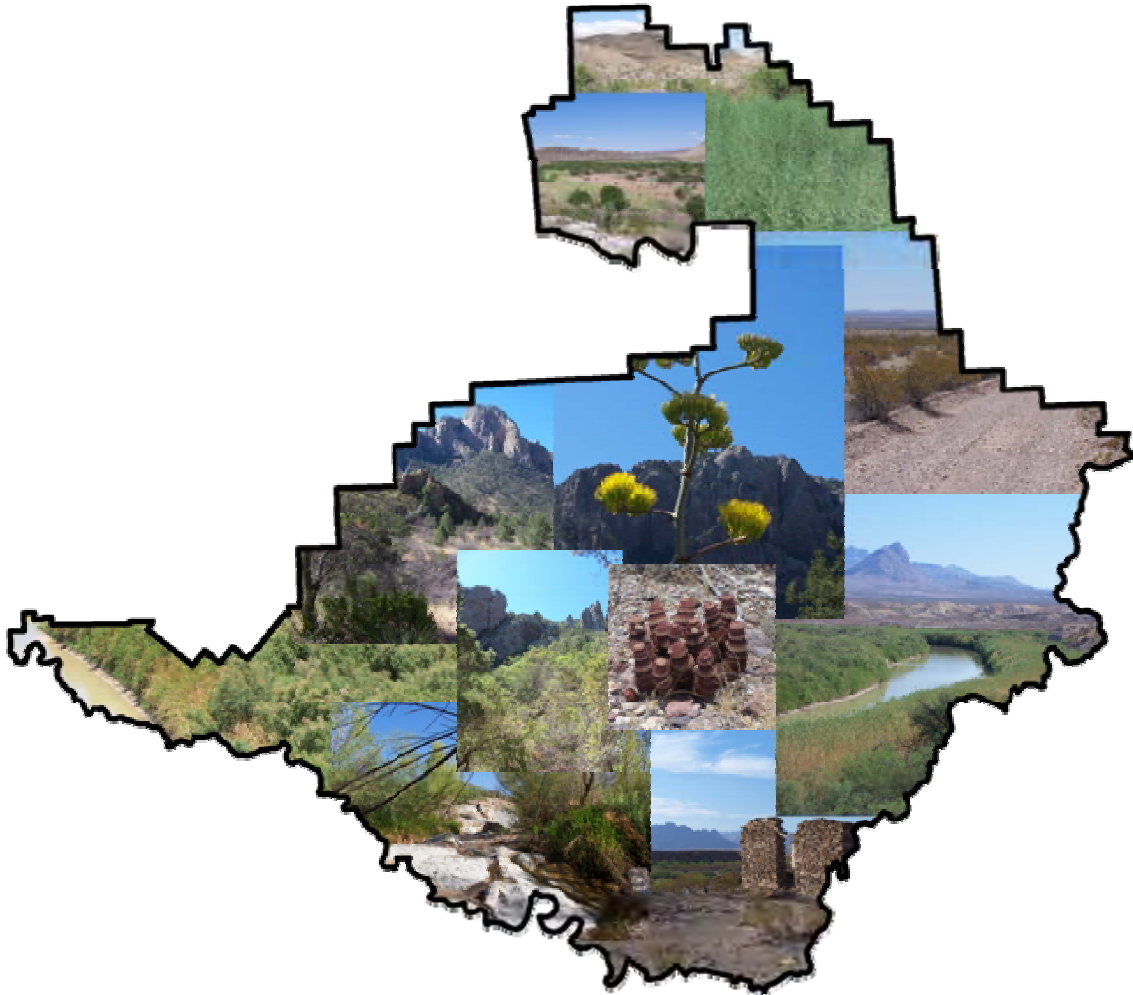


Early Detection of Invasive Species in Big Bend National Park: Remote Sensing and GIS Strategies



Conducted by:

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Executive Summary

Increases in trans-oceanic commerce, terrestrial transportation systems, and varying land use practices have ensured that many nonnative, invasive plant species have either accidentally or deliberately been introduced into places that are well beyond their native range. Once established, invasive species have the ability to displace native plant and animal species (including threatened and endangered species), disrupt nutrient and fire cycles, and alter the character of the community by enhancing additional invasions.

