for each plot (Wills et al 1997; Filipescu and Comeau 2007):

$$H_s = -\sum_{i=1}^{n_s} \frac{B_i}{B} \ln\left(\frac{B_i}{B}\right)$$

where B and B_i are the total stand basal areas and the basal area of trees of species i , respectively, and n_s is the number of tree species. I then conducted the analysis with a process of machine learning that uses boosted regression trees, which combines decision trees and a boosting algorithm with a form of logistic regression (Elith *et al.* 2008). I fitted the model in R

(R Development Core Team 2006 version 2.14.1) using the gbm package version 1.5-7 (Ridgeway 2006). I determined the optimal model following the recommendations of Elith *et al.* (Elith *et al.* 2008) which was the final model containing at least 1000 trees. I included randomness into the models to reduce over-fitting and also to improve accuracy and speed of the model selection process (Friedman 2002) which is having a bag fraction 0.6. I calculated the response variance explained, the area under the receiver operator characteristic curve (AUC), and the overall accuracy based on the aggregated cross-validation results. We evaluated the reliability and validity of the optimal model as fair ($0.50 < \text{AUC} \leq 0.75$), good ($0.75 < \text{AUC} \leq 0.92$), very good ($0.92 < \text{AUC} \leq 0.97$), or excellent ($0.97 < \text{AUC} \leq 1.00$) based on the value of AUC (Hosmer and Lemeshow 2000). I then used the gbm library to derive the relative influence of each potential explanatory variable in the optimal model and constructed partial dependence plots and fitted values plots for the most influential variables (Elith *et al.* 2008).

CHAPTER III

RESULTS

3.1. Historical Trends in Range Expansion

As indicated by the FIA records from 2011, Japanese stiltgrass spread extensively throughout the forestlands of Tennessee during an eleven-year period (Figure 1). The range expansion has been particularly dramatic in the northern latitudes (Figure 2). The presence of Japanese stiltgrass almost doubled from 269 plots (7.5%) in 2000 to 404 plots (11.3%) in 2011. Maury County had the largest increase in plots invaded followed by Giles, Madison, and Moore County. The highest increase in invasions were plots not previously invaded.

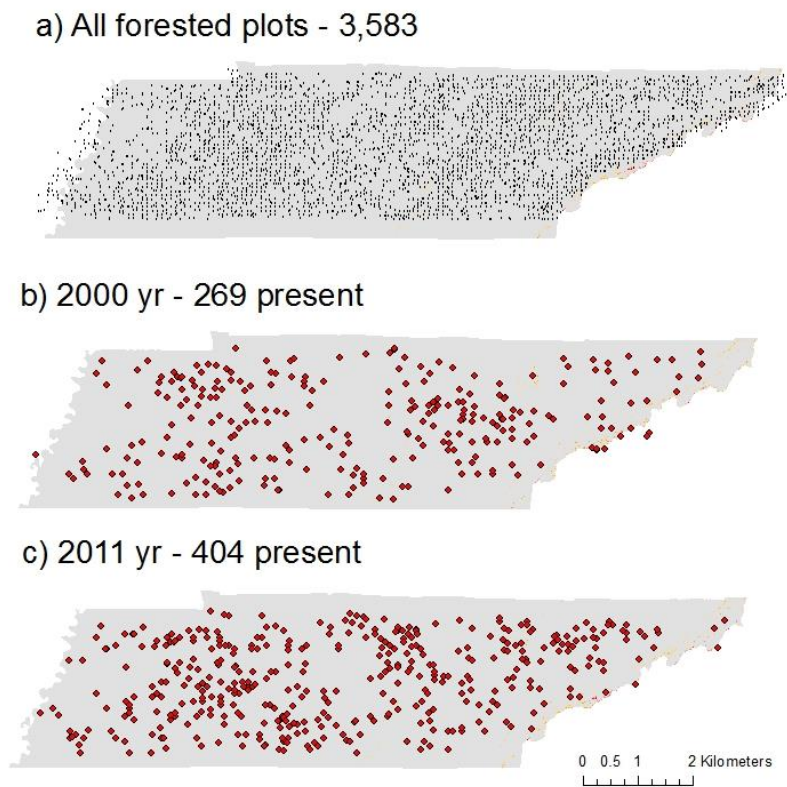


Figure 1. Presence (red dots) of Japanese stiltgrass in (a) forested plots sampled in Tennessee in (b) 2000 and (c) 2011 as part of the Forest Inventory and Analysis Program of the U.S Forest Service (USDA 2013).

