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Incorporating Risk of Reinvasion to Prioritize Sites for Invasive Species Management

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ABSTRACT: The relationship between landscape pattern and the distribution and spread of exotic species is an important determinant of where and when management actions are best applied. We have developed an interdisciplinary approach for prioritizing treatment of harmful, nonnative, invasive plants in National Park landscapes of the Mid-Atlantic USA. The approach relies upon a detailed model of reinvasion risk that combines information on: (1) global factors representing park-level infestation from seed and sprout, (2) landscape factors including disturbance-based spread vectors and neighborhood seed density, and (3) local factors determining establishment probability based on habitat suitability. Global seed rain estimates are derived empirically from park inventory data and modified by information on species reproductive strategies. Landscape-level propagule pressure is modeled spatially using species life history characteristics including dispersal attributes, connectivity to nearby plant populations, and increased propagule pressure through disturbance. The local-scale habitat suitability model uses a Mahalanobis distance approach, parameterized from plant inventory plot data and GIS-based data on plot wetness, land cover, slope, radiation, and soil characteristics. We illustrate the model for *Ailanthus altissima* (tree-of-heaven) in Antietam National Battlefield Park. The results of the *A. altissima* modeling highlight regions of the park where eradication would be most prudent and feasible based on current infestation patterns and landscape heterogeneity. Although the success of different treatment modalities is often considered in invasive species management, a spatially explicit assessment of likely treatment success is rarely undertaken. Our approach provides a valuable tool to assist natural resource practitioners to prioritize management options in confronting biological invasions.

Index terms: *Ailanthus altissima*, Antietam, decision support tool, invasive plants, National Parks, species distribution models

INTRODUCTION

Invasive species are creating major ecological and economic problems around the world (Mack et al. 2000; Pimentel et al. 2005; Vila et al. 2011). Even relatively pristine and protected ecosystems are not immune. For example, over 700 nonnative species have been found inside U.S. national park borders, and it is estimated that more than 10,522 km² (approximately 5% of park lands) are dominated by nonnative, invasive plant species (2008 EPMT annual report). Civil War battlefield parks in our region of the northern Virginia and Maryland Piedmont and Appalachian Ranges have seen a dramatic increase in exotic plants from 50 years ago when Japanese honeysuckle (*Lonicera japonica* Thunb.) was viewed as the only serious threat to native vegetation (Fleming and Weber 2003). Ecosystem services compromised by plant invasions include aesthetics of the visitor experience, bird and mammal watching, preservation of historic structures, and many more (Celesti-Grangy and Blasi 2004; Drummond 2005; Wainger et al. 2010). Given the breadth of the problem, both in terms of species and land area, combating biological invasions has become a priority of natural areas management.

Removal of nonnative invasive plants can provide a number of ecological and

economic benefits, as removing the exotic species may restore native ecosystems and their associated ecosystem services. However, public agencies such as the National Park Service (NPS) rarely have enough resources to treat all exotic species or the entire landscape. Instead, managers must make difficult decisions about which species or locations are treated first. Treatment options are often prioritized based on species characteristics (e.g., Ou et al. 2008; Randall et al. 2008) with the goal of completely eradicating the highest priority species. This uniform approach does not consider explicitly the importance of spatial heterogeneity across the landscape, which may affect the impact of the exotic species or the outcome of management activities (Hamilton et al. 2006). A more comprehensive approach would focus on weed populations in addition to species traits (Skurka Darin et al. 2011). This consideration would allow eradication efforts to be prioritized in a spatially explicit way, based on factors such as the potential of each population to spread, including to reinvade previously treated sites.

We present the reinvasion risk portion of an interdisciplinary, spatially-explicit model for assessing potential treatment options for nonnative, invasive plants. The ultimate goal of most invasive species management is to restore a site to its natural function,

structure, and composition. Many excellent reviews are available of the types of criteria that should be considered to achieve this goal (e.g., Cipollini et al. 2005; Hiebert et al. 2009; Sebert-Cuvillier et al. 2010). In this paper, we focus on one component of this decision-making process that is integral to maintaining a minimum post-treatment cover of invasive species. We outline a multi-scale approach to assessing risk of reinvasion of the treated species that combines information about species biology, habitat suitability, disturbance, and spatially explicit estimates of propagule pressure. The approach applies sound ecological principles to this important determinant of treatment success at an immediate level of complexity that can be reasonably considered by management agencies. We provide an example of the approach to describe the spatial pattern of reinvasion risk for *Ailanthus altissima*

(Mill.) Swingle (tree-of-heaven) in Antietam National Battlefield Park.

METHODS

Study Area

Antietam National Battlefield Park (Antietam NBP) is a 1320-ha mixed land-cover landscape located 110 km from Washington, D.C. in the NPS's National Capital Region (Figure 1). It receives approximately 200,000 visitors per year. The park was established in 1890 and is mandated to preserve the landscape as it was during the Battle of Antietam in a mixture of open fields and small woodlots. Forest cover (temperate deciduous forest) comprises 24% of the total area of the park (Townsend et al. 2009), with significant forested areas occurring along Antietam

Creek on the west side of the park and the Potomac River east of the park. The land surrounding the park is a mixture of mostly agricultural with some residential and forested areas. Consideration of the landscape context is especially important to understanding spatial processes that contribute to biological invasions in small, fragmented, suburban parks such as Antietam NBP (Lookingbill et al. 2007).

The park is divided into 53 management units with relatively homogeneous land cover for administrative and logistical purposes (Figure 2). The management units are bordered by fixed references such as roads, trails, and streams, and the units range in size from 11 to over 800 hectares (average 31 ha). Park-owned land not contiguous with the rest of the units in the park was excluded from the analysis. Like many National Parks, Antietam NBP