The consequences of nonnative invasions in our National Parks have not gone unnoticed. Approximately 2.6 million acres (1.1 million ha) of national park lands are effected. Big Bend National Park (BIBE) in southwestern Texas is an example. Over 65 exotic plant species were documented in BIBE in 1998. In the region around BIBE, the National Biological Information Infrastructure reports over 200 exotic plants that could potentially influence the Park's environmental quality, biotic health, and biotic integrity.

Developing early detection methods for invasive species could result in substantial economic savings and circumvent negative ecological impacts. Components of early detection protocols include: 1) knowledge on the current presence or absence of a given plant species in or near management units, 2) knowing vectors and pathways of plant dispersal, and 3) understanding the likelihood of the plant establishing or spreading inside management units. This knowledge can then be used to plan a rapid response to remove or control the invasive plant.

Predictive modeling using remotely sensed and Geographic Information System (GIS) data can provide resource managers with a tool for early detection of invasive plants, especially over large landscapes. Once areas of invasive species occurrence are predicted, ground reconnaissance can be more effectively used for verification and control. Over the last decade, there has been a substantial increase in the use of remotely sensed and GIS data to model invasive species distributions or potential habitats. This increase coincides with improved remote sensors, computer technology, and classification techniques.

We evaluated the efficacy of using remotely sensed and GIS data as tools to support early detection of invasive plants in BIBE. To accomplish this evaluation, we identified four objectives for this study: 1) identify a subset of invasive plant species that occur within BIBE that may be useful for landscape scale modeling, 2) create spatially explicit models of predicted distributions of targeted species using remotely sensed and GIS data, 3) conduct a risk assessment of areas for invasive plant incursion, and 4) conduct computer simulations of common field sampling designs to evaluate efficient ground survey methods to detect invasive species in BIBE.

We choose eight invasive plant species for our predictive modeling: giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), Lehmann's lovegrass (*Eragrostis lehmanniana*), horehound (*Marrubium vulgare*), buffelgrass (*Pennisetum ciliare*), Russian thistle (*Salsola kali*), Johnson grass (*Sorghum halepense*), and saltcedar (*Tamarix* sp.). These species are a concern because they rapidly colonize disturbed areas, outcompete native species, and alter natural processes that

further promote invasive species invasions and establishment. We also built a predictive model of all eight species combined to investigate areas where management efforts could impact multiple species.

Spatial locations on existing known occurrences in BIBE were provided by Park staff. This dataset included present and historic plant locations from opportunistic sightings and planned roadside surveys from 2001 to present. We conducted additional ground surveys in June of 2006 throughout BIBE to ascertain additional locations of target species populations. We used a systematic (opportunistic) sampling approach to optimize our chance of detecting additional invasive plant populations. We systematically surveyed roads and trails throughout BIBE. Arroyos, springs, and cross-country surveys were also conducted.

We used Landsat 7 ETM+ data for this study because of the large size (roughly 320,000 ha (790,734 ac) subjected to analyses. We used three seasons of spectral data across five years (Fall 1999, Summer and Fall 2000, Spring and Fall 2001, Spring 2002, and Spring 2003) to account for dynamic habitat conditions that occur in the study area due to discrete rainfall events and to provide a means to capture phenological differences among the targeted species and the surrounding landscape. We used MaxEnt software to generate predictive habitat models for each target species. MaxEnt uses the principle of maximum entropy to estimate the target probability distribution that has the broadest distribution compatible with the information available. Outputs include a probability surface with values from 0-100, tabular and graphical representations of model performance, and model variable contribution. We randomly withheld 10% of the occurrence points for each target species to assess model performance.

Model performance was assessed based on threshold-dependent and threshold-independent metrics. Threshold-dependent metrics require a known threshold to classify a response as presence/absence, or suitable/not suitable. We selected a threshold that provided an equal tradeoff between the test data sensitivity and specificity. We also evaluated model performance using threshold-independent receiver operating characteristic (ROC) curves. The advantage of ROC analysis is that area under the ROC curve (AUC) value provides a single measure of model performance, and can be interpreted as the likelihood that habitat quality in the predictive model is correctly classified at a random positive site and a random negative site.

