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SPATIAL AND TEMPORAL PATTERNS OF INVASIVE EXOTIC PLANT SPECIES IN RESPONSE TO TIMBER HARVESTING IN A MIXED MESOPHYTIC FOREST OF EASTERN KENTUCKY

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THESIS

A thesis submitted in partial fulfillment of the

requirements for the degree of Master of Science in Forest and Natural Resource

Sciences in the College of Agriculture, Food and Environment

at the University of Kentucky

By

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2019

ABSTRACT OF THESIS

SPATIAL AND TEMPORAL PATTERNS OF INVASIVE EXOTIC PLANT SPECIES IN RESPONSE TO TIMBER HARVESTING IN A MIXED MESOPHYTIC FOREST OF EASTERN KENTUCKY

Invasive exotic species (IES) responses to silvicultural treatments eight years after timber harvesting were examined and compared to one-year post-harvest IES survey in University of Kentucky's Robinson Forest. The temporal effects of harvesting were further compared between harvested and non-harvested watersheds. Analyses were performed to identify IES spatial distribution and determine the relationships between IES presence and disturbance effects, biological, and environmental characteristics. IES prevalence was higher in the harvested watersheds and was influenced by canopy cover, shrub cover and disturbance proximity. *Ailanthus altissima* and *Microstegium vimineum* presence in the study area has decreased over time. Comparing to the 1-yr post-harvest study which only identified direct harvesting effects (e.g. canopy cover and disturbance proximities) as significant predictors, the 8-yr post-harvest survey results suggest that while harvesting effects and disturbance proximity still play an important role, environmental characteristics have also taken precedence in predicting IES presence. Overall IES prevalence has decreased but invasive plant species richness has increased over time. Results indicate that IES eradication may not need to be conducted immediately after harvesting, and when needed, can primarily target IES hotspots where low canopy cover, proximity to disturbance, and southwest facing slopes convene on the landscape.

KEYWORDS: invasive exotic species, timber harvesting, regeneration, *Ailanthus altissima*, *Microstegium vimineum*, temporal dynamics

Ben Rasp

April 25th, 2019

SPATIAL AND TEMPORAL PATTERNS OF INVASIVE EXOTIC PLANT SPECIES IN RESPONSE TO TIMBER HARVESTING IN A MIXED MESOPHYTIC FOREST OF EASTERN KENTUCKY

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CHAPTER ONE: INTRODUCTION

Invasive exotic species (IES) are “species that are non-native to the ecosystem under consideration and whose introduction causes or is likely to cause economic or environmental harm or harm to human health” (Beck et al. 2008). IES cause biological disruptions to natural ecosystems by decreasing native population sizes, not allowing maturation and reproducing, and ultimately resulting in the loss of endemic biodiversity (Adams and Engelhardt 2009; Orr et al. 2005; Olson et al. 2011; Beauvais et al 2016; Lurgi et al. 2016). IES also pose threats to ecosystem functions by deteriorating ecosystem processes such as nitrogen cycling and forest productivity (Levine et al. 2003; Grimm 2013), and degrade ecosystem services such as timber production, carbon sequestration, water quality regulation, and habitat (Pejchar and Mooney 2009; Olson et al. 2011; Staudt et al. 2013; Sun et al. 2013). Invasive plant species are homogenizing biodiversity by replacing specialist species and weakening ecosystem resilience to disruptive events (Bradley and Mustard 2006). The economy is not safe as well, with invasive plant species causing losses in agriculture, forestry, water treatment, and other segments of the U.S. economy (Bergman et al. 2000). The total economic impact of invasive plant species in the United States had been estimated at approximately $25 billion annually (Pimentel et al. 2005; Gurevitch et al. 2011). With both the ecological and economic impacts that IES affect, there is a need to understand the underlying mechanisms of species invasion process and what long term effects IES have on native ecosystems.

Weakened native ecosystems have a greater susceptibility to being invaded by IES because of disturbance. For an invasion to be considered successful, IES must overcome a variety of invasion filters (biological, physical, and environmental) along four spatial-temporal stages throughout the invasion process (Bartuszevige et al., 2006; Theoharides and Dukes, 2007). Defining the invasion stages allows enlightening comparisons of the importance of species traits, habitats, and disturbances at each stage (Theoharides and Dukes 2007). The first stage, the transport stage, is defined as a species moving over great distances from their native range with a major filter being the geographic distance IES travel. Colonization is the second stage, during which the IES propagules survive the abiotic filters (climate, resource availability, etc.) in the new habitat. Thirdly is the establishment phase, where IES develop expanding, self-sustaining populations and are mostly resisted by biotic factors (e.g., competition). Finally, the fourth stage is landscape spread, where IES disperse into other sites on a landscape and could be limited by landscape heterogeneity, IES genetic and dispersal traits, and meta-population dynamics. From a land management perspective, being able to understand the interactions between IES dispersal extent, the processes to overcome invasion filters, disturbance area proximity, and responses to disturbances may yield efficient prevention and eradication strategies when dealing with IES on the landscape. Disturbances can make the landscape more conducive to species invasion by modifying biological and environmental conditions and alleviating IES dispersal limitations (Hobbs and Huenneke 1992; Holl 2002; Huebner and Tobin 2006; Belote et al. 2012; Beauvais et al 2016). Plant invasion dynamics often involves the interaction between disturbance events and specific life history traits pertaining to the transport and colonization of IES on the landscape (D'Antonio et al. 2004; Theoharides and Dukes 2007; Eschtruth and Battles 2009).

Timber harvesting is one of the major disturbance agents in Appalachia (Holl 2002; Huebner and Tobin 2006; Devine 2011; Belote et al. 2012). Timber harvest operations potentially bypass invasion filters by transporting a variety of IES utilizing various dispersal mechanism into previously inaccessible habitats (Landenberger et al. 2007; Olson et al. 2011). The combination of a strong IES propagule pressure and timber harvest disturbance make conditions ideal for many IES to overcome invasion filters. Disturbances created by timber harvesting have also been shown to remove environmental filters by expanding habitats that satisfy colonization requirements, thus rendering that community more susceptible towards invasion (Gilliam 2002). On a landscape scale, skid trails created by timber harvesting (connecting harvested and un-harvested areas), can also create conditions for IES to overcome invasion filters (Hobbs and Huenneke 1992; Gibson et al. 2002; Gilliam 2002; Holl 2002; Zenner and Berger 2008; Belote et al. 2012). Forest management practices need to consider these problems for controlling invasive plant species. However, many silvicultural methods that are developed for the intended purpose of creating favorable conditions for desired trees, are often taken advantage of by undesirable invasive species (McNab and Loftis 2002). This led to creating best management practices (BMPs) to optimize silvicultural schemes while mitigating such negative effects on the ecosystem.

When designing best management practices to reduce the likelihood of IES invasion on a post-harvest landscape, scientists and land managers must consider IES propagule sources (Gustafson and Gardner 1996), potential dispersal corridors (Von Der

Lippe and Kowarik 2007) and colonization requirements (Rouget and Richardson 2003). The success of BMPs lies in accuracy of the predictive models to predict IES interactions between species traits and disturbances in the context of spatially heterogeneous landscapes. However, upon reviewing the literature, there are three major limitations that can be described when it comes to these studies (Ebeling et al 2008). First, most applied IES studies are conducted on a limited spatial scale, hence do not match theoretical predictions that demonstrate there are scale-dependent differences in resource competition and biases against long-range dispersing species (Brown and Peet, 2003). In addition, a few studies (Gilliam 2002; Holl 2002; Bartuszevige et al 2006) that are conducted at landscape scales produce results unsuitable for use in invasive plant control management schemes common to land managers. This is because these studies assumed that there is equal propagule pressure throughout a homogenous landscape, which overlooks interactions between microsites and propagules. This does not allow scientists and land managers to pinpoint the most effective way of mitigating or eliminating IES from a landscape. Secondly, information is needed on how IES invasion process evolves over time as forests recover from the harvesting disturbances. The rapid change of biological and environmental conditions within a few years after a timber harvest in a typical temperate forest can drastically impact establishment and spread opportunities for IES. One of the most obvious conditions to change is light availability. Due to the fast-growing tree regeneration, a harvested landscape can reach canopy closure in less than 10 years, hence significantly modifying the understory light availability. Other environmental conditions such as water retention (rising due to root infiltration and duff layer creation) and soil nutrient concentrations (e.g. altered by disturbance by forest operations, bare mineral soil disturbance, and depending on biodiversity of the area) can change due to the amount of regeneration that occurs in such a short time frame. The role of IES during this process is unclear, but the consensus is building that IES is a passenger not a driver of such changes to the landscape (MacDougall and Turkington 2005). Lastly, timber disturbance might interact with other disturbance agents in influencing IES invasion process. In Central Appalachia, one of the most pervasive disturbances is surface mining and forests timber harvesting exists with surface mining in a landscape matrix. Surface mining affects biology, soil, land cover, topography, hydrology, and geology of the operation zone and the landscape surrounding it. Surface mining reclamation can sometimes exacerbate the problem by intentional seeding of invasive plant species or by compaction to reduce erosion runoff that causes problems with natural reforestation processes. How disturbances caused by strip mines and timber harvesting interact microsite environment to determine invasion process remains elusive. Set by these limitations (small scale, multiple disturbance interactions, and spatial heterogeneity change over time), information identifying how disturbances interact with stages of the invasion process is difficult to determine what exactly progresses the invasion process at the landscape level. Without large-scale investigations, it will be difficult to develop BMPs suitable for implementation during forest management practices.

Starting in 2008 and completed in 2009, a Stream Management Zone (SMZ) project was conducted in the University of Kentucky Robinson Forest to examine how timber harvesting affected water quality downstream, with the intent of revising Kentucky BMPs for water quality management (Bowker 2013; Witt et al 2016). The project had one control and three harvested watersheds, where a two-aged deferment harvest was conducted. The SMZ study provided a perfect opportunity to observe long-term effects of timber harvesting to forests in Appalachia. Devine (2011) conducted a post-harvest survey after one full growing season to categorize the invasive plant species response to timber harvesting and found that timber harvesting removed invasion filters which expedited initial IES invasions throughout the harvested watersheds. Although such initial IES response studies support the idea that disturbances bring about negative effects such as non-native species invasion, there is a gap in the literature that does not answer the questions of how IES invasion changes over time, where IES are more likely to be found year to year, how IES diversity fluctuates over time, and what biological and/or environmental variables determine IES abundance on the landscape.

This study is designed to address the aforementioned questions by spanning the study area over multiple watersheds with varying level of timber harvest disturbance intensity and proximity to adjacent mined area, considering the influences of both timber harvesting and surface mining, and comparing the results about IES distribution in eight-year post-harvest watersheds directly to the initial post-harvest plant survey (Devine 2011). Such comparisons are the start of depicting the full picture of the invasion process. Our study will further the understanding of how invasive plant species respond to timber harvesting over time.

Our specific objectives were to (1) identify the landscape patterns of IES presence and richness in the harvested watersheds and the control watershed in Robinson Forest, (2) quantify the influences of biological and environmental variables that explain the IES distribution on the landscape, and (3) compare the differences between the 8-year post harvest IES response and the 1-year post harvest IES response.

CHAPTER TWO: MATERIALS AND METHODS

1. Study Area

The study area is in the University of Kentucky Robinson Forest, a 14,800-acre experimental forest in southeastern Kentucky that spans Breathitt, Knott, and Perry Counties. Robinson Forest is mostly comprised of secondary growth oak-hickory and mixed mesophytic forests that range from 40 to 100 years old. Located within the Northern Cumberland Plateau Ecological Subregion, the landscape is characterized with deeply incised drainages, narrow ridges, and steep slopes. Soils consist of shallow to moderately deep, well-drained, rocky or stony, silty clay to loam formed from sandstone and shale colluviums and residuum (Devine 2011).