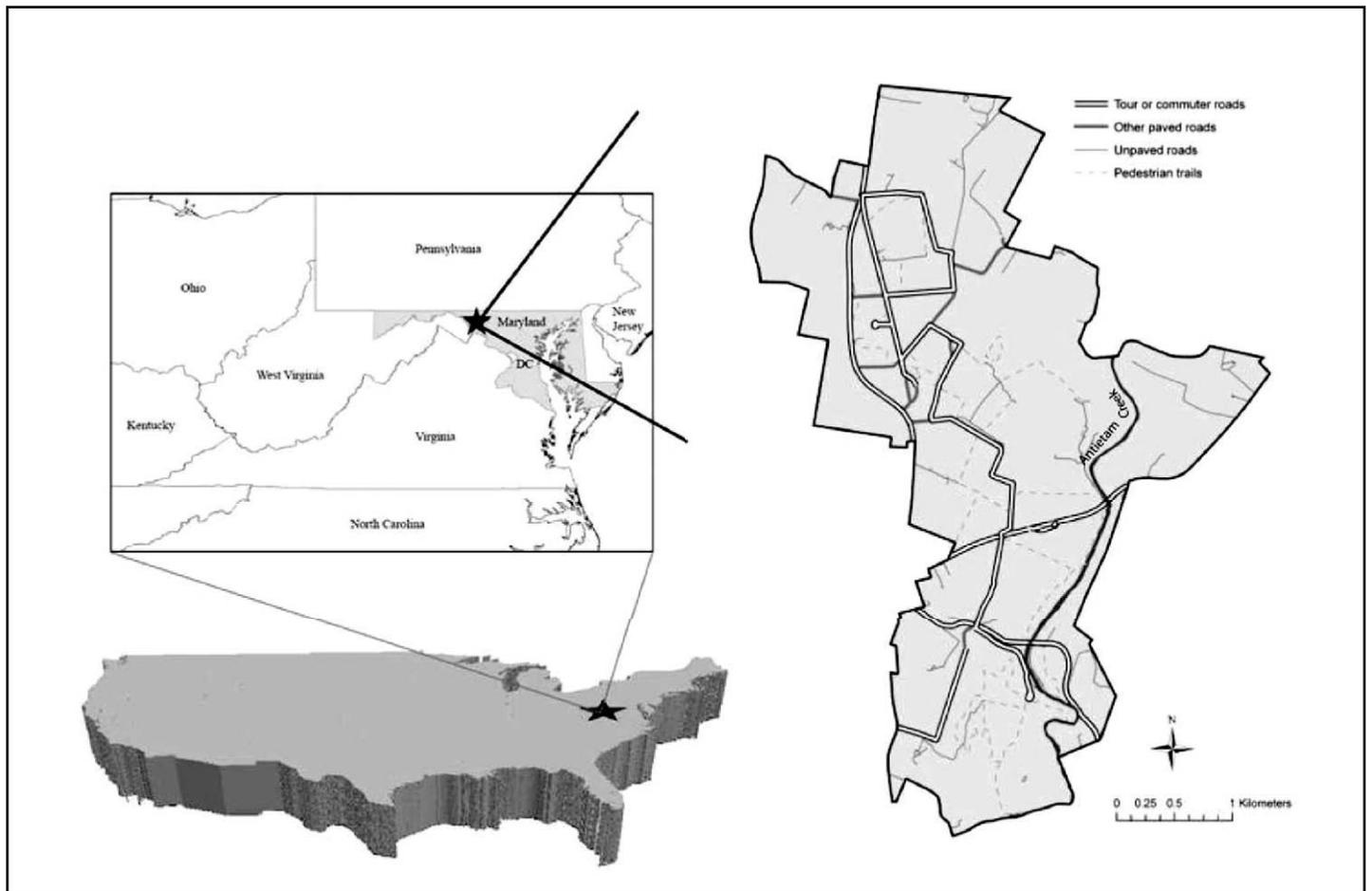


Figure 1. Map of Antietam National Battlefield Park located 110 km from Washington, DC. The roads, trails, rivers and park boundary were important inputs in calculating the spatially explicit risk of reinvasion at a site.

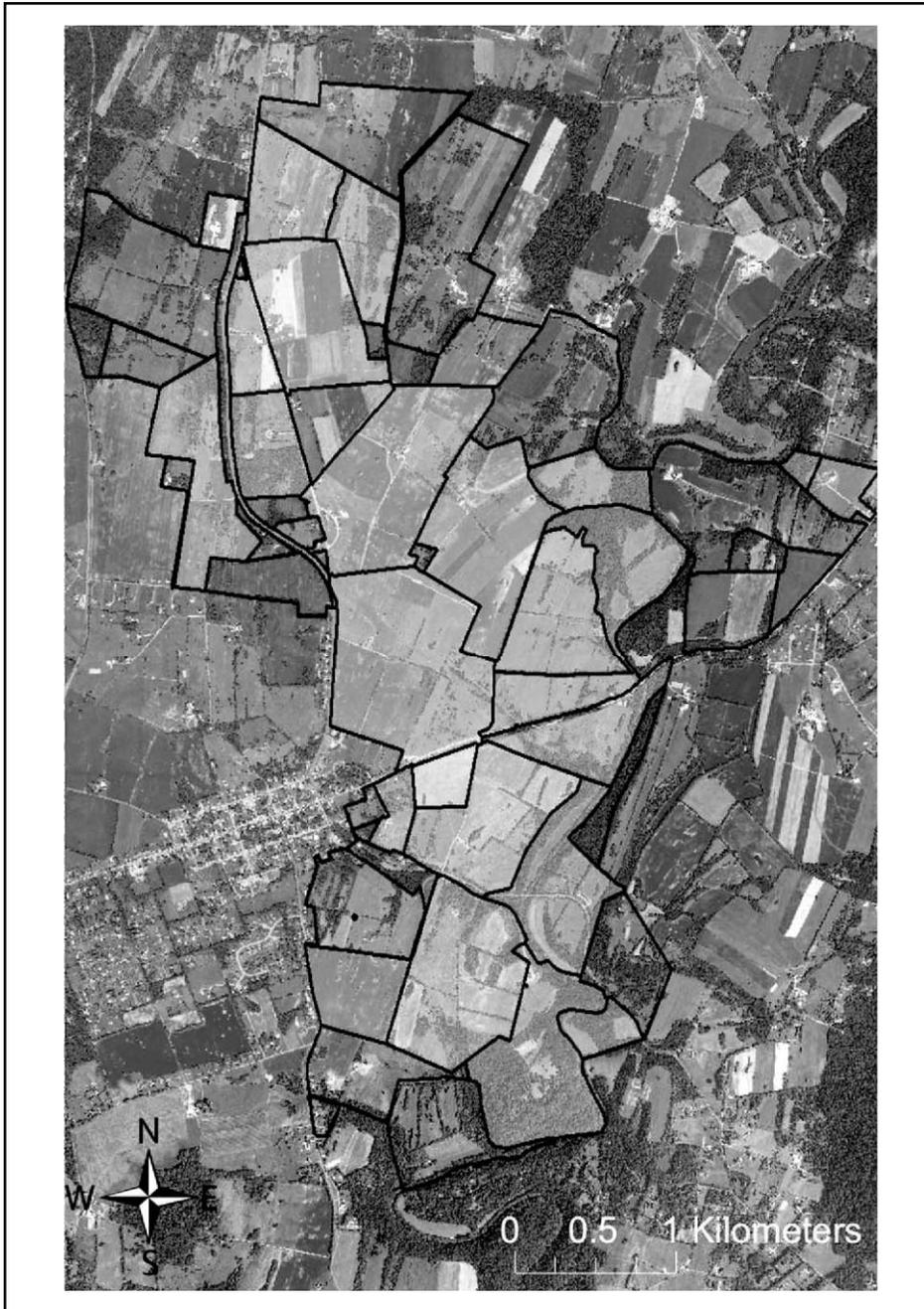


Figure 2. Management units for the Antietam National Battlefield range in size from 11 ha to 800 ha. Lighter shaded units are those that are owned and administered by the park. Although the model was built for the entire park, only the 25 lighter shaded units were ranked in the final risk map.

does not currently own or manage all of the land delineated in its legislative boundary. Only units currently owned and managed by the park were ranked in the model (25 units shown in lighter shading in Figure 2). However, because data were available for the entire legislative boundary, this extent was used for modeling purposes.

Focal Species – *Ailanthus altissima*

Although many natural area managers must confront challenges associated with multiple species potentially reinvading a treated site, we present a simplified case study for a single focal species: *Ailanthus altissima*. For multi-species assessments,

the model we present here could be run multiple times, and the risk of reinvasion assigned additively or as ranked categories where the highest rank was assigned to sites with either high risk for a single species or moderate risk for more than one species (Jurado-Exposito et al. 2003).