Our risk analysis incorporated a system of numerical ranks and weighting of six spatial variables: 1) occurrence locations, 2) habitat suitability, 3) disturbance, 4) hydrologic features, 5) wind direction, and 6) fire history. Numeric weights and ranks represented each variable's relative importance in the establishment and spread of our target invasive plant species, and allowed us to combine multiple stressors, habitats, and invasive species occurrences into a spatial model that represents areas most at risk of invasion by nonnative species. The final risk surface was created using an additive Boolean overlay and resulted in classes of low, moderate, and high risk.

We used program SAMPLE to run sampling simulations that identified the most efficient method for sampling each species and evaluated if adaptive sampling techniques would benefit sampling effort. SAMPLE allows for the evaluation of single-stage, two-stage, and stratified sampling designs in the context of adaptive and non-adaptive strategies using simulations generated from GIS datasets.

A total of 1,464 invasive plant locations were documented for the eight species of interest. The number of occurrence points used to generate individual species models ranged from 26-627. Small sample sizes (< 50 occurrences) occurred with giant reed, horehound, and Russian thistle. The influence of small sample sizes on model performance and variable selection is unknown.

All predictive plant models performed better than a random prediction, with the exception of the Russian thistle model. Test AUC values ranged from 0.77 to 0.99. Confidence intervals (95%) built around test AUC values indicate models did not vary across discrimination classes, with the exception of Russian thistle. Horehound and giant reed models yielded high test AUC values, narrow 95% confidence intervals (0.99 ± 0.02), high model gain values (3.4) and low errors of omission ($\approx 0\%$; $P < 0.01$). However, model assessment values for these species may not be robust due to the low number of test samples. Likewise, AUC estimates and 95% confidence intervals for the Johnson grass and saltcedar models yielded high AUC values ($\approx 0.92 \pm 0.02$) with moderate errors of omission ($\approx 15\%$; $P < 0.01$).

Model test AUC values indicated moderate discrimination for buffelgrass (0.88 ± 0.01), Bermuda grass (0.85 ± 0.05), and Lehmann's lovegrass (0.81 ± 0.02). Errors of omission were high for Bermuda grass (27%) and Lehmann's lovegrass (28%), but were moderate for buffelgrass (19%). We observed similar levels of discrimination when combining all plant locations into one predictive model. The combined target plant predictive model resulted in a test AUC value of 0.82 ± 0.004 , and a 27% error of omission. Combining data from all target plants likely created a habitat generalist dataset which may have reduced our ability to differentiate suitable habitats (resulting in a moderate AUC value) or detect vegetation phenology differences using the maximum entropy approach. However, the larger sample size resulted in greater model precision (low AUC standard deviation). Our Russian thistle model yielded a moderate AUC value (0.77) with 95% confidence intervals ranging from 0.58 to 0.96 and a high error of omission (33%), effectively reducing the reliability of the model.

We used analyses provided in MaxEnt to guide us toward parsimonious models. However, even with these procedures, the number of variables retained in each model was relatively large and varied between 15 and 35 spectral classes. Our results indicated that models with larger sample sizes (number of plant occurrences) yielded models with greater number of variables. Further, the greater number of variables in the models reduced the likelihood that the model could differentiate plant occurrence locations from random background pixels. These results likely reflect MaxEnt's attempt to estimate the plants broadest probability distribution given an increase in variability associated with greater numbers of sample locations and the increased variability associated with greater number of model variables. Our giant reed, horehound, and Russian thistle models may be overfit due to their low number of occurrence locations. We recommend additional model selection processes including correlation analyses, or evaluating multiple models using Akaike's Information Criterion (AIC).

Giant reed and horehound had the most limited predicted distribution in BIBE, with approximately 2% of the park surface modeled as potential habitat. Giant reed predicted habitat was fairly restricted to the Rio Grande corridor while horehound predicted habitat was primarily around developed areas near the Chisos Mountains and small patches near Rosillos Mountains.