In 2008, a SMZ project was conducted in Robinson Forest to examine how timber harvesting affected water quality downstream (Witt et al 2016). Several watersheds in Robinson Forest were chosen to be harvested, three of which were North Shelly Rock, West Shelly Rock, and South Shelly Rock. The adjacent Little Millseat watershed was chosen to act as a control and was not harvested. All watersheds are in the northwestern portion of Robinson Forest as part of the Clemons Fork watershed (Figure 1). For the SMZ harvested watersheds, a commercial two-aged deferment harvest targeting a residual basal area of 2.3 to 3.4 m^2ha^{-1} was applied to three watersheds in the summer of 2008, which served as the harvest treatment. Harvested watersheds fulfilled the Kentucky BMP for Stream Management Zones (Devine 2011; Witt et al 2016). Stream buffers were created based on stream classification of either perennial, intermittent, or ephemeral (Svec et al 2005). Bulldozers were used to construct skid trails largely along the contour, track-mounted feller bunchers and chainsaws were used for felling, and wheeled grapple and cable skidders were used to skid the timber to defined landings for loading onto trucks. *Dactylis glomerata* L. (orchard grass) and *Triticum aestivum* L. (winter wheat), both exotic species, were sowed onto the skid trails and water bars to help control erosion. Unmerchantable tree tops were left on site and, in some cases, aligned with the skid trails during the harvest. Harvest operations were completed in the summer of 2009.

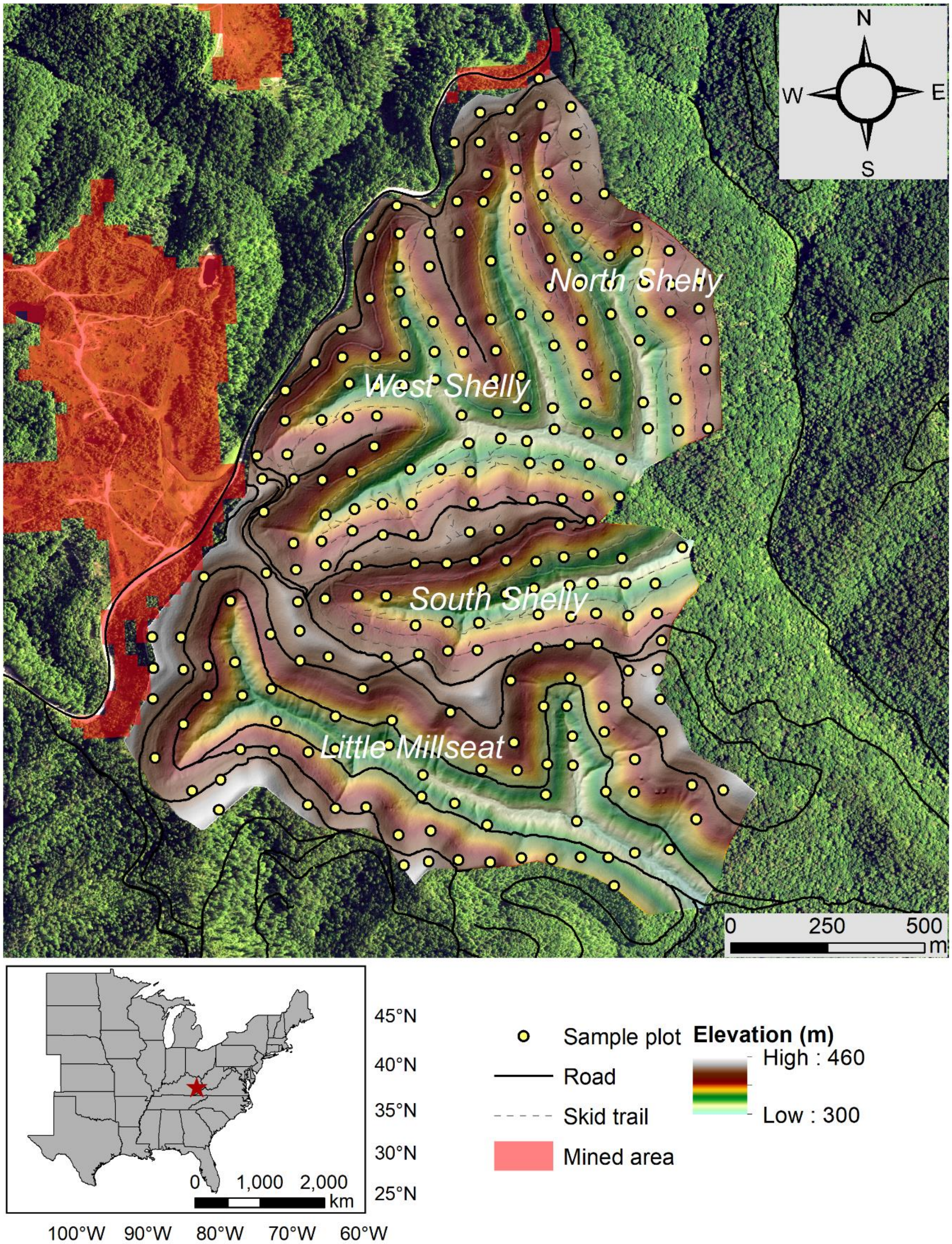

Figure 1 Study area with sampling plots and showing elevation, roads, skid trails, and adjacent mined areas

Prior to the timber harvesting completed in 2009, the dominant canopy tree species were *Liriodendron tulipifera* L. (yellow poplar), *Quercus rubra* L. (Northern red oak), and *Quercus alba* L. (White oak), with *Acer rubrum* L. (Red maple) and *Fagus grandifolia* Ehrh. (American Beech) as the dominant understory species. These tree species provide the primary sources of seeds, seedlings, and sprouts for the regenerating forest. There are reclaimed surface-minded lands on the outer edge of the study watersheds that contain a range of IES. Fei et al. (2009) surveyed Robinson Forest and found 11 IES mostly along roads and forest edges aligned with the reclaimed surface mines, including *Microstegium vimineum*, *Ailanthus altissima*, *Lonicera maackii*, *Elaeagnus umbellata*, and *Rosa multiflora*.

2. Sampling procedure and GIS operations

Post-harvest surveys were conducted eight years after the timber harvest during the summer of 2017 in the aforementioned watersheds of Robinson Forest (Figure 1). The plot survey network was established by Devine (2011) and was utilized for the eight year post harvest survey. The sampling order of the watersheds was North Shelly Rock, South Shelly Rock, West Shelly Rock, and then Little Millseat. To capture variations within the landscape, sampling plots were randomly selected from a systematic grid with centers 78 meters apart and oriented on cardinal directions (Huebner 2007; Devine 2011). We used the same random sampling plots from Devine (2011) and delineated the plots onto ArcGIS 10.4. These point locations were uploaded to a Garmin eTrex 20x® GPS unit to locate centers as accurately as possible. However, in thick canopy (typically >75% canopy cover), triangulation was required from an open patch (<20% canopy cover) to determine the plot center using a GPS unit, compass, and topographic map. A nested plot design consisting of a B-plot (1/300 acre) nested within an A-plot (1/100 acre) was used to provide consistency with, and allow for comparisons to one year measurers Devine 2011 (Figure 2). The cardinal points of the nested plot would be marked with flagging to delineate the plot. In the A-plot, tree DBH (including IES that are tree species) were taken if > 5 cm and their locations noted on a plot grid. The canopy coverage was visually estimated from the plot center using a 1” x 3.5” PVC pipe at the center of the A plot. Through the pipe, there was an average of 4-5 square foot view of the canopy (assuming a 15-20 ft. tall canopy). In the B-plot, tree species individuals were recorded and then were classified by five height classes (<0. 15 m, 0.15-0.30 m, 0.30-0.61 m, 0.61-1.22 m, >1.22 m) per species. In the B-plot, the crown diameter was determined for each woody shrub and the m^2 of the horizontal crown project was determined. These were summed and used to estimate total woody shrub canopy cover on each plot. Woody and herbaceous IES ground coverage was observationally estimated of area occupied by percentage in the B-plot. Any IES located in the B-plot were drawn on the plot grid. Percent estimates of ground cover were observationally estimated bare ground (this includes bare rock and streams), briar, woody, herb, vine, and fern in the B-plot. If a section of a skid trail, road, or stream were present in the plot, those features would be drawn and noted. Finally, native species richness was recorded of the B-plot by counting all native species found.

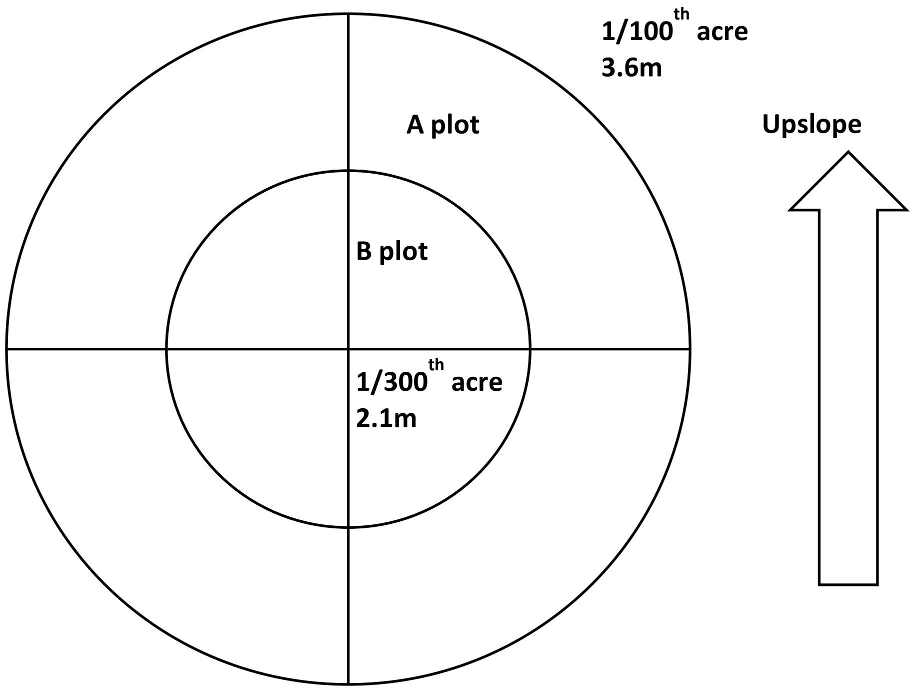

Figure 2 Nested plot design for field data collection

GIS variables representing the influences of landform and anthropogenic landscape features (e.g., roadway, skid trails) including elevation, slope, southwestness, topographic position index (TPI), slope position class (SPC), and topographic wetness index (TWI), were derived from a five-foot resolution DEM and various vector format GIS data in ArcMap 10.4 (Figure 3). These variables were used to examine relationships between environmental characteristics and IES occurrence on the landscape level (Boyd and Foody, 2011; Bradley and Mustard 2006). Southwestness is a cosine transformation of aspect minus 225 degree such that its value ranges from -1 (northeast-facing) to 1 (southwest-facing). TPI is the elevational difference between each DEM cell and the mean elevation of a user-specified neighborhood around that cell. Higher TPI represents higher topographic position relative to the surrounding areas. Slope position class (SPC) is a discrete reclassification of TPI in which TPI less than -1 standard deviation (SD) of the landscape-level TPI is considered valley, TPI greater than -1 SD but less than -0.5 SD is considered lower slope, TPI greater than -0.5 SD but less than 0.5 SD is considered middle slope, TPI greater than 0.5 SD but less than 1 SD is considered upper slope, and TPI greater than 1 SD is considered ridge (Weiss 2001). TWI is a widely used topographic attribute designed to quantify the effect of local topography on hydrological processes and for modeling the spatial distribution of soil moisture (Qin et al. 2011). It is computed as the logarithm of the ratio between upslope contributing area per unit contour length and tangent transformation of local slope. Euclidean distances to skid trails, roads, and stream buffer zones were derived through ArcMap 10.4 using the geoprocessing *Near* tool (Figure 4). A canopy cover GIS map derived from a LiDAR point cloud data that was obtained in 2014 was also utilized in data analysis (Staats 2015). These variables were chosen based off our knowledge of IES interactions with disturbance and physical environments. We created these GIS-derived variables either to compare with a previous study (Devine 2011) or to further investigate the IES relationship with disturbance in a recovering forested landscape. Finally, we rated the threat level of IES found in our survey based on the Kentucky Invasive Plant Council (2013) (KY-IPC) (1 = severe, 2 = significant, and 3 = not considered a threat). These species grouped based on the KY-IPC threat level were further categorized by growth form: grass, shrub, and tree.