Ailanthus altissima is native to Central China. It was first introduced in 1784 by a gardener in Philadelphia, Pennsylvania (Fryer 2010), and is now found throughout most of the continental United States and eastern Canada (USDA Plants database). This highly pollution- and drought-tolerant deciduous tree thrives in urban environments (Rank 1997). It is commonly found in the eastern United States within oak-hickory (*Quercus-Carya*) and maple-birch-beech (*Acer-Betula-Fagus*) forest communities (Fryer 2010). Fruits are thin, flat, wind-dispersed samaras (Dirr 1998) and may disperse via water (Kowarik and Saumel 2008). *Ailanthus altissima* is photosynthetically efficient in high light, grows rapidly, and suppresses competition with allelopathic chemicals (Marek 1988; DeFeo et al. 2003). It is tolerant of a wide range of ecological stresses and resprouts vigorously when cut, making its eradication difficult and time consuming (Fryer 2010).

Information on the distribution of *A. altissima* within Antietam NBP was taken from a vascular plant inventory of the park (Engelhardt 2005). As part of the inventory, presence of every species was recorded in each of 78 forest plots located within the contiguous park area. We supplemented the inventory data with an additional 12 plots that were strategically located in smaller forest patches to achieve a wider range of patch size and connectivity (Minor et al. 2009). Sampling protocol followed the procedure established by the North Carolina Vegetation Survey (Peet et al. 1998), which is also widely used by the Forest Service and The Nature Conservancy. Sampling locations were 0.04 ha in size (20 m × 20 m).

Data on past management of *A. altissima* in the park were gathered from the National Capital Region Exotic Plant Management Team (EPMT) and used to estimate the

baseline risk for reinvasion. From 2001 to 2008, over 300 separate visits were made by the EPMT to *A. altissima* infested sites. Of the data kept for these visits, we found 27 occurrences of treatments that were followed by monitoring one to four years later. These records allowed us to gauge site reinvasion as a measure of treatment effectiveness (i.e., number of acres recorded as invaded in post-treatment monitoring divided by the pre-treatment acreage of infestation). This baseline estimate represents response from multiple different treatment modalities with the most common being “hack and squirt” (hacking into the bark of the trunk to expose vascular tissue to which herbicide is applied). Other methods include the application of herbicide on basal bark (no cutting required), cut stump (the tree is first cut and removed), and foliar tissue (leaves).

The Model

The reinvasion of a species following eradication from a site is governed by many of the same forces that shape invasion processes in general. Our approach to quantify reinvasion risk used data on

historical treatment effectiveness to provide a mean baseline risk for the park. We then distributed risk spatially by ranking the 25 management units based on reinvasion risk factors at three hierarchical scales: (1) global factors representing park-level infestation from seed and sprout, (2) landscape factors including disturbance-based spread vectors and neighborhood seed density, and (3) local factors determining establishment probability based on habitat suitability (Figure 3). Scores for each factor were combined and standardized by the mean value from the EPMT treatment data to provide our final estimates of overall reinvasion risk following an eradication effort.

Global Factors

Each management unit was assigned an index for reinvasion risk at the global level based on two measures: (1) global propagule pressure, and (2) species reproductive strategy. Global propagule pressure is a measure of the relative abundance of an invasive species within the region. Landscapes that have high populations of an invasive species in the general region

will have high likelihood of being invaded or recolonized after treatment. Areas of lower overall propagule pressure with fewer populations in the region make preferred targets for management actions based on the strategy that the best chance of successful eradication is in the initial stages of species establishment (Mack and Lonsdale 2002). We derived a species abundance estimate for the entire Antietam NBP by multiplying the percent of the 90 forest inventory plots (described above) on which *A. altissima* was observed by the percent of the park that was forested. We assumed that reproductively mature trees would be found only in forest habitat and not in other, more managed, habitat types of the park such as croplands, pastures, and developed areas. This estimate of global propagule pressure can theoretically range from 0, if the species is absent from the park or there is no forest cover, to 1 if the park is completely forested and the species is everywhere.

The second global-level parameter included in the model was a measure of the reproductive strategy of the species being targeted. We created a reproductive multiplier based on three factors: seed reproduction, re-

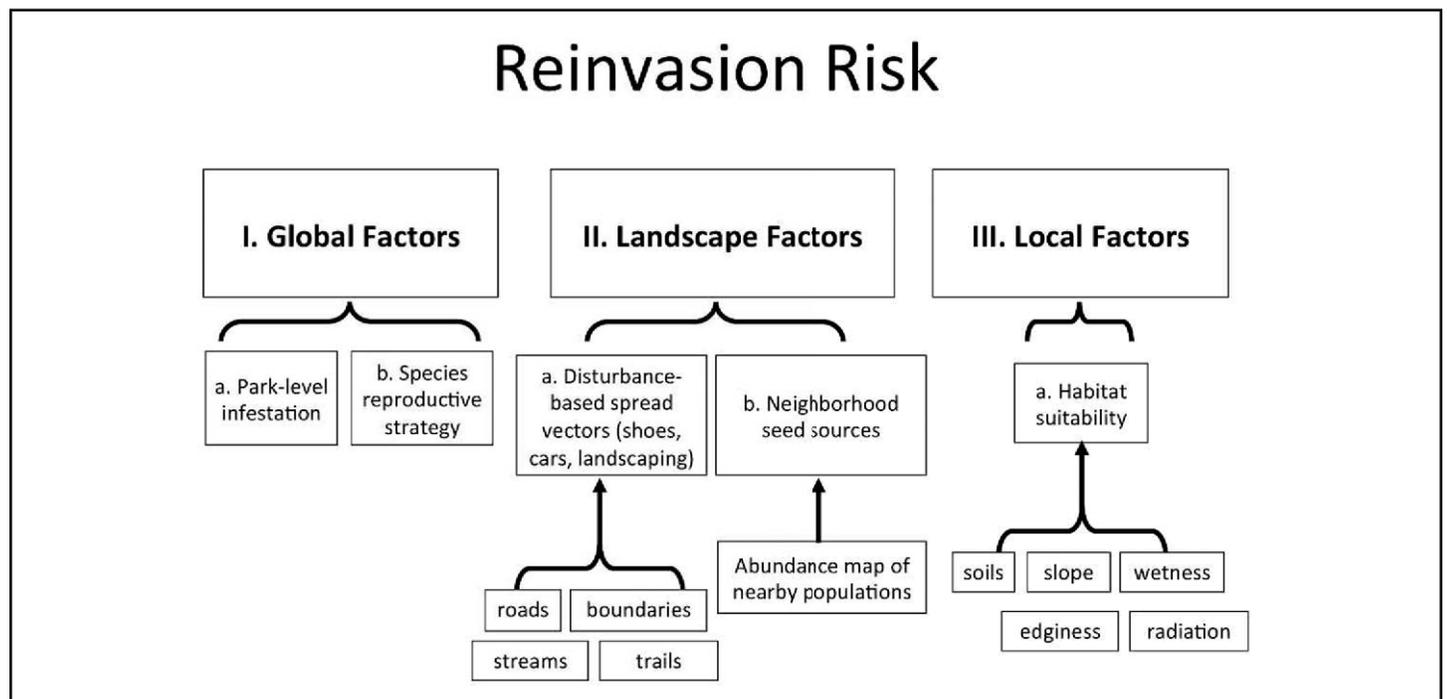


Figure 3. Overview of reinvasion model. The risk of a species reinvading a site after treatment is a function of factors at three hierarchical scales: global, landscape and local. Information from the bottom-most layers in the diagram was combined in the model to represent the multiscale pressures that determine reinvasion of a treated site.

sprouting, and seed banking (Appendix A). Because *A. altissima* reproduces asexually but does not seed bank, it was assigned a final score of 2 (1 for seed production, 1 for re-sprouting, and 0 for seed banking). The estimate of global propagule pressure from the forest inventory data was multiplied by this score. From a management perspective, this park-level variable allows an entity like EPMT the ability to evaluate risk among multiple parks and multiple species within their jurisdiction. For example, *A. altissima* has been reported within 14 national parks in Maryland and Virginia alone (EDDMapS 2012), all at differing abundances and, therefore, differing risks of reinvasion according to the global propagule pressure parameter.