We estimated approximately 14% of BIBE was suitable for saltcedar. Saltcedar occurs in large stands along the Rio Grande with smaller stands fairly common in arroyos and springs that have adequate seasonal water for establishment. Buffelgrass is a potentially increasing problem for BIBE. Approximately 20% of BIBE was predicted to have suitable conditions for this plant. Large stands of buffelgrass already occur in the southern part of the Park, but this grass can be found throughout BIBE. Bermuda grass is another large problem in BIBE. We estimated approximately 27% of BIBE to be suitable for Bermuda grass. This grass is already well established in developed areas, campgrounds, and along the Rio Grande where it has displaced native vegetation. Johnson grass had the largest predicted amount of potential suitable habitat, approximately 40% of BIBE. Johnson grass is well established along the North Rosillos Road, but potential habitats occur throughout the Park.

Based on model performance, our proof-of-concept approach worked well for giant reed, horehound, Johnson grass, and saltcedar, and moderately well for buffelgrass, Bermuda grass, and Lehmann's lovegrass. The efficacy of using remotely sensed data to model the potential distribution likely was influenced by invasive patch size, habitat generality, and similar environmental conditions surrounding invasive plant populations. Modeling approaches that employ presence/absence analytical approaches may help to differentiate these communities. Further, integrating other spatial information (e.g., disturbances, and soil moisture regimes), high spatial resolution data, or larger sample sizes may improve model performance.

Our risk assessments were designed to evaluate the risk of landscape parcels being further invaded or established by our target invasive plants. Areas at risk included Panther Junction west to Croton Springs, Chisos Basin, Tornillo Flat near U.S. 385, Tornillo Creek near State Highway 118, Rio Grande Village, Boquillas Canyon, drainage between Dominguez Spring and the Rio Grande, riparian habitats from Cottonwood Campground to Santa Elena Canyon, East Corazones Draw, and paved roadways, especially Chisos Basin Road, Rio Grand Village Road, Boquillas Canyon Road, State Highway 118, Ross Maxwell Scenic Drive, and Persimmon Gap Road (U.S. 385).

The amount of BIBE at high risk to invasion varied from approximately 188 ha (465 ac) to 1,957 ha (4,836 ac). Assessments using all eight target species indicated approximately 3,976 ha (9,825 ac) are at high risk of invasion and 25,027 ha (61,843 ac) are at moderate risk in BIBE. Moderate risk landscapes may develop into high risk landscapes if invasive plant species are found in close proximity, disturbance regimes enhance habitat suitability, or suitable vectors and pathways are developed linking landscapes. Although the acreage at risk represents < 10% of the entire Park, habitats are not equally at risk. For example, habitats that host greater human infrastructures (roads, trails, campgrounds, and developed areas), are at greater risk to incursion by invasive plants. Likewise, our analyses indicated that riparian habitats may be 3-times more likely to become invaded than other general habitat types.

We evaluated six sampling designs using simulations derived from known plant occurrences and potential distributions in BIBE. Our results indicate that for single-stage designs, an adaptive simple random sampling procedure could provide reliable estimates for six (75%) of the species evaluated. We were unable to evaluate complex sampling designs with our datasets. However, sampling procedures should consider areas at risk of invasive plant incursion. Including areas at

risk as a stratum or using the risk assessment to alter plant probabilities may increase the likelihood of detecting rare species and increase the utility of field sampling procedures for early detection protocols.

Managing plant invasions is challenged by discrepancies in timing when invasions occur and when invasions are observed or recorded. Understanding the likelihood of an invasive plant occurrence across large landscapes helps in establishing early detection protocols which may promote the implementation of proactive management practices to decrease the prevalence of nonnative species through eradication or control efforts, and maintain ecosystem integrity. Landscape scale habitat modeling and risk assessments provide powerful tools to help in this endeavor. Our habitat models and associated risk analysis was an effective approach to assess landscapes for risk to incursion of invasive plants. These models can inform park managers regarding the potential introduction of invasive species, their vectors and pathways, and the allocation of resources for the control of invasive species. In addition to prioritizing areas to monitor or control invasive plants, spatial models can be used to evaluate the effects of invasive species occurrences on Threatened and Endangered species, species of concern, recreational activities, and planned management actions.