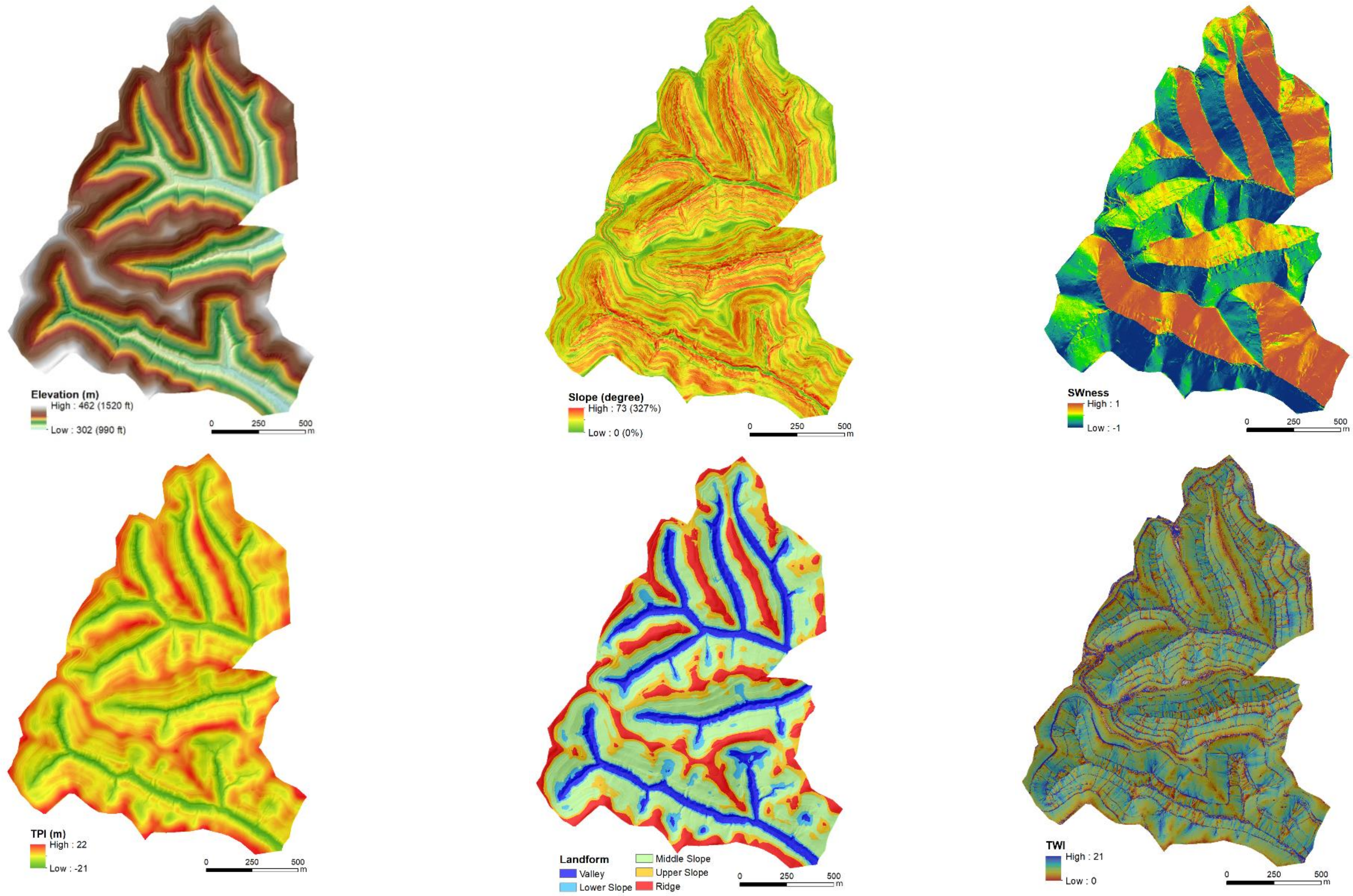

Figure 3 Environmental variables representing landform influences. From left to right and top to bottom: elevation, slope steepness, southwestness, TPI, slope position classification, and TWI

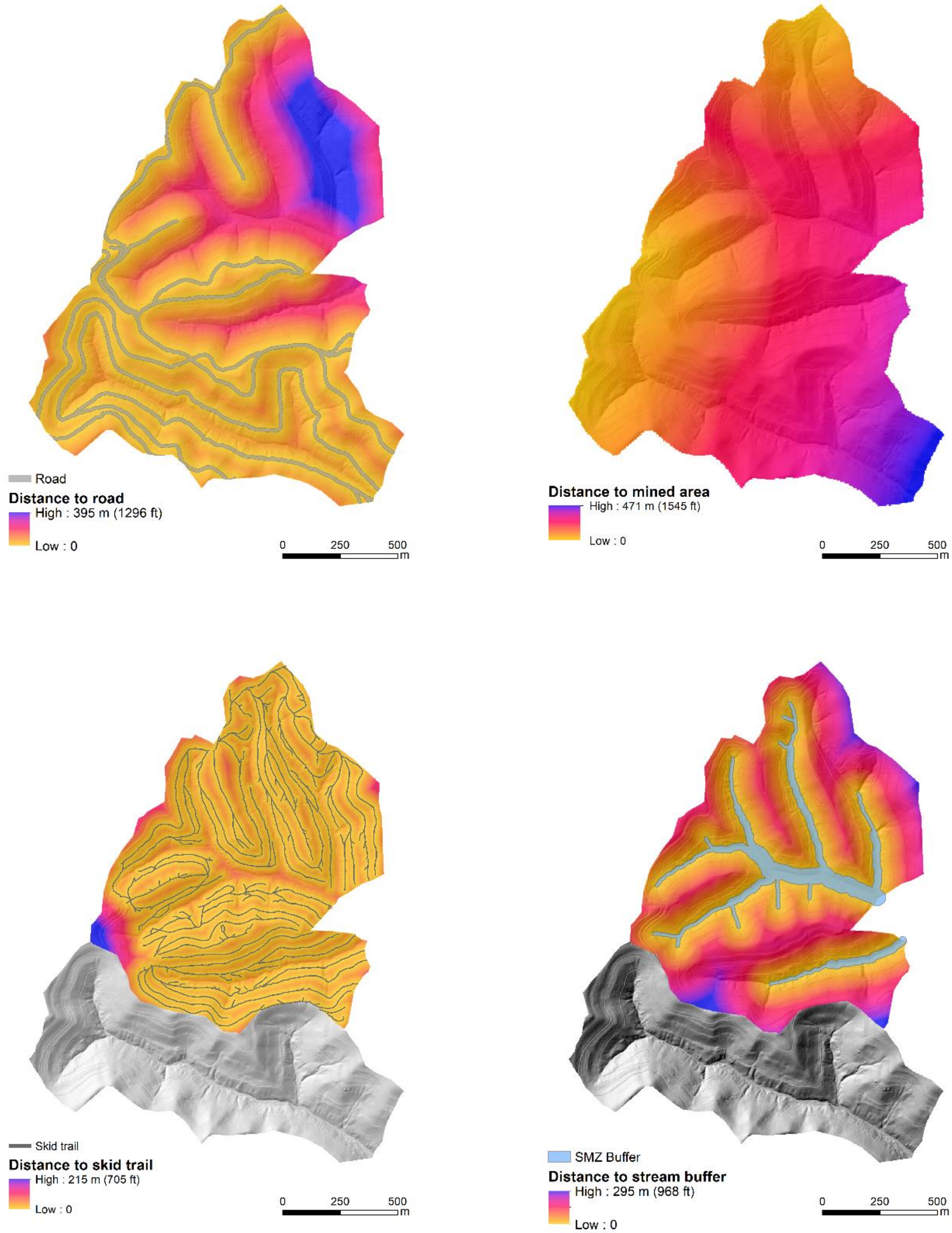

Figure 4 Environmental variables representing anthropogenic influences. From left to right and top to bottom: Euclidean distance to road, Euclidean distance to mined areas, Euclidean distance to skid trails, and Euclidean distance to stream buffers

3. Statistical Analysis

To address our first objective (*identifying the landscape patterns of IES presence and richness in the harvested watersheds and the control watershed*), ArcGIS 10.4 was used to map overall IES presence and richness of the 249 sampling plots in the study area. The overall IES presence was further broken down to show spatial patterns of observed KY-IPC significant threat level invasive species group (IPC1) and its sub-groups by growth forms (grass: IPC1G, shrub: IPC1S, and tree: IPC1T). Two principal IES, *Microstegium vimineum* and *Ailanthus altissima*, were also shown in separate maps. The proportional test was used to compare the proportion of IES across the four watersheds. This test was also conducted at the species group and individual species levels. We chose the proportional test because it is the most suitable for hypothesis testing of the binary data, while the traditional ANOVA test is only appropriate for continuous data. To determine how IES proportions differ between watersheds, the pair-wise proportional test was conducted. Overall variability of IES species richness was summarized by a contingency table in which the count of each IES richness level (varying from 0 to 7) was tabulated by watershed. A hurdle model was used to determine if there were any significant differences in IES richness among the watersheds. The hurdle model was chosen because it can overcome the overdispersion and excess zeros issues in hypothesis testing of the count data (Zeileis et al. 2008).

Generalized linear modeling (GLM) was used to address our second objective (*quantifying the influences of biological and environmental variables that explain the IES distribution on the landscape*). To reduce the collinearity of the predictor variables, the correlation test was used to determine if there were any significant correlation between any pairs of the environmental and biological variables for the modeling purposes. Variables that exhibited more than 75% correlations with a primary predictor variable were removed from the modeling selection procedure. Uni-variate logistic regression models were used to determine if an individual variable had better predictive power when log transformed or not for being utilized in the subsequent model selections. A multi-variate logistic regression model was utilized to identify any significant relationships of the predictor variables with IES presence. To avoid overfitting problems of too many predictor variables comparing to sample size, we first used an AIC-based forward selection procedure to find the best biological- and environmental-only model that had the least possible number of predicting variables without significantly sacrificing predicting power. Then, the AIC-based backward selection was used to identify the most parsimonious model using the best biological and environmental variables identified from the forward selection process. This model selection process was used for the following groups of IES: *A. altissima* (as AIAL), *M. vimineum* (as MIVI), all severe threat level IES found (as IPC1), and all IES found (as IES). Finally, the nagelkerke test was chosen to evaluate the predicting power of the models using the Nagelkerke pseudo R-squared value (Cragg and Uhler 1970; Nagelkerke 1991). A Welch 2-sample t-test was used to test if there are differences between the presence plots and absence plots for each predictor variable that was deemed significant by the most parsimonious GLM model. Boxplots were created to show the distributional difference of predictor variables between the presence and absence group. Histograms were paired with the boxplots to show the frequency (measured as the percentage of total counts) distribution and visualize the influence of the predictor variable on IES distribution on the landscape.

To answer the last objective (*compare the differences between the 8-year post harvest IES response and the 1-year post harvest IES response*), ArcGIS 10.4 was used to map 1-yr post-harvest overall IES presence of the 249 sampling plots in the study area. The proportional test was used to see the proportion of IES present Devine's (2011) survey. An IES observation table of all IES found during the one-year post-harvest survey was created. Finally, IES data results of the eight-year post-harvest survey were compared to Devine's (2011) to understand the temporal effect on IES prevalence in a landscape.

CHAPTER THREE: RESULTS

The landscape pattern of IES among sampling plots in the study area showed that the harvested watersheds in general, had higher IES prevalence than the control watershed (Figure 5). North Shelly Rock had the highest proportion of IES presence at 73% of the plots, followed by South Shelly Rock at 57%, West Shelly Rock at 51%, and Little Millseat at 21% (Table 1). The pair-wise proportional test showed that Little Millseat was significantly different from harvested watersheds. North Shelly Rock and West Shelly Rock were significantly different from each other but not from South Shelly Rock, respectively. The spatial distribution of all severe threat level IES based on KY-IPC by growth form (grass, shrub, and tree) were mapped (Figure 6). The shrub- and tree-growth forms of IES were found mostly along the roads and skid trails and severe threat grasses were more likely on the skid trails (Figure 6).