Landscape Factors

We included two measures of landscape-level propagule pressure: (1) neighborhood pressure (the potential seed sources surrounding the treatment site), and (2) disturbance-mediated spread from roads, trails, the park boundary, and streams.

Neighborhood propagule pressure was calculated as the distance-weighted sum of seed contributions from all other cells on the landscape to each 30-m cell. Cells were later aggregated to the scale of the management unit as described below. Ideally, detailed spatially explicit distribution data for the species of interest would be used to estimate seed contributions. However, these data are rarely available, and even where spatial inventories exist, this information becomes quickly outdated for fast-spreading, highly invasive species of concern. Our study of *A. altissima* in Antietam NBP provides an example of the common case where a detailed map providing complete spatial coverage of the species was not available. Therefore, we classified the landscape in and around Antietam NBP according to habitat suitability for *A. altissima* (see description below), with the assumption that areas with greater habitat suitability would produce more seed. We expanded the habitat suitability map 100 m beyond the park boundaries to account for some seed rain from neighboring lands.

Estimating the maximum dispersal distance and form of the kernel for a specific invasive species are time/labor intensive activities (e.g., Martinez and Gonzalez-Taboada 2009). In the absence of such data, literature reviews plus well-defined and stated assumptions are useful to assign kernels for predictions of potential neighborhood spread. These kernels generally reflect the observations that the majority of seeds fall close to home. We selected a negative exponential distribution with a tail distance of 200 m for representing the dispersal kernel of *A. altissima* (Figure 4). Landenberger et al. (2007) determined that exponential kernels provided the best fit between *A. altissima* dispersal observations and predictions, and other literature suggests a maximum dispersal distance ≤ 200 m for the species (Kota 2005; Landenberger et al. 2007), which is generally on the order of other wind dispersed trees (*Acer* spp. Dunn et al. 1991; He and Mladenoff 1999; *Fraxinus americana* L., He and Mladenoff 1999). Based on the above, we propose that a majority of propagules (approximately 99%) fall within 200 m of their source. Therefore, the probability (P) of a seed dispersing a given distance is calculated as:

$$P = -e^{\theta d} \quad \text{Eq. 1}$$

where d is the distance and θ is a decay coefficient. Setting $P = 0.01$ for $d = 200$ m, we can solve for θ , and use this value ($= -0.02 \text{ m}^{-1}$) and the distance between focal (i) and source (j) cells to calculate P_{ij} for any combination of cells on the landscape.

Neighborhood propagule pressure was then calculated for each 30-m grid cell in the park using a moving window analysis:

$$NPP_i = \sum HSI_j * P_{ji} \quad \text{Eq. 2}$$

where NPP_i is the neighborhood propagule pressure for the focal pixel i . The contributions from all other pixels j on the landscape (to focal pixel i) were calculated as the product of the habitat suitability index at pixel j (HSI_j) and the probability (P_{ji}) of a seed dispersing from j to i . Thus, locations that were barely suitable for *A. altissima* (i.e., HSI_j values close to zero)

contributed fewer propagules than sites of high quality for the species (e.g., neighboring forest edge areas). It is important to note that neighborhood propagule pressure does not take into account the suitability of the site itself (i.e., HSI_i). In this part of the model, habitat suitability is only used to weight the strength of propagule pressure from neighboring contributing pixels. Final scores of neighborhood propagule pressure were relativized from 0–1 based on the maximum values observed for the study region.

We included a separate measure of disturbance-mediated spread (i.e., landscape pressure within the management unit that considered dispersal along roads, trails, park boundaries, and waterways). The fat-tails that are characteristic of most dispersal kernels highlight the importance of infrequent, long-distance dispersal events that cannot be modeled by diffusion processes alone (Clark 1998). These events become more common along roads, trails, and other linear elements associated with landscape disturbances (Tyser and Worley 1992; Von derLippe and Kowarik 2007). Because *A. altissima* seeds remain viable in water, streams provide another potential long-distance dispersal pathway in the park (Kaproth and McGraw 2008; Kowarik and Saumel 2008), and the importance of flood disturbance in promoting plant invasions has been well documented for the nearby C&O Canal National Historical Park (Pyle 1995). Roads, trails, and streams all are well represented in Antietam NBP (Figure 1). The park boundaries were considered an additional potential source of propagules, because private property abutting the park boundary may contain untreated populations of the species and/or the boundaries may serve as dumping grounds for yard waste containing source material.

We assigned each management unit in the park a probability of receiving additional seeds based on the weighted density of roads, trails, streams, and park boundaries found in the unit. The approach is similar to that proposed by Dullinger et al. (2009), and we used default values similar to those provided by Williams et al. (2008) to assign weights for *A. altissima* (Table 1). The busiest roads – those on