Table of Contents

Executive Summary	i
Table of Contents	vi
List of Tables	ix
List of Figures	x
List of Appendices	xiv
Preface.....	xv
Acknowledgments.....	xvi
Chapter 1	1
Invasive Species Issues.....	1
Invasive Species on National Park Lands	1
Early Detection Needs	2
Remotely Sensed and GIS Data for Invasive Plant Early Detection	3
Research Objectives.....	5
Study Area Description.....	6
Chapter 2	8
Invasive Species in Big Bend National Park.....	8
Prioritizing Invasive Species for Modeling Efforts	10
Species Characteristics.....	12
Giant Reed	12
General Description	12
Vectors and Pathways.....	12
Habitat Associations	13
Bermuda Grass.....	13
General Description	13
Vectors and Pathways.....	14
Habitat Associations	14
Lehmann’s Lovegrass	14
General Description	14
Vectors and Pathways.....	15
Habitat Associations	15
Horehound.....	15
General Description	15
Vectors and Pathways.....	16
Habitat Associations	16
Buffelgrass	16
General Description	16
Vectors and Pathways.....	17
Habitat Associations	17
Russian Thistle.....	17
General Description	17
Vectors and Pathways.....	18
Habitat Associations	18
Johnson Grass	18

General Description	18
Vectors and Pathways	19
Habitat Associations	19
Saltcedar.....	19
General Description	19
Vectors and Pathways	20
Habitat Associations	20
Chapter 3	21
Invasive Species Predictive Habitat Models	21
Introduction.....	21
Remote Sensing Considerations	21
Types of Remotely Sensed Data	23
Low Spatial Resolution Sensors	23
Medium Spatial Resolution Sensors	23
High Spatial Resolution Sensors.....	23
Hyperspectral Sensors.....	24
Analytical Approaches to Data Modeling.....	25
Methods.....	26
Target Species Occurrences.....	26
Spatial Datasets.....	26
Predictive Habitat Models.....	27
Variable Selection.....	27
Model Performance.....	28
Results and Discussion	29
Sample Locations.....	29
Performance of Predictive Models.....	30
High Discriminate Models.....	40
Moderate Discriminate Models.....	40
Low Discriminate Model	41
Model Variable Selection	42
Amount of Predicted Suitable Habitats.....	43
Predictive Model Considerations.....	44
Conservation Implications	45
Chapter 4	47
Invasive Species Risk Assessment	47
Introduction.....	47
Methods.....	48
Numeric Ranks of Risk Variables.....	50
High Ranked Variables	50
Moderate Ranked Variables.....	50
Low Ranked Variables.....	50
Numeric Weights Within Risk Variables	51
Habitat Suitability	53
Disturbance	53
Hydrologic Features.....	53
Wind Direction.....	54

Fire History	55
Building Risk Surfaces	55
Results and Discussion	56
Categories of Risk.....	56
Areas at Risk of Invasion.....	57
Conservation Implications	67
Chapter 5	70
Sampling Design Simulation	70
Introduction.....	70
Methods.....	72
Datasets Used for Simulations.....	72
Sampling Designs Evaluated	73
Single-Stage Designs	73
Two-Stage Designs	74
Simulation Settings	74
Evaluation Metrics	74
Results and Discussion	75
Giant Reed	75
Bermuda Grass.....	76
Lehmann's Lovegrass	77
Horehound.....	78
Buffelgrass	78
Russian Thistle.....	79
Johnson Grass	80
Saltcedar.....	81
All Eight Species.....	81
Summary Results and Discussion.....	81
Limiting Factors.....	83
Conclusions and Implications	83
Literature Cited	86
Appendices.....	104