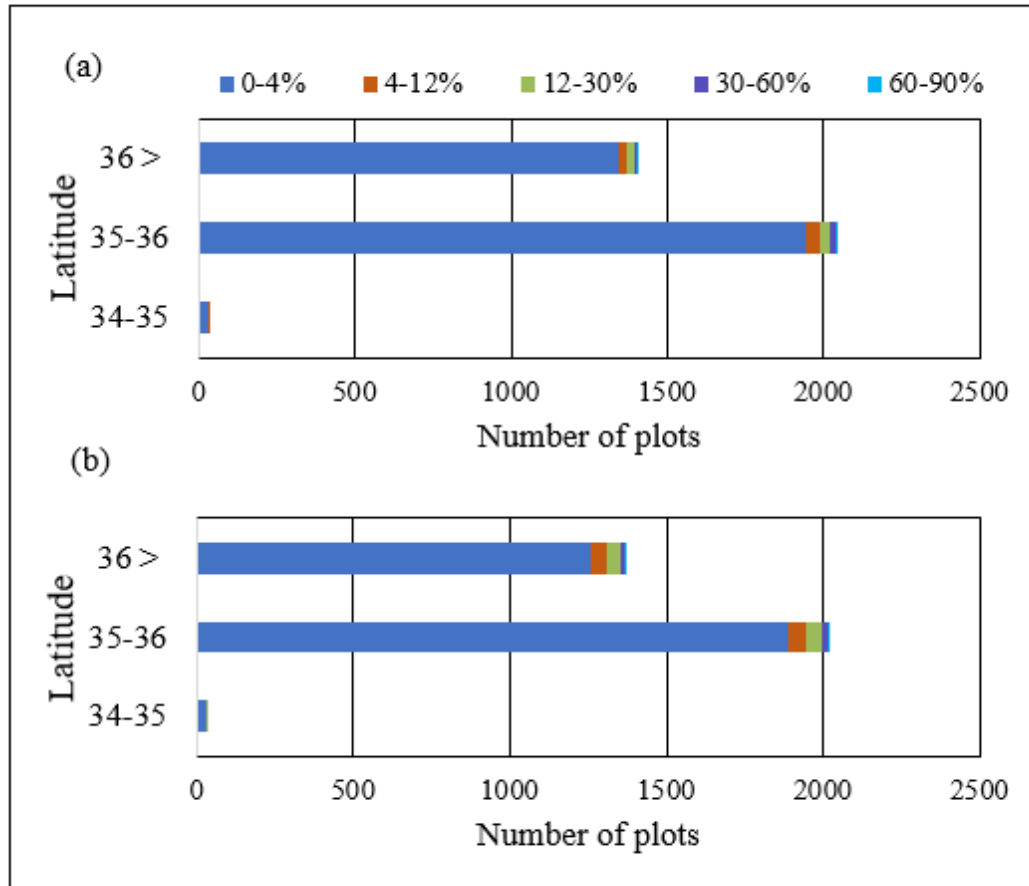


Figure 2. Comparison of the spatial distributions of Japanese stiltgrass during (a) the first survey (2000–2005) and (b) the second survey (2005–2011) conducted by the Forest Inventory and Analysis Program of the US Forest Service (USDA 2013). Results are summarized in terms of the number of plots in each of 3 latitudinal ($^{\circ}$ N) bands (34–35, 35–36, >36) in which the mean percent coverage of Japanese stiltgrass was <4, 4–12, 12–30, 30–60, and >60, respectively.