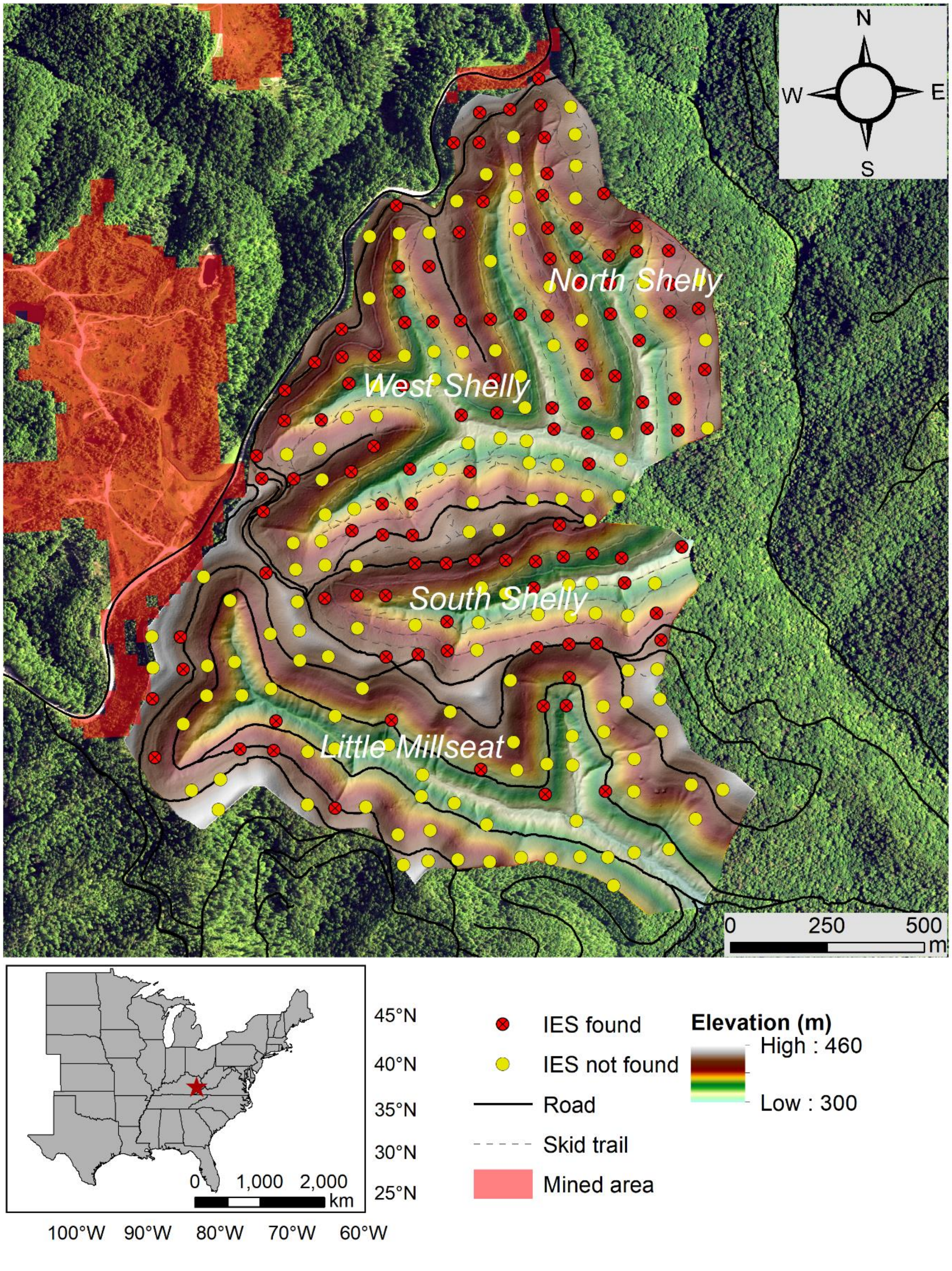

Figure 5 Overall IES presence and absence in sampling plots among all watersheds of the study area

Table 1. *Proportion test comparing IES presence proportions among watersheds*

Sample estimates: Presence proportion	North Shelly Rock	South Shelly Rock	West Shelly Rock	Little Millseat	p-value
IES:	0.730^{c}	0.571^{bc}	0.511^{b}	0.211^{a}	<0.001***
IPC IES total:	0.622^{c}	0.357^{b}	0.340^{b}	0.105^{a}	<0.001***
IPC-G IES grass:	0.568^{c}	0.262^{b}	0.170^{b}	0.026^{a}	<0.001***
IPC-S IES shrub:	0.135	0.071	0.138	0.092	0.605
IPC-T IES tree:	0.054^{ab}	0.048^{ab}	0.117^{b}	0^{a}	0.016*
AIAL:	0.054^{ab}	0.048^{ab}	0.117^{b}	0^{a}	0.016*
MIVI:	0.541^{c}	0.167^{b}	0.096^{ab}	0.026^{a}	<0.001***

IPC means species that were deemed as a severe threat by the Invasive Plant Council.
Pair-wise proportional test results are represented by codes a, b, and c. These codes equate to significantly different groups.
P-value significance codes are: 0 '***' 0.001 '**' 0.01 '*' 0.05 '.'

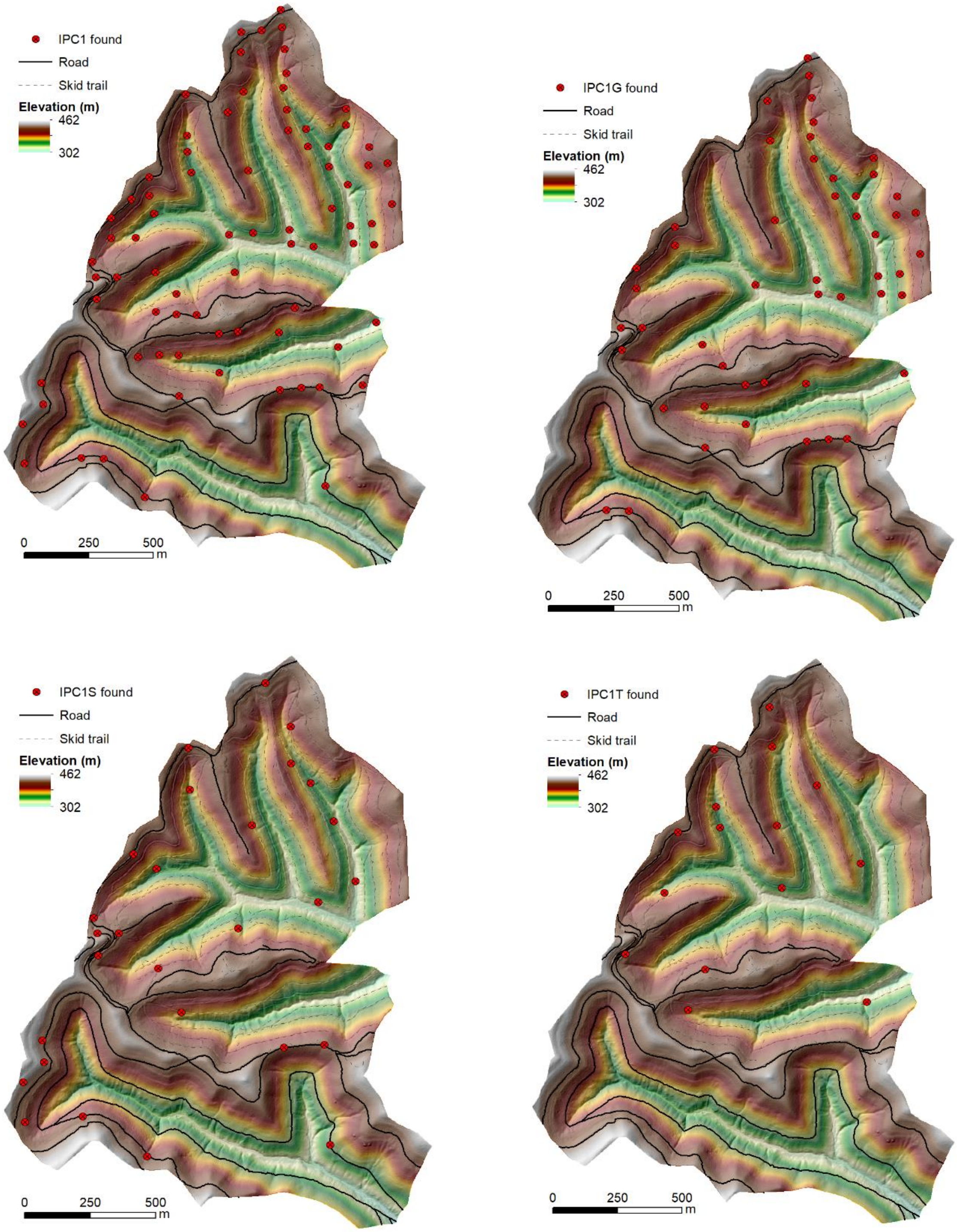

Figure 6 All severe threat-level (KY-IPC) IES groupings in clockwise order starting in the top left: total, grass-, tree-, and shrub- growth forms in sampling plots among all watersheds of the study area

The KY-IPC severe threat level invasive plant species proportions were 62% in North Shelly Rock, 36% in South Shelly Rock, 34% in West Shelly Rock, and 11% in Little Millseat (Table 1). The pair-wise proportional test showed Little Millseat had significantly lower KY-IPC IES proportion (11%) than any other watershed ($p \leq 0.001$). In contrast, North Shelly Rock had significantly higher KY-IPC IES proportion (62%) than any other watershed ($p \leq 0.001$). West and South Shelly Rock had similar KY-IPC IES proportions, 34% and 36%, respectively. Grasses listed as a severe threat occupied 57% in North Shelly Rock, 26% in South Shelly Rock, 17% in West Shelly Rock, and 3% in Little Millseat of the total IES ($p \leq 0.001$) (Table 1). The pair-wise proportional test showed Little Millseat had significantly lower invasive grass species proportion than any other watershed and North Shelly Rock had significantly higher invasive grass species proportion than other watersheds ($p \leq 0.001$). West and South Shelly Rock had similar IES proportions to one another. Pair-wise proportional test for shrubs considered as a severe threat showed no significant difference among watersheds (Table 1) ranging with KY-IPC proportions ranging 7% to 14%. Finally, the severe threat level tree proportions showed significant differences among watersheds (Table 1). The pair-wise proportional test showed West Shelly Rock had significantly higher invasive tree species proportion (12%) than any other watershed. In contrast, Little Millseat had significantly lower invasive tree species proportion (0%) and North and South Shelly Rock had similar invasive tree species proportions (5%).

The invasive grass species (*M. vimineum*) and tree species (*A. altissima*) were mapped out (Figure 7) and were observed in proximity to roads and skid trails. *M. vimineum* had the highest proportion in North Shelly Rock (54%), then 17% in South

Shelly Rock, 10% in West Shelly Rock, and lastly 3% in Little Millseat. The pair-wise proportional test showed North Shelly Rock had significantly higher proportions of *M. vimineum* (Table 1). Little Millseat and South Shelly Rock are significantly different from each other but not from West Shelly Rock, respectively. *A. altissima* proportions ranged 0% in Little Millseat, 5% in North and South Shelly Rocks, and 12% in West Shelly Rock that exhibited significantly higher proportion (p=0.016) than other watersheds (Table 1). In contrast, Little Millseat had significantly lower proportion of *A. altissima* while North and South Shelly Rock had similar proportions (Table 1).

The harvested watersheds had higher IES richness than Little Millseat (Figure 8). The highest invasive species richness plots were primarily found in the harvested watersheds. There were 17 IES found throughout the study watersheds (Table 2). The top three herbaceous IES were *M. vimineum, Poa pratensis,* and *Schedonorus arundinacea* (38, 33, and 29 observations, respectively). The top three woody IES were *Lespedeza bicolor*, *A. altissima,* and *Lonicera maackii* (19, 15, and 15 observations, respectively). The invasive species richness varied among the study watersheds (Table 3). South Shelly Rock and West Shelly Rock watersheds had the highest species richness observed in any one plot with seven IES found. Since there were too many zeros to use a Poisson regression model for invasive species richness (Dispersion = 1.723, p=0.003), the hurdle model (Table 4) was used for further determining IES richness differences among these four watersheds. South Shelly Rock and West Shelly Rock watersheds had significantly higher count model coefficients than Little Millseat and North Shelly Rock (p=0.023 and p=0.028, respectively), suggesting these two watersheds had higher invasive species richness (Table 4). All of the harvested watersheds had significantly positive hurdle model coefficients, suggesting they have higher proportions of invasive species presence than the control watershed (p-values < 0.001) (Table 4).