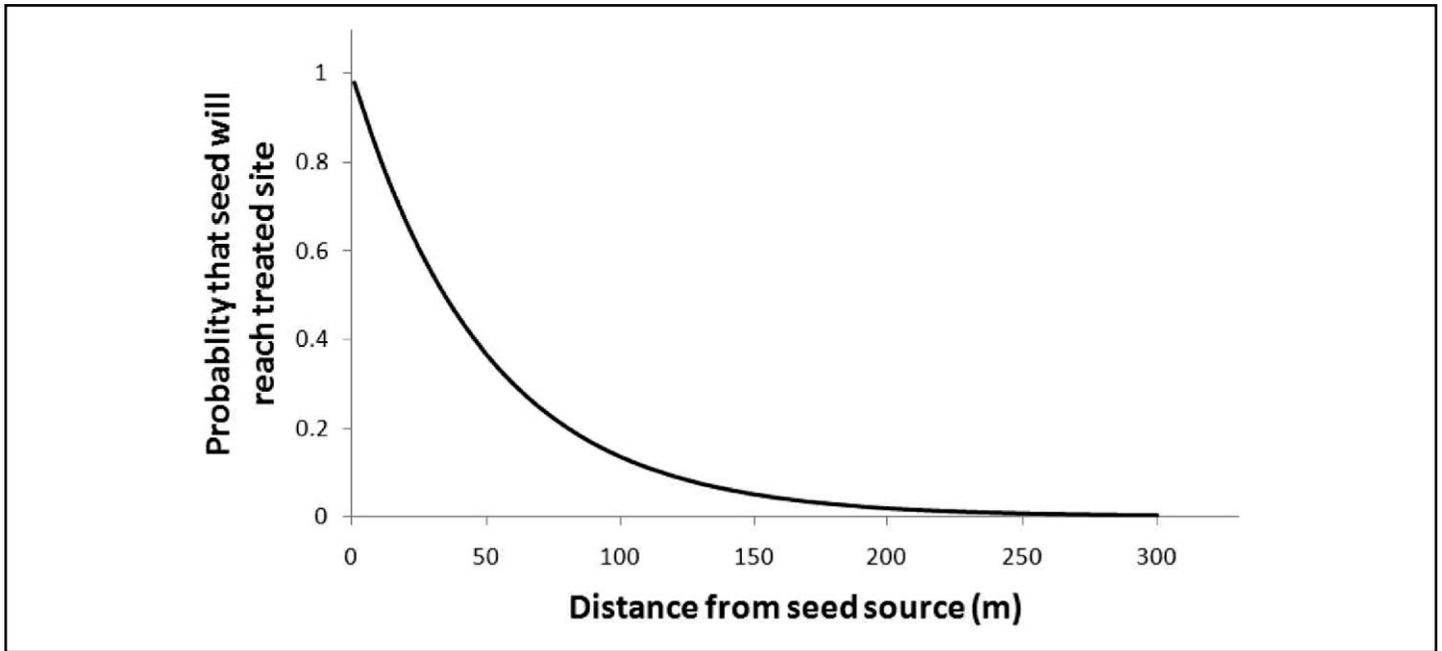


Figure 4. Neighborhood dispersal kernel for *A. altissima* corresponding to $P = 0.01$ for $d = 200$ m.

the historic tour routes or roads used by local commuters – were given a weight of 4. Other paved roads were assigned a weight of 3. Unpaved roads and streams were assigned a weight of 2, and trails and park boundaries were assigned a weight of 1. The weighted sum of road, trail, park boundary, and stream length was divided by the area of the management unit to calculate a density; this provided a proxy of disturbance intensity, and thus propagule pressure, for each unit. Final estimates of disturbance-based propagule pressure were scaled to range from 0–1 based on the maximum value found for a management unit. Neighborhood and disturbance-based propagule pressure values were averaged to

get an overall score of landscape pressure in each of the units.

Local Factors

The third and final component of the model quantified the suitability of the management unit for reestablishment of *A. altissima* assuming the presence of seeds or rootstock. The heterogeneity of the physical environment is an important control on biological invasion processes (Hastings et al. 2005; Renofalt et al. 2005), and understanding the influence of fine-scale environmental variability on species distributions is necessary to determining where and when to apply invasive species

management actions (Bradley and Mustard 2006). We used a multivariate habitat model to rate the relative quality of every 30-m grid cell on the landscape based on Mahalanobis distance (Mahalanobis 1936) from a set of ‘ideal’ environmental conditions for *A. altissima* derived from observations of known occurrences of the species in the park. Mahalanobis distance (MD_{ij}) is a measure of the environmental dissimilarity of a point i from the mean environmental conditions of the species j , calculated as:

$$MD_{ij} = \sum_{l=1}^p \sum_{k=1}^p w_{kl} (x_{ik} - x_{jk})(x_{il} - x_{jl}) \quad \text{Eq. 3}$$

Table 1. Original weights (i.e., multipliers) used to calculate disturbance-based spread of propagules and two alternative weighting parameter sets.

	Original Weights	Un-weighted	Alternative Weights
Park boundaries	1	1	1
Trails	1	1	1
Unpaved Roads	2	1	5
Paved Roads (not tour or commuter routes)	3	1	10
Tour or Commuter Roads	4	1	20
Streams	2	1	1

where MD_{ij} sums the distances of a point from the ideal state along pairs of environmental axes k and l (e.g., radiation, soil pH) and $l \neq k$. In our *A. altissima* model, p equals the 5 environmental variables described below. These distances are weighted by the inverse of the covariance between the two variables w_{kl} . Thus, Mahalanobis distance corrects for the correlation structure of the environmental matrix x by weighting differences more heavily when variables k and l are uncorrelated (McCune and Grace 2002). This correction is especially useful for habitat modeling as environmental predictor variables are often correlated (Clark et al. 1993). An additional benefit of Mahalanobis distance is that it does not require data on the “absence” of a target species (Tsoar et al. 2007). In the final calculation, the smaller the distance, the more similar the site to the idealized habitat for the species and the higher the reinvasion risk.

We calculated ideal environmental conditions using characteristics from the 47 plant inventory plots in which *A. altissima* was present. For each plot we estimated soil wetness, slope, solar radiation, soil pH, and distance to forest edge. Slope, wetness, and solar radiation were calculated from a 30-m digital elevation model. Wetness was estimated using Beven and Kirkby’s (1979) topographic convergence index. Solar radiation was an estimate of direct and diffuse radiation over the entire year that included topographic shading (Pierce et al. 2005). Soil pH was obtained from digital soil maps of Washington County developed by the National Cooperative Soil Survey. Finally, distance to forest edge was measured from a land cover map created using Ikonos satellite imagery (4-m resolution). Because the input variables differed in terms of their characteristic scales of variability (Figure 5), the habitat models built from them are able to explain and incorporate multiple scales of variation in the targeted species’ distribution.

We then calculated the Mahalanobis distance to the centroid of this multivariate idealized habitat space for each pixel on the landscape, resulting in a raster map of habitat suitability. To exclude areas in the park where *A. altissima* could not

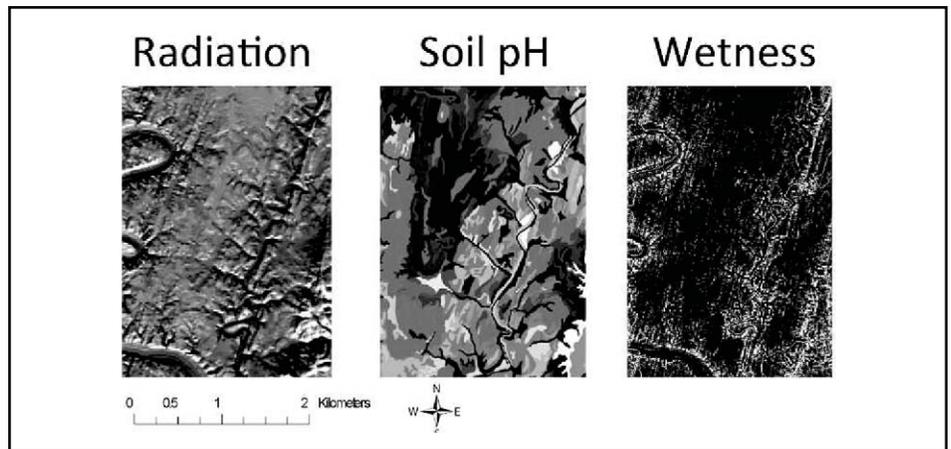


Figure 5. Establishment probability input layers include pH, soil wetness, and radiation which have very different characteristic spatial scales of variability.

become established despite potentially suitable habitat (e.g., open fields that are maintained by mowing), we superimposed an exclusionary land-cover layer over the raster map. The final habitat suitability index (*HSI*) was calculated as:

$$HSI_i = 1 - \frac{MD_{ij}}{MD_{max}} \quad \text{Eq. 4}$$

where j approximates the ideal habitat conditions for the species and MD_{ij} measures how dissimilar a given pixel i is from the reference condition. To rescale the index from 0–1, we divided by the highest dissimilarity value observed in the park, MD_{max} . Subtracting the resulting score from 1 created an index where high suitability (and therefore high reinvasion risk) is represented by a score of 1 and low suitability is represented by a score of 0. The last step was to aggregate the raster data to the relevant scale of decision-making by calculating the average pixel value for each management unit.