3.2. Potential Determinants of Invasion

We explored 300 combinations of tree complexity (ranging from 5 to 9) and learning rate (ranging from 0.0001 to 0.01); these produced models with between 700 and 1,500 trees. The optimal model had a tree complexity of 7, a learning rate of 0.0001, and a total of 1,020 trees. Model predictive deviance was 0.809 ± 0.001 with 80.6% of the total response variance explained.

The AUC score was 0.773 ± 0.022 (“good” ability to discriminate between species presence and absence). Recursive feature elimination tests showed that ten variables could be removed from the model before the resulting predictive deviance exceeded the initial predictive deviance of the model with all variables.

Examination of the relative contribution of the predictor variables indicated that the top four accounted for approximately 90% of the contribution in the overall model. Of the four most influential variables, three were forest features and one was a landscape feature. Species diversity was the most influential variable, contributing 30.68%. Basal area, elevation, and stand age were the second, third, and fourth most important variables, contributing 22.87%, 20.45%, and 16.69%, respectively. Forest features and landscape features had total contributions of 78.6% and 20.45%.

Partial dependence plots indicated that Japanese stiltgrass occurrences were associated species diversity higher than 0.5. Forest features usually included a basal area less than 300 m^2 , stand age between 10-20 years and greater than 100 years, site productivity higher than $11.5 \text{ m}^3 \cdot \text{ha}^{-1} \cdot \text{year}^{-1}$, and natural regeneration within the past five years. Occurrences were more likely in plots that were lower than 400 m in elevation.