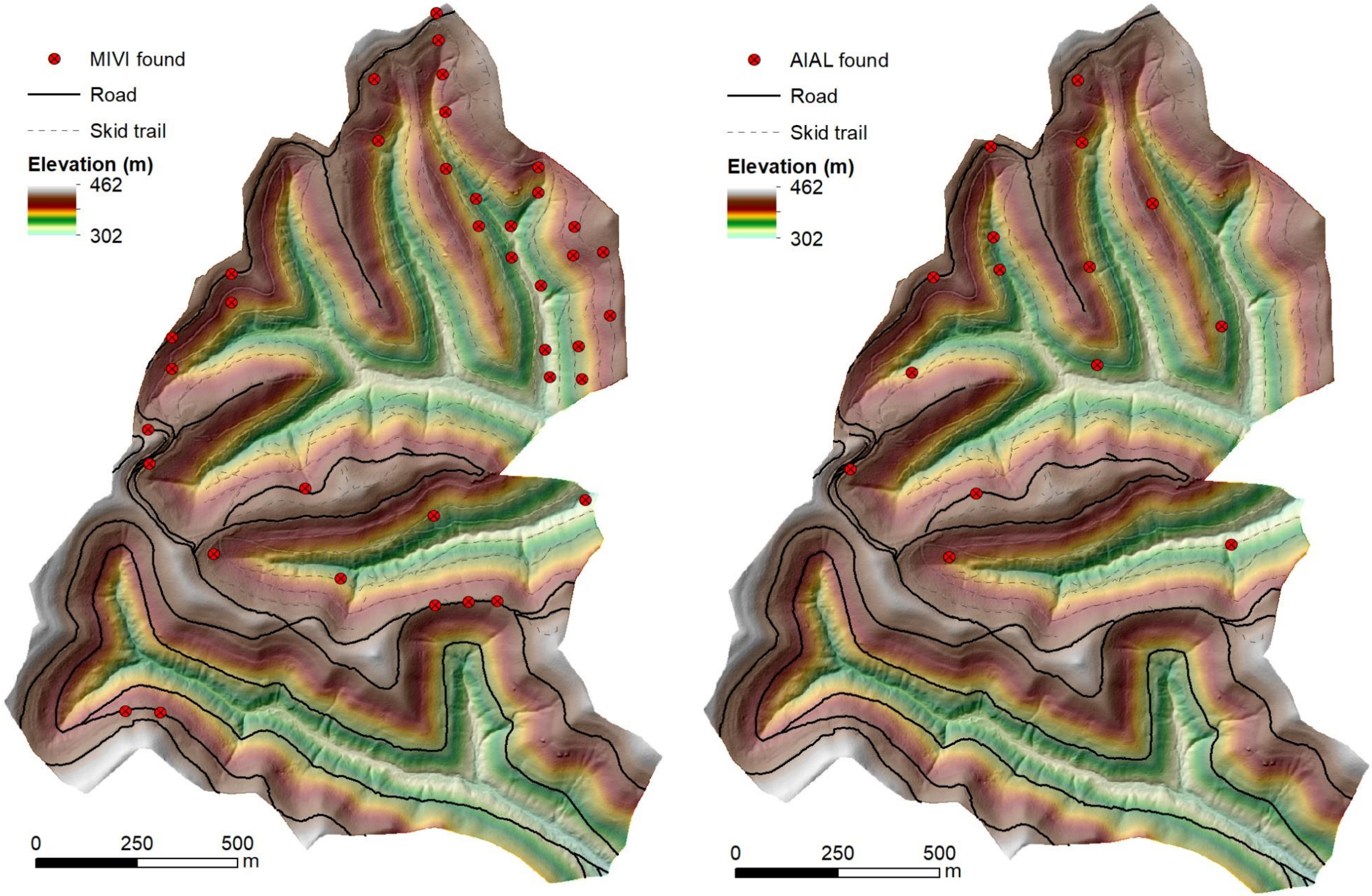

Figure 7 *M. vimineum* and *A. altissima* presence in the sampling plots

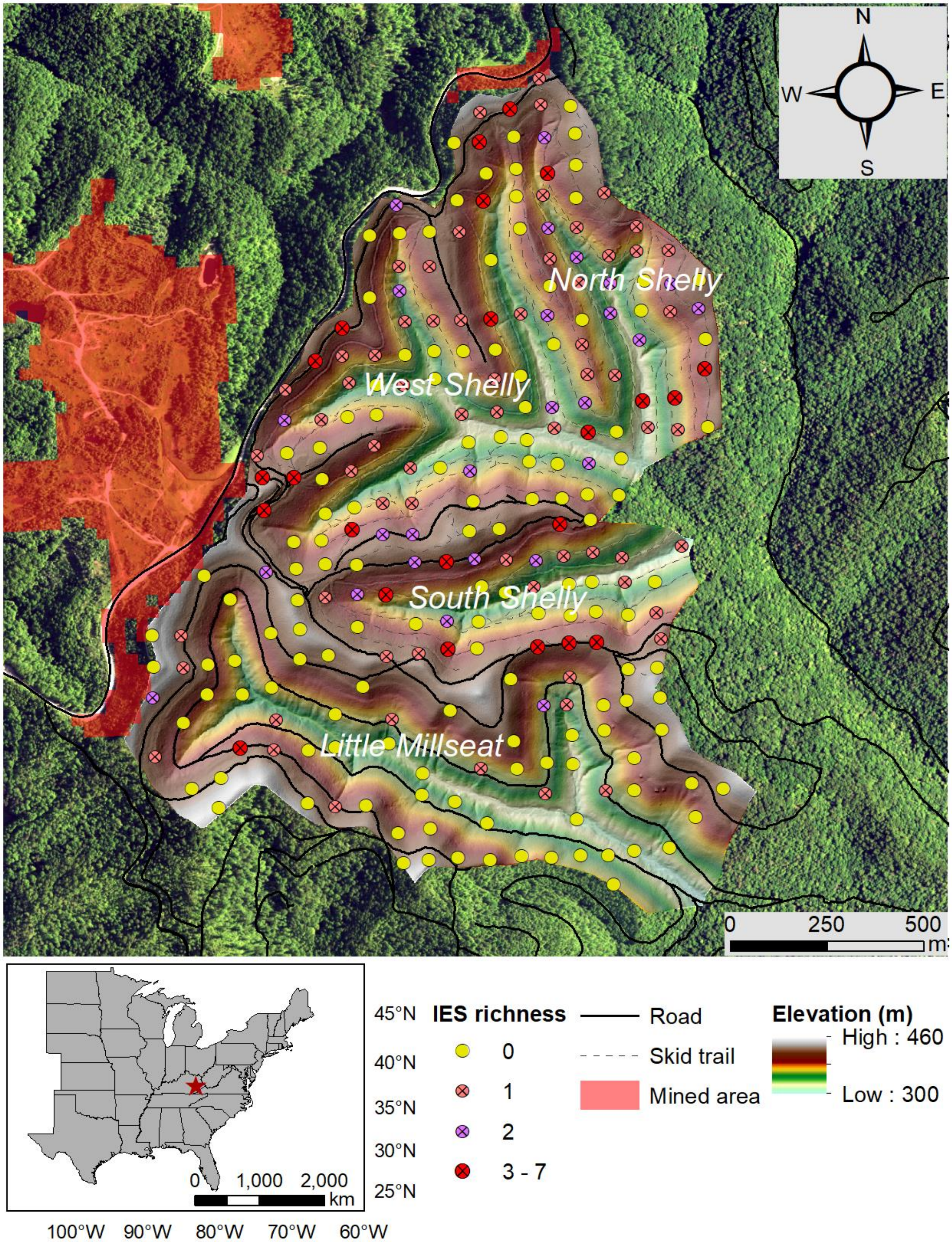

Figure 8 IES richness in sampling plots among all watersheds of the study area

Table 2. *Eight-year post-harvest observations of IES throughout systematic plot sampling scheme*

Scientific Name	Common Name	USDA Code	Observations	Growth Form	Threat level
Ailanthus altissima	Tree of Heaven	AIAL	15	Tree	1
Celastrus orbiculatus	Oriental Bittersweet	CEOR7	1	Vine	1
Dactylis glomerata	Orchard Grass	DAGL	11	Grass	3
Elaeagnus umbellata	Autumn Olive	ELUM	10	Shrub	1
Lespedeza bicolor	Shrubby Lespedeza	LEBI2	19	Shrub	2
Lespedeza cuneata	Bush Clover	LECU	4	Shrub	1
Ligustrum sinense	Chinese Privet	LISI	1	Shrub	2
Lonicera japonica	Japanese Honeysuckle	LOJA	5	Vine	1
Lonicera maackii	Amur Honeysuckle	LOMA6	15	Shrub	1
Microstegium vimineum	Japanese Stiltgrass	MIVI	38	Grass	1
Miscanthus sinensis	Chinese Silvergrass	MISI	12	Grass	1
Morus alba	White Mulberry	MOAL	1	Tree	2
Paulownia tomentosa	Princess Tree	PATO2	1	Tree	1
Poa pratensis	Kentucky Bluegrass	POPR	33	Grass	2
Rosa multiflora	Multiflora Rose	ROMU	5	Briar	1
Schedonorus arundinacea	KY 31 Fescue	SCAR7	29	Grass	3
Sorghum halepense	Johnson Grass	SOHA	7	Grass	1

Threat levels: 1 = Severe, 2 = Significant, 3 = Not on KY – IPC watch list

Table 3. *IES species richness observations in all watersheds*

IES species richness	Little Millseat	North Shelly Rock	South Shelly Rock	West Shelly Rock
0	60	10	18	47
1	12	15	12	26
2	3	8	5	10
3	1	3	5	3
4	0	1	0	5
5	0	0	1	2
6	0	0	0	0
7	0	0	1	1

Table 4. *Hurdle model comparing IES richness among all watersheds*

	Estimate	Std. Error	Z value	Pr(>│z│)
Count model coefficients (truncated Poisson with log link):				
(Intercept)	-0.560	0.429	-1.306	0.192
North Shelly Rock	0.630	0.486	1.297	0.195
South Shelly Rock	1.061	0.467	2.273	0.023*
West Shelly Rock	0.990	0.451	2.197	0.028*
Zero hurdle model coefficients (binomial with logit link):				
(Intercept)	-1.322	0.281	-4.698	<0.001***
North Shelly Rock	2.315	0.465	4.979	<0.001***
South Shelly Rock	1.609	0.420	3.832	<0.001***
West Shelly Rock	1.322	0.349	3.789	<0.001***

Signif. codes:	0 ***	0.001 **	0.01 *	0.05 ·

The variables included in the final models (AIAL, MIVI, IPC1, and IES models) were identified as the significant predictor variables in the control and harvested watersheds separately (Table 5). For the harvested watersheds the IES model identified LiDAR-derived canopy cover, slope steepness, southwestness, and distance to skid trail were negatively related to IES presence while southwestness was positively related (Table 5). Shrub cover was identified as an important predictor variable in the preceding AIC-based model selection with a negative relationship to IES presence, but was not significant in the harvested watershed overall IES presence model results (was not identified in the AIAL model). In the harvested watershed IPC1 model, canopy cover derived from LiDAR, shrub cover, slope steepness, elevation, distance to road, distance to skid trail, and distance to mined areas had negative relationships with severe threat level IES. All but slope steepness were significant predicting variables for IPC1 presence (Table 5). All variables except slope steepness were significant predictors for IPC1 presence (Table 5). Two IES of major concern were singled out to selecting predictor variables for the harvested watersheds. The MIVI model identified canopy cover derived from LiDAR, basal area per plot, shrub cover, TWI, elevation, distance to mined areas, and distance to stream buffer as significant predictors for *M. vimineum* (Table 5). TWI and distance to stream buffers had a positive relationship with *M. vimineum* presence while the other predictors had a negative relationship with *M. vimineum*. The AIAL model identified southwestness, TWI, and distance to skid trails as significant predicting variables for *A. altissima* presence. Southwestness and distance to skid trails had a negative relationship while TWI had a positive relationship with *A. altissima* presence. Distance to mined areas was identified as an important predicting variable in the preceding model selection with a negative relationship to *A. altissima* presence, even though the p-value of the corresponding coefficient was not significant.