List of Tables

Table 1-1.	Invasive plants where remote sensing and GIS data aided in the detection and modeling of the plants distribution. This list is not inclusive of all plants modeled with remote sensing and GIS data	4
Table 2-1.	Exotic plant species documented in Big Bend National Park in 1998.....	9
Table 2-2.	Invasive species selected for remote sensing and GIS spatial analyses in Big Bend National Park	11
Table 3-1.	Spatial and spectral resolution of remote sensors that may be useful for detecting potential invasive plants or modeling their habitats. This list is not inclusive of all possible remote sensors available.	22
Table 3-2.	Dates and seasons of imagery used to model potential invasive plant habitat in Big Bend National Park.	26
Table 3-3.	Confusion matrix to derive measurements of model performance	29
Table 3-4.	Number of georeferenced target invasive plant occurrences (individual plants or populations) used to construct predictive habitat models in Big Bend National Park.	30
Table 3-5.	Performance measurements for invasive species predicted habitat models in Big Bend National Park.	40
Table 3-6.	Amount of predicted suitable habitat in BIBE for each target invasive plant species estimated using the 'Equal Test Sensitivity and Specificity' threshold from MaxEnt.	44
Table 4-1.	Risk analysis datasets used to estimate vectors and pathways for target invasive plants in Big Bend National Park	49
Table 4-2.	Numeric ranks given to risk assessment variables to represent each variable's importance in the establishment and spread of our target invasive plant species in Big Bend National Park	50
Table 4-3.	Numeric weights given to variable characteristics used to estimate areas at risk of invasion in Big Bend National Park	52
Table 4-4.	Amount of area at risk to invasion by target invasive plants in Big Bend National Park.....	57
Table 5-1.	Matrix of sampling designs evaluated using SAMPLE software (Smith and Nichols 2006) for each invasive plant species in Big Bend National Park.....	73
Table 5-2.	SAMPLE output metrics used to evaluate sampling designs for each of the target invasive plant species in BIBE (Smith and Nichols 2006)	74
Table 5-3.	Summary of sampling designs that could potentially yield robust estimators for invasive plant populations in Big Bend National Park	82

List of Figures

Figure 1-1.	Spectral reflectance curves derived from Landsat 7 ETM+ for vegetation types in Big Bend National Park.....	3
Figure 1-2.	Big Bend National Park in southwestern Texas was used to test the efficacy of using remotely sensed data for the early detection of invasive plants in the Chihuahuan Desert.	7
Figure 3-1.	Two Landsat 7 ETM+ scenes (3040 and 3041) were conjoined to provide complete coverage of Big Bend National Park.....	27
Figure 3-2.	MaxEnt program settings used to estimate potential invasive plant habitats in Big Bend National Park	28
Figure 3-3.	General area surveyed (red) for target invasive plants in Big Bend National Park in June 2006.....	30
Figure 3-4.	Potential distribution of giant reed in BIBE modeled using maximum entropy of presence locations.....	31
Figure 3-5.	Potential distribution of Bermuda grass in BIBE modeled using maximum entropy of presence locations.....	32
Figure 3-6.	Potential distribution of Lehmann's lovegrass in BIBE modeled using maximum entropy of presence locations.....	33
Figure 3-7.	Potential distribution of horehound in BIBE modeled using maximum entropy of presence locations.....	34
Figure 3-8.	Potential distribution of buffelgrass in BIBE modeled using maximum entropy of presence locations.....	35
Figure 3-9.	Potential distribution of Russian thistle in BIBE modeled using maximum entropy of presence locations.....	36
Figure 3-10.	Potential distribution of Johnson grass in BIBE modeled using maximum entropy of presence locations.....	37
Figure 3-11.	Potential distribution of saltcedar in BIBE modeled using maximum entropy of presence locations.....	38
Figure 3-12.	Potential distribution of invasive habitats modeled from all eight target species in BIBE using maximum entropy of presence locations.	39
Figure 3-13.	Invasive plant model discrimination classes based on test AUC and 95% confidence intervals.	41
Figure 3-14.	Relationship between sample size, gain derived from training samples and the number of model variables retained in the eight invasive plant predictive habitat models.	42