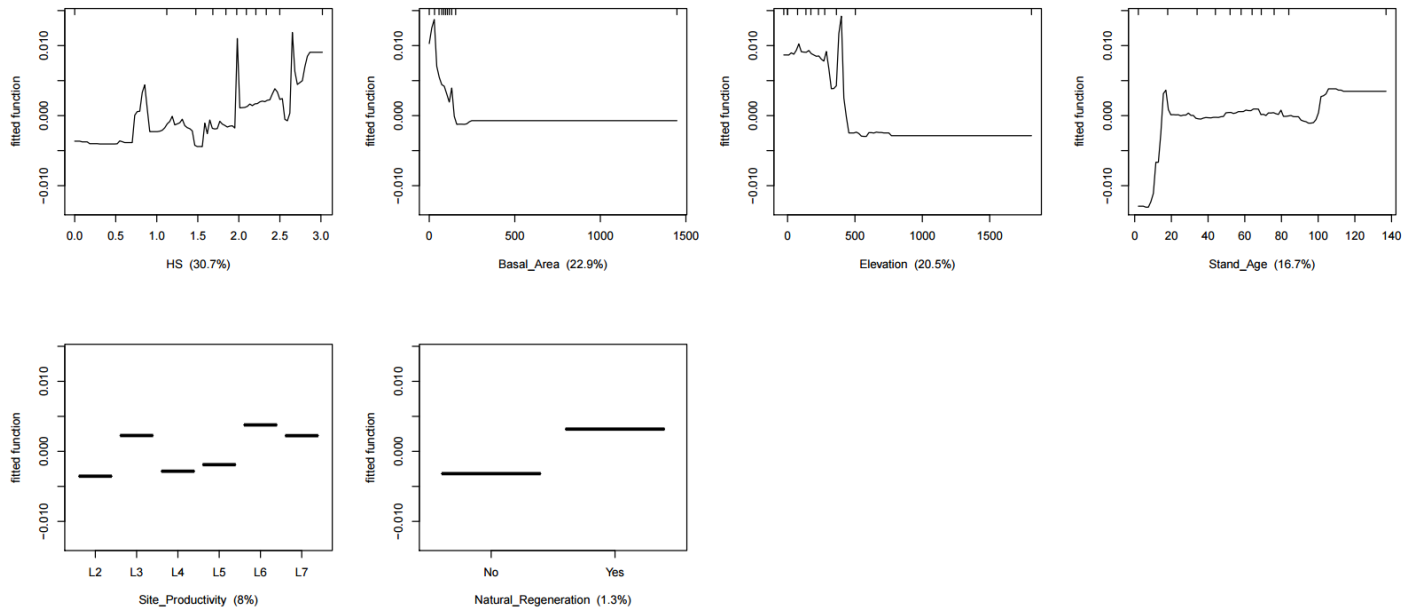


Figure 4. Partial dependence plots for the explanatory variables included in the optimal boosted regression tree models for Japanese stiltgrass based on analyses of the six most influential variables. Hash marks at the top of each plot indicate distribution of sample plots along the range of the indicated variable. X-axis labels include the relative contributions (%) of each influential variable included in the final model (see Table 1 for the description of variables). Y-axes are based on the logit scale used for the indicated variable.

CHAPTER IV

DISCUSSION

Predicting the spread of invasive species has captivated the interest of ecologists and land managers for many years (Barden 1987; Alpert et al. 2000; Guo and Norman 2013). Managers can prioritize their management strategies to specific areas by identifying the pattern of invasion based on environmental suitability models. Our study estimated the likelihood of invasion of Japanese stiltgrass within Tennessee by associating landscape features, forest features, forest management activities, species diversity, and disturbance factors with its presence or absence based primarily on analysis of data collected during the FIA inventory period ended in 2011. The presence of this species almost doubled within an eleven-year period. Sites in northern latitudes were invaded more frequently than those in the south (Figure 2), and the species was found in approximately 10% of the surveyed sites. However, uninvaded sites in Tennessee contributed to almost 90% and given its ability to swiftly invade, as shown by comparison of 2000 vs. 2011 presence-absence records, and its current distribution, there is potential for further invasion into these high risk areas.

The range expansion documented by our map of the study area signifies that the average invasion of Japanese stiltgrass per forested plot increased from 0.05 to 0.111 (Figure 2). Previous work in various species distribution models confirm the growth of the species population. Miller and Matlack (2010) experimentally tested the diffusive spread of this species in mature forests of Ohio and West Virginia. These authors found that it expanded between 0.80 to 1.60 m within 2 years of a 3-year observational study. On a finer spatial scale in patch growth, Rauschert et al.

suggested a slower dispersal rate for this species and its dependence on other environmental conditions such as anthropogenic vectors. Based on temporal and spatial scales, these studies prove the expansion of Japanese stiltgrass is non-linear and experience fluctuations in its dispersal processes. Furthermore, significant variability of its distribution indicates potential dispersal limitations for the non-native.

As our results suggest, a broad range of environmental conditions appear to facilitate Japanese stiltgrass invasions. Four potential factors contributed the most to the presence of this noxious weed: species diversity, basal area, elevation, and stand age. The boosted regression tree analysis identified that species diversity higher than 0.5 had the highest relative contribution of 30.7% to the presence of Japanese stiltgrass. We can assume that as the diversity of the tree species increase the probability of Japanese stiltgrass present increases. Our findings are consistent with a previous study that sought to characterize the distribution and spread of this non-native invasive grass in the southern Blue Ridge Mountains. For instance, Anderson et al. (2013) found the probability of Japanese stiltgrass was elevated under a high diversity of herbaceous species. Nord et al. (2010) concluded similar results that a positive relationship exists between species richness and seeding recruitment of Japanese stiltgrass suggesting diverse sites are more vulnerable to invasions. A diverse stand is likely to include species with functional traits, such as nitrogen fixing species, which can limit competition and enhance the exploitation of available resources (Forrester 2014).

Apart from suitable forest features, elevation also appears to facilitate Japanese stiltgrass invasions. From our regression analysis, elevation less than 400 m is associated with its dominance. Existing invasions have occurred in a similar range between 277 to 1,800 m (Anderson et al. 2013; Craig et al. 2015). Warren et al. (2011) observed the niche requirement of Japanese stiltgrass along a wide scale of environmental gradients from the southern Appalachian Mountains to the Georgia piedmont. They noted that a potential drawback of using a broad-scale predictors like elevation is the unintended consequences on non-target or native species because the stand appears as a homogenous monoculture.