Table 5. *Coefficients with standard errors of the predictor variables of the final AIC-based selection models*

	Harvested watersheds				Little Millseat	
Variables	AIAL	MIVI	IPC1	IES	IPC1	IES
Intercept	0.265(2.135)	15.733(5.837)**	18.053(4.823)***	2.966(0.638)***	16.238(8.271)*	10.749(4.883)*
CVL		-0.016(0.008)*	-0.013(0.007) ·	-0.013(0.007) ·	-0.131(0.084)	-0.110(0.049)*
BA		-11.102(5.047)*				
SC		-0.380(0.160)*	-0.208(0.101)*	-0.093(0.058)	-0.911(0.531) ·	-0.514(0.266) ·
SLP			-0.030(0.021)	-0.045(0.020)*		
SWN	-0.732(0.443) ·			0.572(0.247)*		
TWI	0.199(0.120) ·	0.327(0.109)**				
ELV		-0.033(0.013)**	-0.026(0.009)**			
D2R			-0.254(0.123)*		-0.636(0.281)*	-0.443(0.187)*
D2S	-0.518(0.190)**		-0.356(0.154)*	-0.279(0.157) ·		
D2M	-0.622(0.412)	-1.186(0.406)**	-1.003(0.348)**		-0.844(0.398)*	
D2B		0.425(0.120)*				
Psuedo R^2	0.173***	0.347***	0.282***	0.236***	0.460***	0.245***

CVL, BA, SC, SLP, SWN, TWI, ELV, D2R, D2S, D2M, D2B denote the spatial covariates: canopy cover derived from LiDAR, basal area per plot, shrub cover, slope steepness, southwestness, TWI, elevation, distance to nearest road, distance to nearest skid trail, distance to nearest mined area, and distance to stream buffer, respectively. D2R, D2S, D2M, and D2B were log-transformed. Pseudo R^2 value is Nagelkerke (Cragg and Uhler).
P-value significance codes are: 0 '***' 0.001 '**' 0.01 '*' 0.05 '·'

For Little Millseat, there were not enough presence observations of any single IES to be modeled in the GLM framework; so only IPC1 and all IES groups were modeled. The full model addressing overall IES presence was negatively correlated with canopy cover, shrub cover, and distance from skid trail (Table 5). The full IPC1 model resulted in negative correlations between severe threat level IES presence and canopy cover, shrub cover, distance to road, and distance to mined areas (Table 5).

Using Welch Two-Sample t-test on the key predictor variables identified by the IES models for the relationships with IES presence, there was differing results between the control watershed and the harvested watersheds. In the control watershed (Little Millseat), distance to road ($p<0.001$) and canopy cover derived from LiDAR ($p=0.294$) was negatively related to IES presence (Figure 9). Mean distance from road for the sampling plots without IES was 26 meters and mean value for IES presence plots was 11 meters away from the road (Figure 9). A large proportion of the sampling plots that had IES presence and absence were skewed to greater distances from the road. Mean value of canopy cover derived from LiDAR for IES absence was 97% and mean value for IES presence was 94% (Figure 9). A large proportion of the plots that had IES presence and absence were skewed to lower canopy coverage.

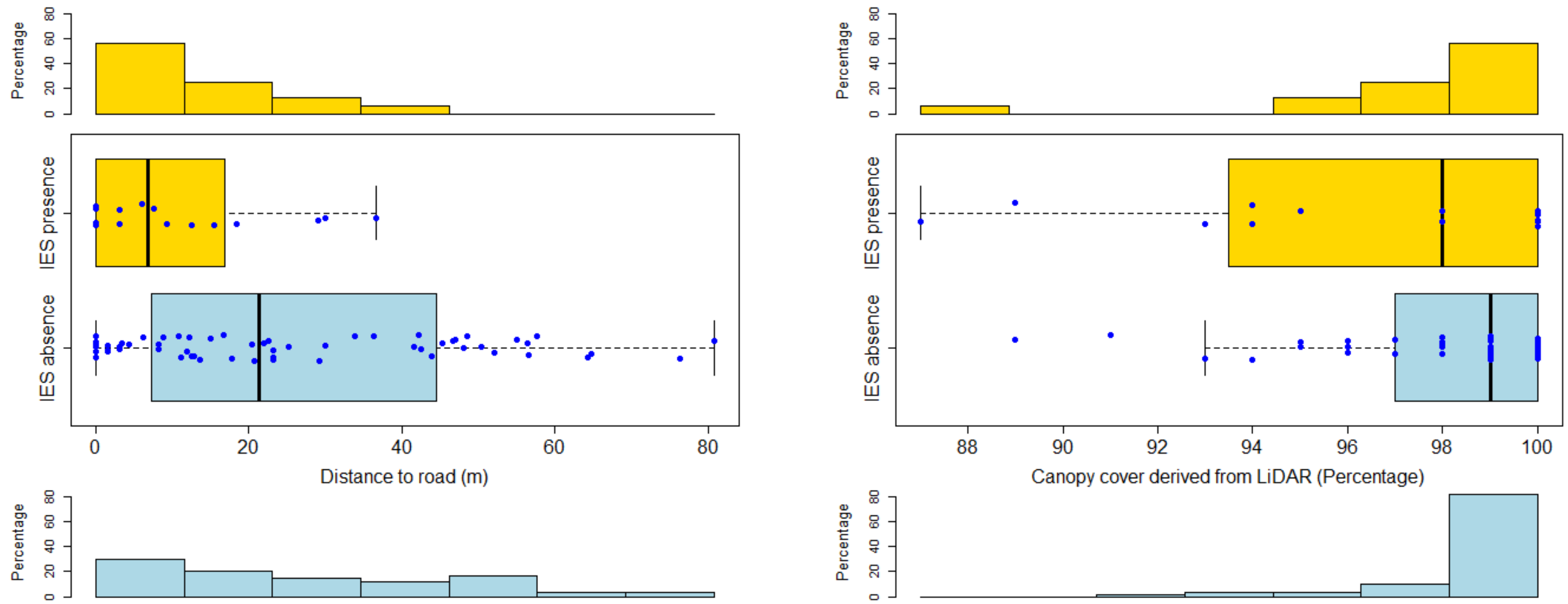

Figure 9 Boxplots and histograms showing the distribution of the key predictor variables for the IES presence group and absence group in Little Millseat. From left to right, the variables are distance to road and canopy cover derived from LiDAR

In the harvested watersheds, LiDAR-derived canopy cover, slope steepness, and distance to trail had a negative relationship with IES presence. The mean canopy cover in the sampling plot without vs. with IES was 76% and 59% respectively, indicating that IES are more likely to be found at sites with significant ground cover exposure. The mean slope steepness for IES absence and presence groups were 25 and 20 degrees, indicating a higher likelihood of finding IES on gentle slopes (Figure 10). Southwestness (p-value = 0.009) had a positively associated relationship in the harvested watershed. The mean values of southwestness for the IES absence and presence groups were -0.145 and 0.138 respectively, indicating higher probability of finding IES on drier slopes (Figure 11). The mean distance to skid trail for IES absence and presence groups were 14 and 12 meters (Figures 11). T-test for the log-transformed distance to skid trail variable had a p-value of 0.001, suggesting skid trails has a significant effect on IES presence. At the individual species level, T-test of proximity to skid trails were not significant for *M. vimineum* and *A. altissima*, mainly due to the fact that there were a significant portion of plots that skewed the mean values of the t-tests (Figure 12). However, the histograms for both species showed that more than 60% of presence plots were within 5 meters proximity to skid trails, while less than 50% of the absence plots were 5 meters away from the skid trail. This suggests a higher probability of finding these two species in the areas closer to skid trails than farther away.

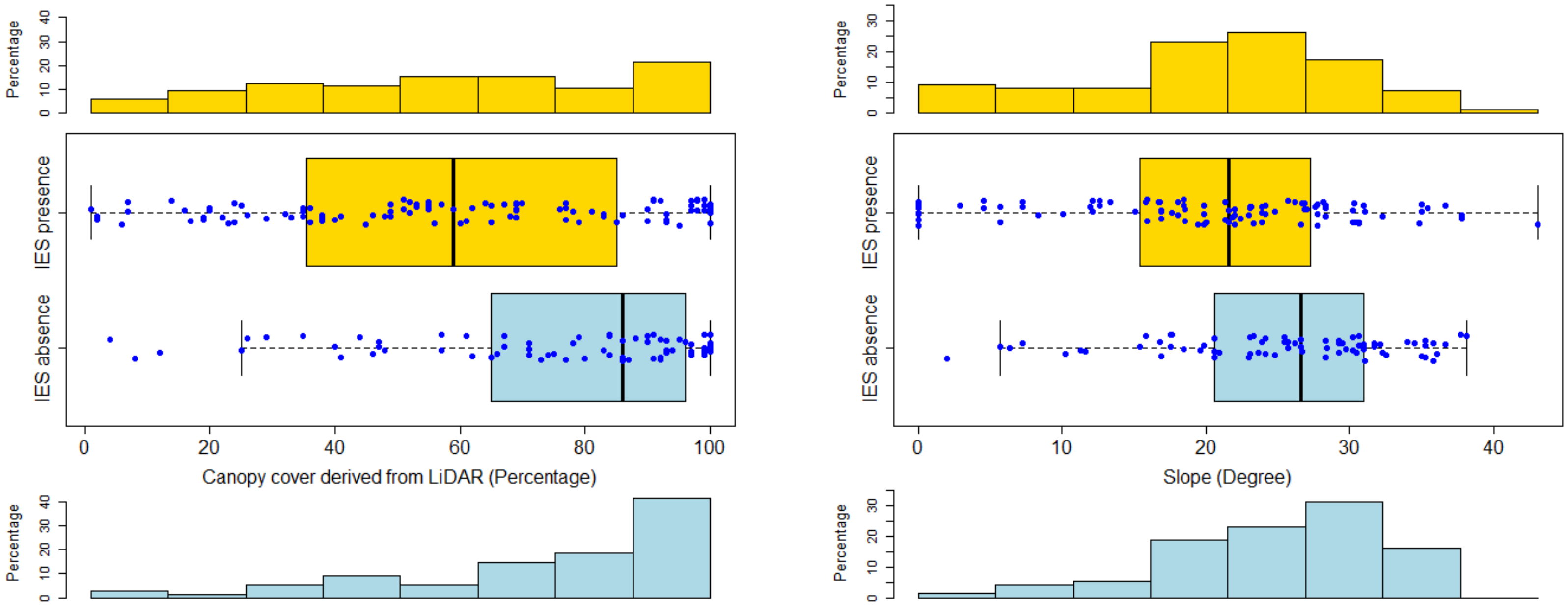

Figure 10 Boxplots and histograms showing the distribution of the key predictor variables for the IES presence group and absence group in the harvested watersheds. From left to right; the variables are canopy cover derived from LiDAR and slope steepness