Reinvasion Risk

To estimate reinvasion risk, we multiplied the values of global and landscape propagule pressure to provide an estimate of overall propagule pressure (*PP*). Values of *PP* ranged from 0–1, with a value of 0 theoretically possible either when a species was not found in a park (global propagule pressure = 0) or for undisturbed sites not close to any potential seed sources (disturbance-based and neighborhood propagule pressure = 0).

The propagule pressure (*PP*) map represents the risk that seeds (or other propagules) from the invasive species will return to a site following treatment. The habitat (*HSI*) map estimates how well the species would reestablish itself, given the presence of seeds, based on the quality of habitat for the species. The two metrics were multiplied together to provide an estimate of site-specific risk of reinvasion at management unit i :

$$\text{Reinvasion risk}_i = PP_i * HSI_i \quad \text{Eq. 5}$$

$$\text{where } HSI_i = 1 - \frac{\text{Mahalanobis Distance}_i}{\text{Mahalanobis Distance}_{max}}$$

and $PP_i =$

$$\text{Global Propagule Pressure} * \left(\frac{1}{2} \frac{\text{Disturbance-based spread}_i}{\text{Disturbance-based spread}_{max}} + \frac{1}{2} \frac{\text{Neighborhood pressure}_i}{\text{Neighborhood pressure}_{max}} \right)$$

Values close to 0 indicate sites not receiving propagules, not good for establishment, or both. Values of this index close to 1 indicate fertile locations for establishment of the species with a high influx of nearby propagules. The final distribution of management unit risk values was recentered around the mean value of reinvasion risk derived from the EPMT treatment effectiveness database. Thus, the risk values provided on the final map are consistent with the best reestablishment data available for the park and can be used by management to directly quantify the probability of successful treatment for a unit.

RESULTS

We calculated the risk of reinvasion at individual management units as a function of risk factors expressed at three different spatial scales, from global seed rain for the park to local-scale variability in habitat quality.

At the coarsest spatial scale, *Ailanthus altissima* was found in a total of 47 of the 90 vascular plant inventory plots for Antietam NBP (52%). For comparison, 20 of 50 (40%) forest inventory plots in nearby Monocacy National Battlefield contained *A. altissima* (Engelhardt 2005). Other nonnative invasive species in Antietam NBP ranged from less than 5% (e.g., *Ampelopsis brevipedunculata* (Maxim.) Trautv., *Celastrus orbiculatus* Thunb.) to greater than 90% (*Alliaria petiolata* (Bieb.) Cavara & Grande, *Rosa multiflora* ex Murr.) of inventory plots. Multiplying the *A. altissima* inventory value (52%) by the percentage of Antietam NBP in forest cover (24%) provides a global propagule pressure score of 0.13 (Figure 6a).

At the intermediate scale, landscape-level propagule pressure combined spatially explicit data on *A. altissima* habitat and disturbance (Figure 6b). The large tracts of forest edge along Antietam Creek create a high neighborhood propagule pressure within this corridor. Combined with the potential for riverine disturbance and seed transport, this region of the park has the highest risk of *A. altissima* reinvasion based on landscape pressures. The road

network of the park is similarly susceptible to reinvasion. Road and trail density is highest along the spine of the park, which contains a busy commuter road (Maryland Route 65) and infrastructure leading to and away from the Visitor's Center. The influence of adjacent properties is also seen in this metric as neighboring lands to the east and northwest of the park legislative boundary are potentially large sources of propagules.

At the finest spatial scale, the habitat suitability index (HSI) shows high variability within the park, with large areas of suitable habitat along Antietam Creek and in the northwest and southeast portions of the park (Figure 6c).

To calculate reinvasion risk for each of the management units owned by the park, we aggregated all scores to the management units and multiplied together the risks from the three different scales. We then recentered these values around the mean risk value from the EPMT treatment effectiveness data. Mean risk for the 25 units was, therefore, 0.74 (st. dev. = 0.39). Four units had a risk <0.3 and would be top targets for treatment. The 13 units with risk values higher than the mean would be poor choices for treatment if the primary criterion of managers was a low probability of reinvasion (Figure 6d).

DISCUSSION

Nonnative invasive plants are currently one of the largest threats to the natural heritage of the United States National Park Service (Allen et al. 2009). Invasive species are responsible for increased park maintenance costs, present risks to cultural and natural resources, and affect visitor safety. They reduce the ability of parks to meet a variety of management goals by influencing visitors' experiences and local species diversity. The effective control of invasive plants is, therefore, paramount to the NPS mission of preserving natural and cultural resources for current and future generations.

Managers seeking to effectively allocate limited resources to combat invasive

species often must choose how to best target effort and funds. To support the decision-making associated with invasive species management, we have developed an interdisciplinary tool that can be used to design cost-effective treatment programs across multiple invasive species and multiple parks. The tool allows different streams of information about site-specific costs, benefits, and risks to be organized in a systematic framework. It can be implemented relatively inexpensively by drawing from existing data sources (e.g., readily available GIS data and data on infestation densities gathered as part of prior treatment activities) and reasonable default values. Depending on data availability (Table 2) and the need for precision as dictated by the management context, a rough estimate of reinvasion risk could be made using default assumptions in a few days to a few weeks. The primary limitation to model implementation is the availability of a high-quality map of current abundance of the invasive species of concern and data on prior treatment success. As time and resources permit or new data become available, the model could be tailored more precisely to local park conditions and specific management demands. Any upfront investment in model development leading to improved treatment success would be offset by future benefits; the annual flow of ecosystem service benefits per management unit are estimated to be in the \$100,000s for several of the management units in Antietam NBP (Wainger et al. 2012).

Spatially explicit analyses of reinvasion risk inform treatment strategies that would generate the greatest long-term reduction of invasive plant coverage and the largest long-term increase in ecosystem services. The reinvasion model presented in this paper describes how sound ecological principles are applied in the tool to adjust the expected benefits of invasive species management. Information on the spatial distribution of reinvasion risk is used to reduce the benefits at a site in proportion to the site's risk. The higher the risk, the lower the probability of delivering future benefits associated with an uninfested site, resulting in lower risk-adjusted benefits.