The majority of the species occurrence have been noted on sites with basal area that is less than 300 m². Basal area is a common measurement used within forestry to estimate tree density and canopy cover. Therefore, low basal area can be a factor of dense stands or small average diameter. In the case of the latter, open canopy cover can dramatically increase the quality of light on the forest floor. Patches of Japanese stiltgrass can occur within the interior forest because of the favorable shift in light and nutrients as a gap opens in the canopy (Cole and Weltzin 2005). Since this non-native grass can invade disturbed and undisturbed habitats, it best for managers to focus on limiting its dispersal into close-canopy forests.

Stage ages between 10 to 20 years and greater than 100 years also appear to facilitate Japanese stiltgrass invasions. The higher abundance in young forest stands by this species may be a function of higher productivity, limited competition, and light availability. Particularly if the site experienced a disturbance, early developmental stages will have rapid accumulation of

biomass and nutrients. Swanson et al. (2011) argues that early-successional forests provide structural complexity, large nutrient fluxes, high productivity, and complex food webs for plant species because the tree canopy yields less influence over the forest site. As a forest matures the canopy closes there is a significant change in the microclimate such as exposure to light. This shift favors shade-tolerant species that can compete and survive in lower light levels. Japanese stiltgrass can persist and grow underneath the forest canopy (Barden 1987; Gibson et al. 2002; Cole and Weltzin 2004). A particular case in West Virginia, populations of Japanese stiltgrass entered into the forest interior via seed dispersal (Huebner 2010). Some have attributed its persistence in shade to phenotypic plasticity by allocating more resources to aboveground tissue to increase the leaf size (Winter 1982; Cheplick 2005). Others have suggested the rapid photosynthesis induction makes its C₄ advantageous over other C₃ plants during high-intense sunflecks (Horton and Neufeld 1998). The fact that Japanese stiltgrass can invade two very different seral stages indicate that its environmental niche requirement is extremely broad. Conditions that become intolerable to some plant species as the forest ecosystem progresses are not a barrier Japanese stiltgrass's distribution.

Our results showed that natural regeneration is conducive for the invasion of Japanese stiltgrass. The interlacing relationship between seedling establishment and site conditions makes natural regeneration a slow process and less reliable than artificial regeneration. Shearer and Schmidt (1999) found that even with enhancing the rate of natural regeneration using management treatments the establishment of the mixed conifer species increased slowly for 5 years. This crucial period after a disturbance or the transition between successional stages creates an open door for aggressive invaders like Japanese stiltgrass to invade. For example, in a highly

degraded Hawaiian dry forest the natural recruitment of natives without restorative efforts could not outcompete the dominance of an invasive grass (*Megathyrsus maximus*) (Ammond et al. 2013). In severely disturbed areas, land managers should consider employing artificial regeneration in the initial phases to limit the competition between the native and invasive species.

In conclusion, our analyses suggest that the range of Japanese stiltgrass in forestlands of Tennessee is expanding. Identifying vulnerable habitats, effective eradication methods, and distribution limitations of prolific invasive species are difficult and continue to drive research in these areas. While there is no direct solution for solving invasive plant problems, providing potential causal factors of invasions may improve management strategies and limit the spread of other non-natives (Wang et al. 2015). Our model demonstrates that FIA data can reflect broad-scale invasive trends. Our model may also guide control strategies for Japanese stiltgrass by facilitating early detection and eradication of newly established invasions in Tennessee.