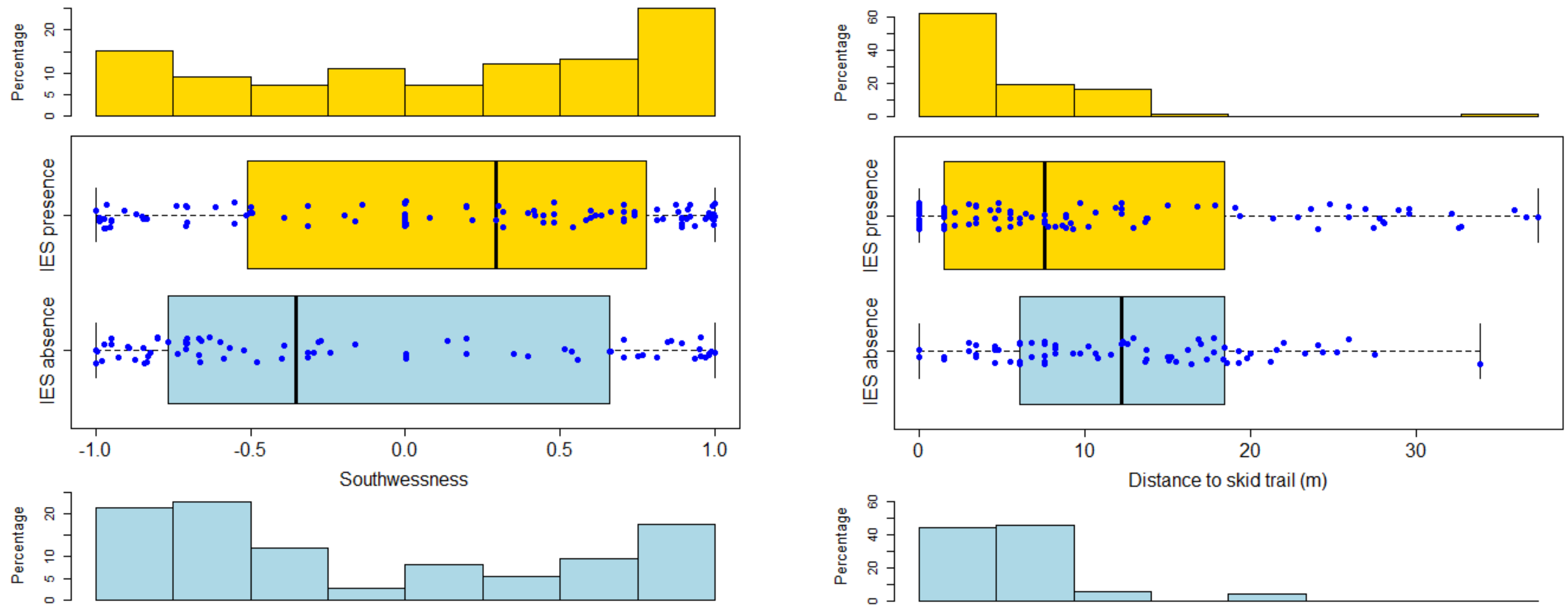

Figure 11 Boxplots and histograms showing the distribution of the key predictor variables for the IES presence group and absence group in the harvested watersheds. From left to right; the variables are aspect (southwestness) and distance to skid trails

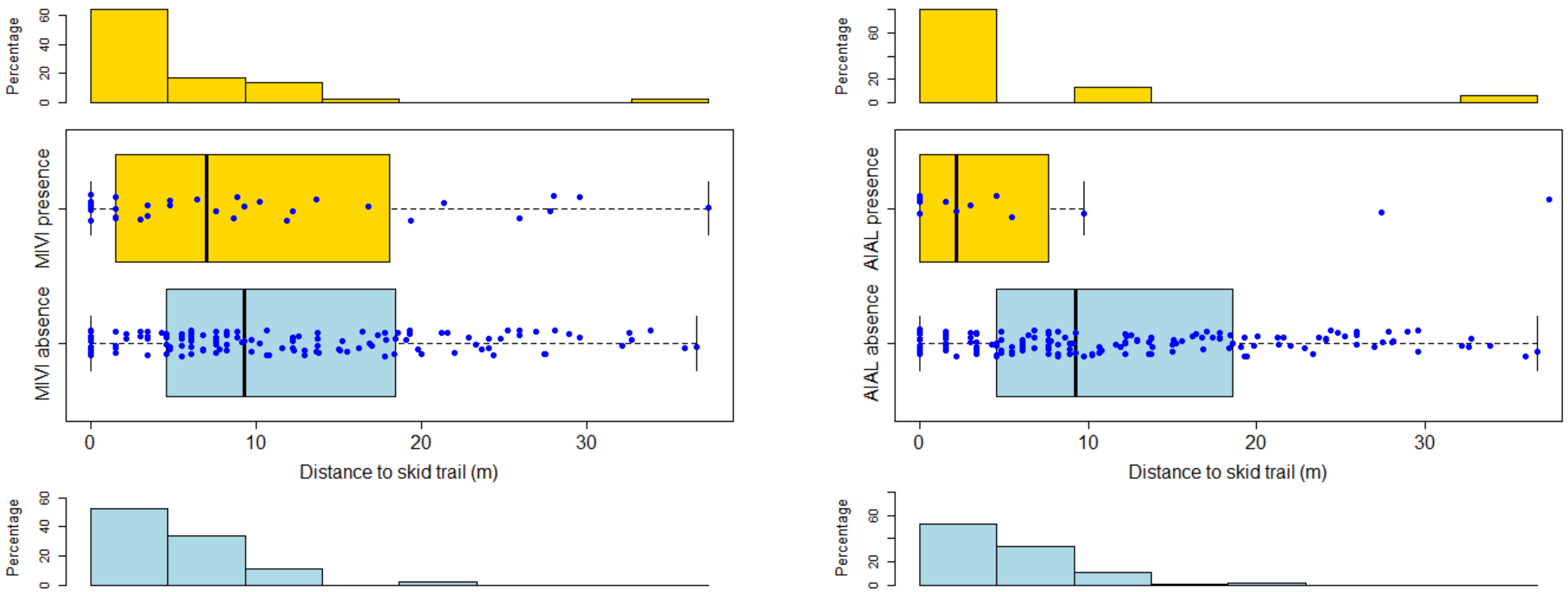

Figure 12 Boxplots and histograms showing the distribution of the key predictor variables for the presence group and absence group for M. vimineum and A. altissima. The variable is distance to skid trails

Higher IES pervasiveness was shown in the one-year post-harvest survey among the sampling plots (Figure 13). IES presence proportion values were lower (15% to 27.5% difference) in all of the watersheds of the eight-year post-harvest survey compared to the one-year post-harvest survey (Table 6). The unharvested watershed had the largest average value decrease of 27.5%, followed by West Shelly Rock (25%), North Shelly Rock (20.7%), and South Shelly Rock (15%) Comparing the one-year post-harvest survey to the eight-year post-harvest survey (Tables 7 and 2, respectively) showed that 10 IES were found in the first-year survey and 17 IES were found in the eight-year survey. All IES found in the one-year post-harvest survey were found in the eight- year post-harvest survey. *A. altissima* and *M. vimineum* significantly decreased in number of observations. *A. altissima* decreased 74.5% from 149 to 38 observations and *M. vimineum* decreased 79.2% from 72 to 15 observations (Tables 7 and 2, respectively). However, some invasive plant species increased observations, including *Elaeagnus umbellate* (5 to 10 observations), *L. bicolor* (4 to 9 observations), *L. maackii* (5 to 15 observations), and *Sorghum halepense* (2 to 7 observations) (Tables 7 and 2, respectively). The number of IES labeled as "severe threat" increased from 8 species to 11 species over time (Tables 7 and 2, respectively), those additional species being *Celastrus orbiculatus, Lonicera japonica,* and *Miscanthus sinensis*.

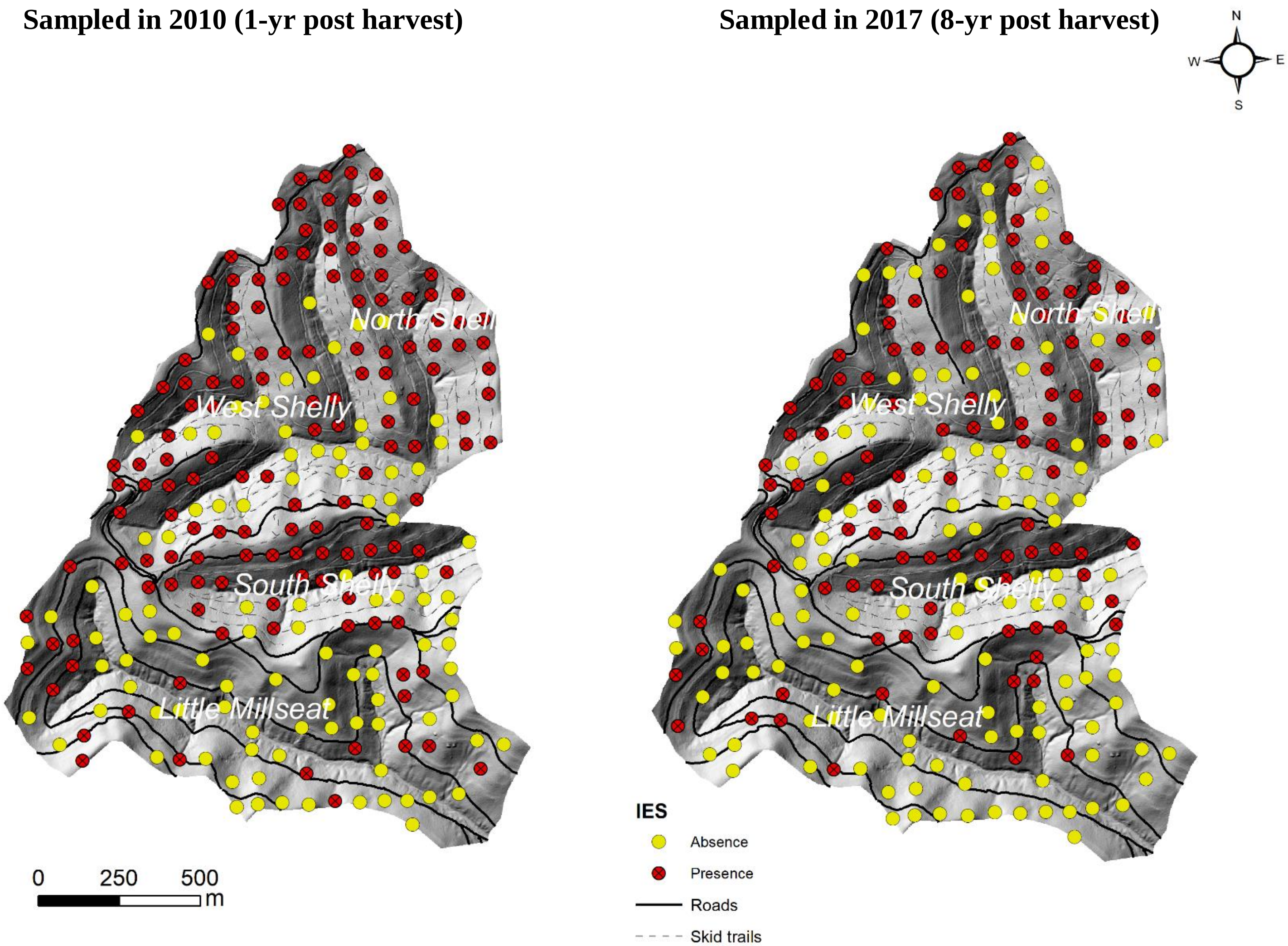

Figure 13 Comparison of IES presence in 1-year post-harvest survey vs. IES presence in 8-year post-harvest survey

Table 6. *Proportion of IES presence of the 1-year vs. 8-year post harvest surveys in all watersheds*

Watersheds	1-year post harvest survey	8-year post harvest survey
Little Millseat	0.289	0.211
North Shelly Rock	0.919	0.730
South Shelly Rock	0.667	0.571
West Shelly Rock	0.681	0.511

Table 7. *One-year post-harvest observations of IES throughout systematic plot sampling scheme*

Scientific Name	Common Name	USDA Code	Observations	Growth Form	Threat level
Ailanthus altissima	Tree of Heaven	AIAL	72	Tree	1
Elaeagnus umbellata	Autumn Olive	ELUM	5	Shrub	1
Lespedeza bicolor	Shrubby Lespedeza	LEBI2	4	Shrub	2
Lespedeza cuneata	Bush Clover	LECU	3	Shrub	1
Lonicera maackii	Amur Honeysuckle	LOMA6	5	Shrub	1
Microstegium vimineum	Japanese Stiltgrass	MIVI	149	Grass	1
Paulownia tomentosa	Princess Tree	PATO2	1	Tree	1
Rosa multiflora	Multiflora Rose	ROMU	7	Briar	1
Schedonorus arundinacea	KY 31 Fescue	SCAR7	8	Grass	3
Sorghum halepense	Johnson Grass	SOHA	2	Grass	1