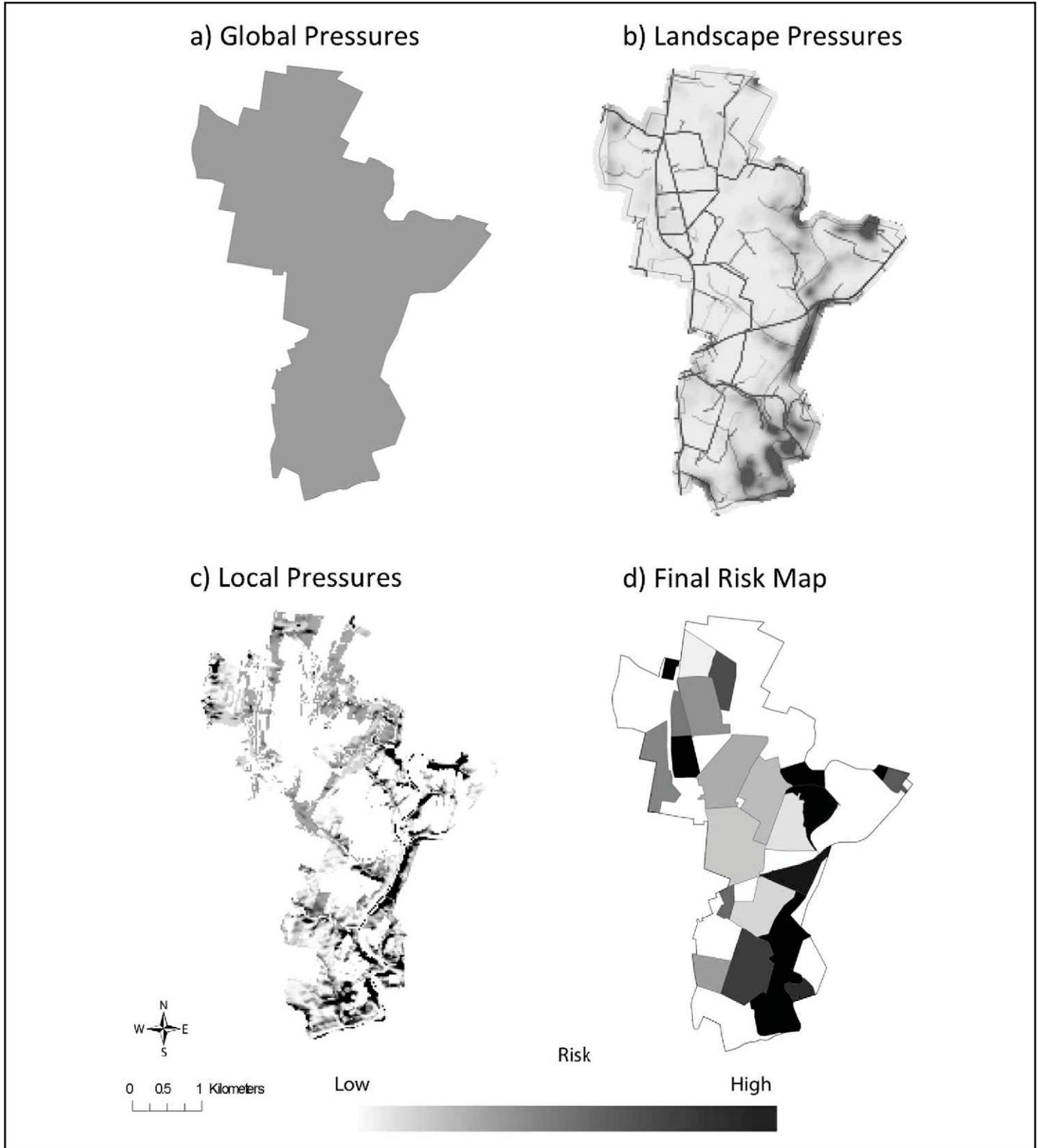


Figure 6. Final *A. altissima* reinvasion risk map for Antietam National Battlefield and associated, derived input layers: (a) global propagule pressures, (b) landscape pressures, and (c) local establishment probability based on habitat suitability.

Table 2. Required input datasets and sources used in analyses.

Variable Description	Source of Data	General Methods
Management Units	Created by NPS staff	boundaries delineate relatively homogeneous units of similar size
Park-level Infestation ¹	Park inventory data	calculated from regional databases or best estimate of the percent of the park infested
Species reproductive strategy	Estimated from literature	see Appendix A for default values
Trails ²	Trails GIS layer provided by park staff	total length of trail per management unit, potentially classified by use
Roads ²	Roads aggregated by intended use	total length of road per management unit, potentially classified by use
Streams ²	United States Geological Survey National Hydrography Dataset	length of streams per management unit
Abundance Map ³	Park staff or modeled distribution	ideally based on empirical data; alternatively, the habitat suitability map could be used in the absence of a detailed map of the species' current distribution
Natural Areas	NPS vegetation maps	only natural (i.e., not heavily managed) areas deemed invadable
Slope, Radiation, Wetness ⁴	United States Geological Survey National Elevation Dataset	topographic variables derived from National Elevation Dataset or other digital elevation model
Edginess ⁴	Ikonos satellite imagery	distance to forest edge calculated using GIS
Soils ⁴	National Cooperative Soil Survey	pH used as an input in habitat model
Treatment Success	Exotic Plant Management Team database	mean treatment success calculated and used as a scalar in the model

¹ used to calculate global propagule pressure
² used to calculate disturbance-mediated spread
³ used to calculate neighborhood propagule pressure
⁴ used to calculate habitat suitability. An alternative approach could be to use an available map of habitat types. For example,

The model facilitates spatially explicit decision-making by focusing on weed populations rather than taking a uniform approach to targeting priority species everywhere (Skurka Darin et al. 2011). The global propagule pressure parameter for *A. altissima* (0.13 for Antietam NBP) is a park-level scalar that adjusts these values based on how common the species is in the park and the reproductive strategies of the species. It assists in prioritizing treatment among multiple parks, which may differ in their global propagule pressure. Landscape-scale propagule pressure (the sum of neighborhood and disturbance-based pressures) is a more local scalar that quantifies the variability of propagule pressure for different management units. It

accounts for the spatial patterning of seed sources and disturbances within a park. The fine-scale influences of environmental heterogeneity are also incorporated in the habitat suitability modeling.

Though this case study is specific to *A. altissima*, it demonstrates the general methods for calculating site restorability based on spatially-explicit species, site, and landscape attributes. Given sufficient input data on regional infestation densities, local propagule pressures, seed dispersal mechanisms, and habitat preferences, the tool could also be applied to almost any species (native or exotic) to predict establishment following a treatment event. This approach could also be adapted for species

that lack the empirical data used to calibrate the model, such as a very rare or newly invasive species. For example, *Humulus japonicus* Siebold & Zucc. (Japanese hops) is just beginning to invade Antietam NBP (observed in only 4% of the vascular plant inventory plots). The species is a shallow-rooted annual vine that is highly invasive along open areas in riparian and floodplain habitat and can greatly restrict tree seedling establishment (Tokuoka et al. 2011). Using this qualitative information on *H. japonicus* habitat preferences, a map of riparian and floodplain habitat types, overlaid with estimates of canopy cover, would provide a reasonable habitat suitability model. It would also be reasonable to inflate the weights for streams in

the disturbance-based spread portion of the model. Isolated observations of this newly invading species would likely be flagged as low risk for reinvasion and a high priority for treatment because both the global and local propagule pressure is near zero.

If information about the invasive species is highly uncertain, an ensemble approach (Jones et al. 2010; Stohlgren et al. 2010) can be readily incorporated into decision support tools through the user interface. For example, if it is unknown whether roads significantly increase seed transport, users may run the model multiple times with differing parameter sets (Table 1) and combine the results to identify management units that are consistently at a high (or low) risk of invasion. In the example provided in Table 1, changing the disturbance weights did not significantly alter the rank ordering of the final risk scores for the management units ($R^2 = 0.87$, $P < 0.01$ between the unweighted and exaggerated weight model

runs). However, notable differences between the disturbance weighting schemes (Figure 7) result in lower risk estimates for the East Woods in the southeast of the park when the importance of roads is elevated for one of the parameter sets in an ensemble run.