REFERENCES

- Aizen, M.A.; Feinsinger, P. (1994) Forest fragmentation, pollination, and plant reproduction in a Chaco Dry Forest, Argentina. *Ecology*, **75**, 330–351.
- Alpert, P., Bone, E.; Holzapfel, C. (2000) Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants. *Perspectives in Plant Ecology, Evolution and Systematics*, **3**, 52–66.
- Ammond, S.A., Litton, C.M., Ellsworth, L.M. and Leary, J.K. (2013) Restoration of native plant communities in a Hawaiian dry lowland ecosystem dominated by the invasive grass *Megathyrsus maximus*. *Applied Vegetation Science*, **16**(1), pp.29-39.
- Anderson, D.P., Turner, M.G., Pearson, S.M., Albright, T.P., Peet, R.K.; Wieben, A. (2013) Predicting *Microstegium vimineum* invasion in natural plant communities of the southern Blue Ridge Mountains, USA. *Biological Invasions*, **15**, 1217–1230.
- Barden, L.S. (1987) Invasion of *Microstegium vimineum* (Poaceae), An exotic, annual, shade-tolerant, C₄ grass, into a North Carolina Floodplain. *American Midland Naturalist*, **118**, 40.
- Blossey, B.; Schroeder, D. (1995) Host specificity of three potential biological weed control agents attacking flowers and seeds of *Lythrum salicaria* (Purple Loosestrife). *Biological Control*, **5**, 47–53.
- Carnevale, N.J.; Montagnini, F. (2002) Facilitating regeneration of secondary forests with the use of mixed and pure plantations of indigenous tree species. *Forest Ecology and Management*, **163**, 217–227.
- Cheplick G.P. (2005) Biomass partitioning and reproductive allocation in the invasive, cleistogamous grass *Microstegium vimineum*: Influence of the light environment. *The Journal of the Torrey Botanical Society*, **132**, 214–224
- Chong, G.W., Reich, R.M., Kalkhan, M.A. and Stohlgren, T.J. (2001) New approaches for sampling and modeling native and exotic plant species richness. *Western North American Naturalist*, 328-335.
- Cole, P.G.; Weltzin, J.F. (2004) Environmental correlates of the distribution and abundance of *Microstegium vimineum*, in East Tennessee. *Southeastern Naturalist*, **3**, 545–562.
- Costanza, R., d'Arge, R., De Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J. and Raskin, R.G. (1997) The value of the world's ecosystem services and natural capital. *Ecological economics*, **25**(1), 3-16.
- Craig, M.E., Pearson, S.M. and Fraterrigo, J.M. (2015) Grass invasion effects on forest soil carbon depend on landscape-level land use patterns. *Ecology*, **96**(8), 2265-2279.

- D'antonio, C.M., Vitousek, P.M. (1992) Biological invasions by exotic grasses, the grass fire cycle, and global change. *Annual Review of Ecology and Systematics*, **23**, 63–87.
- Ehrenfeld, J.G. (2003) Effects of exotic plant invasions on soil nutrient cycling processes. *Ecosystems*, **6**, 503–523.
- Elith J, Leathwick JR, Hastie T. (2008) A working guide to boosted regression trees. *Journal of Animal Ecology*, **77**, 802-813.
- Ellis-Felege, S.N., Dixon, C.S.; Wilson, S.D. (2013) Impacts and management of invasive cool-season grasses in the Northern Great Plains: Challenges and opportunities for wildlife. *Wildlife Society Bulletin*.
- Emery, S.M., Flory, S.L., Clay, K., Robb, J.R.; Winters, B. (2013) Demographic responses of the invasive annual grass *Microstegium vimineum* to prescribed fires and herbicide. *Forest Ecology and Management*, **308**, 207–213.
- Eppinga, M.B.; Molofsky, J. (2013) Eco-evolutionary litter feedback as a driver of exotic plant invasion. *Perspectives in Plant Ecology, Evolution and Systematics*, **15**, 20–31.
- Fairbrothers, D.E. and Gray, J.R. (1972) *Microstegium vimineum* (Trin.) A. Camus (Gramineae) in the United States. *Bulletin of the Torrey Botanical Club*, **99**, 97-100.
- FAO (Food and Agriculture Organization of the United Nations). (2006) Global forest resources assessment 2005. Rome.
- Feyera, S., Beck, E.; Lüttge, U. (2002) Exotic trees as nurse-trees for the regeneration of natural tropical forests. *Trees*, **16**, 245–249.
- Filipescu, C.N.; Comeau, P.G. (2007) Competitive interactions between aspen and white spruce vary with stand age in boreal mixedwoods. *Forest Ecology and Management*, **247**, 175–184.
- Flory, S. L. (2010). Management of *Microstegium vimineum* invasions and recovery of resident plant communities. *Restoration Ecology*, **18**, 103-112.
- Forrester, D.I. (2014) The spatial and temporal dynamics of species interactions in mixed-species forests: from pattern to process. *Forest Ecology and Management*, **312**, 282-292.
- Fox, T.R. (2000) Sustained productivity in intensively managed forest plantations. *Forest Ecology and Management*, **138**, 187–202.
- Friedman JH. (2002) Stochastic gradient boosting. *Computational Statistics and Data Analysis*, **38**, 367-378.