Threat levels: 1 = Severe, 2 = Significant, 3 = Not on KY – IPC watch list

CHAPTER FOUR: DISCUSSION

1. IES landscape pattern

Results from our analysis support our hypothesis that IES richness and presence would be higher in the harvested watersheds than the control watershed. Even separating overall IES presence into different groups based on KY-IPC and pre-selected species, the control watershed consistently had lower IES proportions and richness than the harvested watersheds. This pattern shows that timber harvesting correlates to higher IES landscape presence in disturbed areas than undisturbed areas. The KY-IPC severe threat level species were found mostly in the harvested watersheds, which is supportive that disturbance facilitates IES establishment. The resulting loss in canopy coverage and loss in biological competition opens the forest floor for invasive plants to establish a meta-population in the disturbed zone. When severe threat IES were found in the control watershed, they were still in relative proximity to disturbed areas (roads and mined areas). This pattern was indicative that disturbance will increase the chances of IES presence even in a mature forested stand. *A. altissima* and *M. vimineum* were found near disturbed areas mostly in the harvested watersheds and in proximity to skid trails. Interestingly, *M. vimineum* was found primarily in North Shelly Rock while *A. altissima* was more evenly distributed among the harvested watersheds (Figure 12). Even though North Shelly Rock had the highest numerical IES presence out of all of the watersheds (73% followed by 57% in South Shelly Rock), it was not significantly different from the control watershed (Little Millseat) in terms of invasive species richness (Table 4). The low IES richness combined with high IES presence could be described by the voracious competing effect *M. vimineum* has on native plant species and other IES (Adams and

Engelhardt 2009). Both *A. altissima* and *M. vimineum* are known to increase in abundance through increased light availability (Kota et al., 2007; Oswalt et al. 2007; Rebbeck et al. 2007) and soil disturbance (Marshall and Buckley, 2008a) that is facilitated by timber harvesting. Our results support this known pattern for these two species as they were primarily found in proximity to skid trails and higher light availability. Results showing the lower IES prevalence in Little Millseat suggest that mature forested stands have less probability of having IES present. Interior forests, or at least healthy forested stands, have been found to have less IES richness and abundance than disturbed and edge forests (Davies and Sheley 2007; Calinger et al., 2015; Beauvais et al. 2016).

Out of the 17 IES found, 11 species were of a severe threat level (Table 2), including *A. altissima*, *M.vimineum*, *L. maackii*, *E. umbellata*, *C. orbiculatus*, *L. cuneata* and *Rosa multiflora*, which are considered some of the biggest threats to Appalachian forests (Kentucky Invasive Plant Council 2013; Butler et al. 2015; Calinger et al. 2015). Many of these severe threat level species that were observed also followed the trend of being found near skid trails and other disturbed areas. Finding higher IES richness in the timber harvested sites follows the notion that while IES are the passengers of environmental change, one IES can act as a catalyst for other IES to colonize the landscape (MacDougall and Turkington 2005; Theoharides and Dukes 2007; Calinger et al. 2015). Higher IES richness can also be explained by the higher amount of disturbance to the soil as the timber operations create more damaged areas.

2. Drivers of IES presence

IES presence is influenced by canopy cover, slope steepness, skid trails, and southwestness in the harvested watersheds and by canopy cover, distance to roads, and shrub cover in the un-cut control watershed. The aforementioned characteristics vary greatly in Appalachian forests, creating landscape heterogeneity that significantly affect IES presence (Kumar et al. 2006). Canopy cover derived from LiDAR was the common factor among all watersheds that had a significant negative relationship on IES presence. The increased light availability reaching the forest floor creates a more desirable environment for plant invaders. A close investigation of the LiDAR-derived canopy cover map shows lower canopy coverage in North Shelly and most of the directly southwest facing slopes. While previously thought to be a product of accidental variance of logging intensity (either as a result of slope steepness or a situational issue), this could be resulting from the resource availability on the southwest facing slopes (Li et al 2011; Wang et al 2013; Gilliam et al 2014; Diaconu et al. 2015). These particular facing slopes have higher amounts of solar radiation, which is going to increase evapotranspiration rates (less water availability), adding more stress to the environment and overall decreasing height growth and leaf area development of the regeneration age class. This may result in a higher chance of invasive plant species being present on the landscape.

It was interesting that shrub cover was significant and negatively correlated of IES presence in Little Millseat and the harvested watersheds. Shrub cover is an important driver in Little Millseat, less so in harvested watersheds, but still a significant variable determining IES presence. In this case, shrubs could be acting as a secondary light filter in all watersheds. The only case that did not have shrub cover as a significant variable was the AIAL model. This is predictable because *A. altissima* is a tree species and can outcompete shrubs. This may be due to overall biological competition in the understory layer in the harvested watersheds as regeneration occurs. The vigorous tree regeneration takes up significant amount of space and increases difficulty for shrub establishment.

M. vimineum was negatively correlated with canopy cover and shrub cover (Table 5) and could have been reducing light levels for *M. vimineum* compared to taller invasive shrub and tree species. A number of environmental variables (e.g., TWI and distance to stream buffer) were also identified as important drivers of *M. vimineum* presence on the landscape. TWI is positively associated with *M. vimineum* presence, indicating wetter areas are more prone to *M. vimineum* colonization. This pattern is supported by TWI (Table 5), skid trail proximity (Figure 12), and skid trails having higher TWI values (Figure 3). There is a possibility that constructed skid trails increase soil moisture due to perched water compared to native adjacent soils, thus improving habitat for *M. vimineum*. *A. altissima* was not influenced by any biological factors (even canopy cover) and was tied to distance to skid trails and southwestness. This is interesting because as an IES that is highly tied to light availability and disturbance (Call and Nilsen 2003), one would think that canopy cover would be a significant factor in the AIAL model (Table 5). Other authors (Rebbeck et al., 2007; Devine 2011) have stated that canopy cover, as well as disturbed areas, are a significant predictor for *A. altissima*. This could be explained by that the harvested watersheds have overall less canopy cover than the control watershed. This indicates a relationship to light availability, but as a tree, lower canopy coverage will have less significance to *A. altissima* presence pattern. *A. altissima* has a significant relationship to skid trails, which this species could be using as a corridor for colonization (Devine 2011).

3. Temporal dynamics of IES response

From the one-year post-harvest survey to the eight-year post-harvest survey, IES prevalence among the watersheds has decreased over time, while IES richness has increased. Whereas Devine (2011) found that IES presence was more likely to be close to disturbed sites and less canopy cover, our study found that disturbance proximity was not the sole significant role in IES presence. Topological and biological factors became more significant in predicting IES presence. These factors, such as landscape water flow (TWI), slope steepness, southwestness, and shrub cover, played a significant role in IES presence eight years after the harvest. This could be due to timber harvesting effects that play a crucial, almost overpowering, role in regeneration those first few years after the harvest (Devine 2011). As time goes on, as indicated by our results, the environmental conditions can dictate both native and non-native species landscape patterns. The effects from the environmental conditions can influence the biological conditions that could have an effect on IES presence.

The most notable change of the IES patterns is the decrease of *M. vimineum* and *A. altissima* prevalence between the two time periods. The massive prevalence reduction suggests these two species possess a lower shade tolerance than what the literature suggests (Rebbeck et al., 2007; Adams and Engelhardt 2009). *A. altissima* and *M. vimineum* are known to inhibit succession to a large degree in an area once they get established (Oswalt et al. 2007). But our results suggest quite the opposite. This could be accounted for by the high amount of competition over several years by native species.

Dostál et al. (2013) found that areas with a longer invasion history will stabilize eventually to native species. Similar to this project, they initially found a significant reduction in native species regeneration that accompanied the explosion of invasive species abundance. After ~30 years the invasive species were still present, but not having a stronghold on the landscape. However, it should be noted that some of our study plots had high IES density (10 plots with ≥ 50% cover). This indicates while the majority of the IES will be outcompeted by native species, there will be a few patches of IES that will have taken a stronghold. This could become problematic after several harvest rotations where there is enough propagule pressure from IES to outcompete natives on a landscape scale. For example, Johnson et al. (2015) and Oswalt et al. (2007) observed *M. vimineum* inhibiting important native timber species regeneration (*Quercus* spp.) on different timber harvesting managed lands. The USDA published a report on the central Appalachian forest ecosystem vulnerability stated that invasive species have a negative effect on native species regeneration (Butler et al. 2015). While there was a significant decrease of two major IES, there were increasing numbers of other threatening invasive plants that have been shown to have complicated native regeneration. *L. japonica, C. orbiculatus, M. sinensis* were the new additions over the years, with the vines causing the most harm in other parts of Kentucky as well as the Midwest and Appalachia (Calinger et al 2015, Butler et al. 2015).

4. Management Implications

This study will ultimately aid in streamlining invasive species management for restoration and logging operations. The results show that in just a few years overall IES prevalence decreases if BMP’s are properly followed. This would indicate that monitoring invasive plant species development after harvesting should be done to determine if control measures should be postponed until the outcome of native species competition is evaluated. The results also show that IES will most likely be found where low canopy cover, proximity to disturbance, and southwest facing slopes convene on the landscape. These findings can give restoration services an efficient protocol to identify IES hotspots and remove invasive plant species in a landscape (Lathrop and Bognar 1998). Knowing the predictor variables for IES and addressing those to individual areas can minimize the efforts controlling invasive plant species and the costs that can otherwise greatly escalate with increase of IES prevalence (Theoharides and Dukes 2007). Our results provide a modeling structure to restoration services and timber industries to aid in developing management protocols for both restoration and logging (Aurambout and Endress 2018; Bradley et al. 2018). Lastly, these results showed that LiDAR-derived and Field-estimated canopy cover are similar, though LiDAR provides more accurate canopy cover readings over a landscape, not just at certain points. While field verification and “ground-truthing” will still be relevant, LiDAR can be more widely used for the technology’s accuracy in addressing landscape-wide studies.

5. Limitations and Future work

Field data collection had a few limitations that should have been addressed. Field estimation of canopy cover was visually estimated and could have had more accuracy using a densiometer. We did not distinguish different parts of the skid trail and did not survey solely the skid trails and landings as Devine (2011) did. Devine (2011) was able to test which parts of the skid trails IES were present and was able to procure a more accurate status of IES on the harvested landscape. This was not completed on our part due to time limitations and the thought that it could skew the results. Most plots within the harvested watersheds were no more than 70 meters away from a skid trail. This could also skew our results for harvested watersheds. A counter statement is that due to the density of skid trails (a disturbance) in a timber harvest, probability of IES presence will remain high. Lastly, pre-harvest locations and abundance values of IES were not considered when running our analysis due to not having access to the exact data.

Since this is only eight years post-harvest, future surveys of sampling sites would be beneficial to understand whether IES are going to significantly affect the regeneration rates of important hardwoods of eastern Kentucky forests. Future work could theoretically determine which tree species recruitment is going to be significantly reduced from IES competition based on landscape positioning. For example, pines and some oak species are found in higher abundances on drier slopes than wet slopes. Depending on the species composition and landscape positioning, IES could outcompete a significant portion of trees to lead to lower timber product output.

Future work would consist of continuing with the periodic surveys of long-term IES response to timber harvesting. After a few more surveys, there could be a complete picture of how IES respond to timber harvesting. Other studies that could stem from this would be to study native species regeneration for timber production or to identify the interaction between native and non-native species regeneration.

6. Conclusion

In conclusion, the central question investigated in this research is how invasive plant species interact with a timber harvested landscape over time in a mixed mesophytic forest. The results suggest that overall IES prevalence has decreased but invasive plant species richness has increased over time. In addition, while harvesting effects and disturbance proximity still play an important role in an eight-year post-harvest landscape, environmental characteristics have also taken precedence in predicting IES presence. These results indicate that invasive plant species eradication may not be conducted immediately after the harvesting, and when needed, can primarily target IES hotspots where low canopy cover, proximity to disturbance, and southwest facing slopes convene on the landscape.

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