Reestablishment of native plant communities following removal of an invasive species is not assured even if the eradication is successfully sustained through time. In many cases, additional restoration measures may be required (Harms and Hiebert 2006). Our model could be used to support restoration efforts by quantifying the “risk of invasion” by desired (e.g., native) species for a treated site (Costa et al. 2012). This might allow users to further prioritize sites for restoration, particularly when budgets are small and native species must be recruited by seed dispersal rather than planted. Our model could also be used to prioritize sites for invasive species monitoring (Lookingbill et al. 2012). Specifically,

sites that are at a high risk of reinvasion are also likely to have a high risk of initial invasion, and managers confronted with limited budgets may wish to focus efforts on monitoring sites that are most likely to be invaded.

Our approach assumes that sites with high reinvasion risk should have low treatment priority. However, other factors might outweigh the risk of reinvasion, such as logistics of treatment options, park priority areas, or preferred benefits. In these cases, this part of the decision support tool could be down-weighted, but rarely will probability of treatment success be a factor that can be completely ignored in prioritizing management action. Our model provides a systematic approach for quantifying reinvasion risk that accounts for information on species biology, habitat suitability, disturbance, and spatially explicit estimates of propagule pressure at multiple scales.

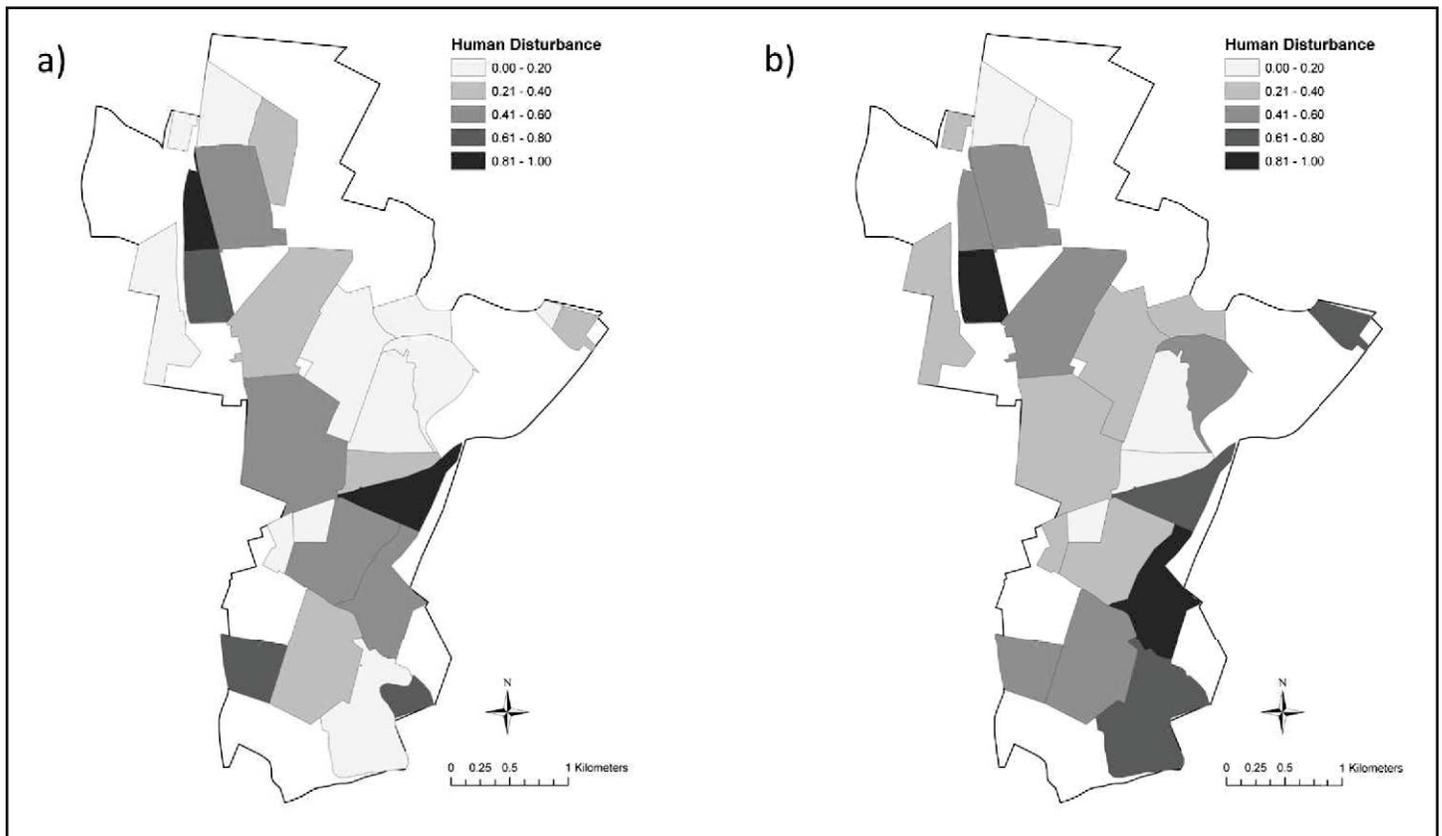


Figure 7. Maps of disturbance-based reinvasion risks given: (a) extreme weighting of roads, and (b) equal weighting of all disturbance factors (see Table 1 for weights).

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APPENDIX

Species-specific reproductive multipliers for *Ailanthus altissima* and other common non-native invasive plants in the National Capital Region network of parks. All species are given a baseline value of 1 based on their seed production (theoretically this could be further adjusted based on species fecundity). This value is increased for species with the ability to reproduce vegetatively from root stock and/or species that are prolific seed bankers as they are especially prone to reappear on treated sites and were given additional weight. This aspatial parameter is used in the calculation of global propagule pressure.

Species	Seed reproduction	Stump sprout	Seed bank	Multiplier scores
<i>Acer platynoides</i>	1	1	0	2
<i>Ailanthus altissima</i>	1	1	0	2
<i>Alliaria petiolata</i>	1	0	1	2
<i>Berberis thunbergii</i>	1	1	0	2
<i>Cirsium arvense</i>	1	1	0	2
<i>Elaeagnus umbellata</i>	1	0	0	1
<i>Euonymus alatus</i>	1	0.5	0	1.5
<i>Euonymus fortunei</i>	1	1	0	2
<i>Humulus japonicus</i>	1	1	0.25	2.25
<i>Ligustrum vulgare</i>	1	1	0	2
<i>Lonicera japonica</i>	1	1	0	2
<i>Lonicera morrowii</i>	1	1	0	2
<i>Microstegium vimineum</i>	1	0	1	2
<i>Polygonum cuspidatum</i>	1	0	0	1
<i>Polygonum perfoliatum</i>	1	0	1	2
<i>Pueraria montana</i>	1	1	0	2
<i>Ranunculus ficaria</i>	1	1	0	2
<i>Rosa multiflora</i>	1	0.5	1	2.5
<i>Rubus phoenocolassius</i>	1	0	0	1
<i>Vinca minor</i>	1	1	0	2