- Fuentes-Ramirez, A., Veldman, J.W., Holzapfel, C.; Moloney, K.A. (2016) Spreaders, igniters, and burning shrubs: plant flammability explains novel fire dynamics in grass-invaded deserts. *Ecological Applications*, **26**, 2311–2322.
- Gibson, D.J., Spyreas, G.; Benedict, J. (2002) Life History of *Microstegium vimineum* (Poaceae), an invasive grass in Southern Illinois. *Journal of the Torrey Botanical Society*, **129**, 207.
- Gordon, D.R., Menke, J.M.; Rice, K.J. (1989) Competition for soil water between annual plants and blue oak (*Quercus douglasii*) seedlings. *Oecologia*, **79**, 533–541.
- Guo, Q.; Norman, S. (2013) Improving restoration to control plant invasions under climate change. *Invasive Plant Ecology*, 203–216.
- Harfouche, A., Meilan, R.; Altman, A. (2011) Tree genetic engineering and applications to sustainable forestry and biomass production. *Trends in Biotechnology*, **29**, 9–17.
- Harrington, R.A., Brown, B.J., Reich, P.B.; Fownes, J.H. (1989) Ecophysiology of exotic and native shrubs in Southern Wisconsin. *Oecologia*, **80**, 368–373.
- Hosmer DW, Lemeshow S. (2000) *Applied Logistic Regression*. New York, NY: John Wiley and Sons, Inc.
- Horton, J.L. and Neufeld, H.S., (1998) Photosynthetic responses of *Microstegium vimineum* (Trin.) A. Camus, a shade-tolerant, C 4 grass, to variable light environments. *Oecologia*, **114**(1), 11-19.
- Huebner, C.D. (2010) Spread of an invasive grass in closed-canopy deciduous forests across local and regional environmental gradients. *Biological Invasions*, **12**(7), 2081-2089.
- Judge, C.A., Neal, J.C.; Derr, J.F. (2005) Preemergence and postemergence control of *M. vimineum* (*Microstegium vimineum*) 1. *Weed Technology*, **19**, 183–189.
- Keenan, R.J., Reams, G.A., Achard, F., Freitas, J.V.D., Grainger, A.; Lindquist, E. (2015) Dynamics of global forest area: Results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, **352**, 9–20.
- Lesica, P.; Martin, B. (2003) Effects of prescribed fire and season of burn on recruitment of the invasive exotic plant, *Potentilla recta*, in a semiarid grassland. *Restoration Ecology*, **11**, 516–523.
- Luken, J.O. (1997) Management of plant invasions: Implicating ecological succession. *Springer Series on Environmental Management Assessment and Management of Plant Invasions*, 133–144.
- Miller, K.V.; Miller, J.H. (2004) Forestry herbicide influences on biodiversity and wildlife habitat in southern forests. *Wildlife Society Bulletin*, **32**, 1049–1060.

- Miller, N.P. and Matlack, G.R., 2010. Biodiversity Research: Population expansion in an invasive grass, *Microstegium vimineum*: a test of the channelled diffusion model. *Diversity and Distributions*, 16(5), pp.816-826.
- Nees, P. (2016) *Microstegium vimineum* (Trin.) A. Camus. *Bulletin OEPP/EPPO Bulletin*, **46**, 14-19.
- Oswalt, C.M., Oswalt, S.N.; Clatterbuck, W.K., (2007) Effects of *Microstegium vimineum* (Trin.) A. Camus on native woody species density and diversity in a productive mixed-hardwood forest in Tennessee. *Forest Ecology and Management*, **242**, 727-732.
- Pimentel, D., Zuniga, R.; Morrison, D. (2005) Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, **52**, 273–288.
- Rauschert, E.S., Mortensen, D.A., Bjørnstad, O.N., Nord, A.N. and Peskin, N. (2010) Slow spread of the aggressive invader, *Microstegium vimineum* (M. vimineum). *Biological Invasions*, 12(3), 563-579.
- Rhodes, I. (1968) The growth and development of some grasses species under competitive stress. I. Competition between seedlings, and between seedlings and established plants. *Grass and Forage Science*, **23**, 129–136.
- Ridgeway G. (2006) Generalized Boosted Regression models. Documentation on the R package ‘gbm’, version 1-5-7. <http://www.i-pensieri.com/gregr/gbm.shtml> (accessed on 4 April 2012).
- Sala, A., Smith, S.D.; Devitt, D.A. (1996) Water Use by *Tamarix Ramosissima* and Associated Phreatophytes in a Mojave Desert Floodplain. *Ecological Applications*, **6**, 888–898.
- Shearer, R.C. and Schmidt, J.A. (1999) Natural regeneration after harvest and residue treatment in a mixed conifer forest of northwestern Montana. *Canadian journal of forest research*, 29(2), 274-279.
- Stohlgren, T.J. (2002) Beyond theories of plant invasions: Lessons from natural landscapes. *Comments on Theoretical Biology*, **7**, 355–379.
- Swanson, M.E., Franklin, J.F., Beschta, R.L., Crisafulli, C.M., DellaSala, D.A., Hutto, R.L., Lindenmayer, D.B. and Swanson, F.J. (2011) The forgotten stage of forest succession: early-successional ecosystems on forest sites. *Frontiers in Ecology and the Environment*, 9(2), 117-125.
- USDA, NRCS (2014) The National Plant Data Center. The PLANTS Database <http://plants.usda.gov> [accessed 15 December 2016].
- USDA. (2013) Southern Nonnative Invasive Plant data Extraction Tool (SNIPET). Available

online at http://srsfia2.fs.fed.us/data_cener/index.shtml. [accessed 13 November 2016].

- Wang, H.H., Koralewski, T.E., McGrew, E.K., Grant, W.E. and Byram, T.D. (2015) Species Distribution Model for Management of an Invasive Vine in Forestlands of Eastern Texas. *Forests*, 6(12), pp.4374-4390.
- Warren, R.J., Wright, J.P. and Bradford, M.A., 2011. The putative niche requirements and landscape dynamics of *Microstegium vimineum*: an invasive Asian grass. *Biological Invasions*, 13(2), 471-483.
- West, C. J. (2002) Eradication of alien plants on Raoul Island, Kermadec Islands, New Zealand. *Turning the tide: the eradication of invasive species*. Gland, IUCN, 365-373.
- Wills, C.; Condit, R.; Foster, R.B.; Hubbell, S.P. (1997) Strong density- and diversity-related effects help to maintain tree species diversity in a neotropical forest. *Proceedings of the National Academy of Sciences USA*, 94, 1252–1257.
- Winter, K., Schmitt, M.R. and Edwards, G.E. (1982) *Microstegium vimineum*, a shade adapted C4 grass. *Plant Science Letters*, 24(3), pp.311-318.