- Figure 4-1. Diagram of how wind direction polygons were created: A) occurrence points were shifted 250 m downwind, B) points were buffered 250 m, C) overlap area of polygons were considered an area where invasive species establishment is enhanced by short distance wind movement of propagules. 55
- Figure 4-2. Areas at risk of giant reed incursion or spread in Big Bend National Park. The likelihood of giant reed incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian and other habitats may be threatened by giant reed as demonstrated by the moderate or high risk areas (orange to red) in the figure. 58
- Figure 4-3. Areas at risk of Bermuda grass incursion or spread in Big Bend National Park. The likelihood of Bermuda grass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of rangelands, fields, riparian areas, and understories of open woodlands may be threatened by Bermuda grass as demonstrated by the moderate or high risk areas (orange to red) in the figure. 59
- Figure 4-4. Areas at risk of Lehmann's lovegrass incursion or spread in Big Bend National Park. The likelihood of Lehmann's lovegrass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function desert shrubland and semi-desert grassland communities may be threatened by Lehmann's lovegrass as demonstrated by the moderate or high risk areas (orange to red) in the figure. 60
- Figure 4-5. Areas at risk of horehound incursion or spread in Big Bend National Park. The likelihood of horehound incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function desert shrubland and semi-desert grassland communities may be threatened by horehound as demonstrated by the moderate or high risk areas (orange to red) in the figure. 61
- Figure 4-6. Areas at risk of buffelgrass incursion or spread in Big Bend National Park. The likelihood of buffelgrass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of semi-desert grassland communities may be threatened by buffelgrass as demonstrated by the moderate or high risk areas (orange to red) in the figure. 62

Figure 4-7.	Areas at risk of Russian thistle incursion or spread in Big Bend National Park. The likelihood of Russian thistle incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of semi-desert grassland communities may be threatened by Russian thistle as demonstrated by the moderate or high risk areas (orange to red) in the figure.	63
Figure 4-8.	Areas at risk of Johnson grass incursion or spread in Big Bend National Park. The likelihood of Johnson grass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian habitats and springs may be threatened by Johnson grass as demonstrated by the moderate or high risk areas (orange to red) in the figure.	64
Figure 4-9.	Areas at risk of saltcedar incursion or spread in Big Bend National Park. The likelihood of saltcedar incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian habitats and springs may be threatened by saltcedar as demonstrated by the moderate or high risk areas (orange to red) in the figure.	65
Figure 4-10.	Areas predicted to be at risk by at least one of the eight target invasive plant species in Big Bend National Park. The likelihood of invasive plant incursion was estimated from spatial analyses of current target invasive plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Primary areas at moderate or high risk (orange to red areas) which may suffer modified ecological stability and function include areas immediately proximate to riparian and spring habitats, roads and human activity centers.	66
Figure 4-11.	Proportion of general habitat types modeled by Plumb (1991) that are in moderate and high risk of invasive plant incursion in Big Bend National Park. Riparian habitat types may have 3-times greater risk than other habitat types.	67
Figure 4-12.	Amount of area (hectares) predicted to be suitable habitat (Chapter 3) and predicted to be at moderate or high risk of invasion by our target invasive plants in Big Bend National Park.	68
Figure 5-1.	Efficiency and the hits to occupancy ratio for giant reed field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006)	75

Figure 5-2.	Efficiency and the hits to occupancy ratio for Bermuda grass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset	76
Figure 5-3.	Efficiency and the hits to occupancy ratio for Lehmann's lovegrass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset	77
Figure 5-4.	Efficiency and the hits to occupancy ratio for horehound field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006)	78
Figure 5-5.	Efficiency and the hits to occupancy ratio for buffelgrass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset	79
Figure 5-6.	Efficiency and the hits to occupancy ratio for Russian thistle field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset	79
Figure 5-7.	Efficiency and the hits to occupancy ratio for Johnson grass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset	80
Figure 5-8.	Efficiency and the hits to occupancy ratio for saltcedar field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006)	81
Figure 5-9.	Efficiency and the hits to occupancy ratio for all eight target invasive plants combined field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage GRID-NA and SRS-GIS-A design could not be estimated for this dataset....	82

List of Appendices

Appendix A.	Invasive plants that have been recognized by the National Biological Information Infrastructure as invading or having the potential to invade the Central Southwest and Gulf Coast Region (http://cswgcin.nbii.gov/). Big Bend National Park occurs in this region	103
Appendix B.	General soil units in Big Bend National Park (Cochran and Rives 1985).....	108
Appendix C.	Coordinates (UTM, NAD 83, Zone 13) of invasive plant occurrences in Big Bend National Park used to model their potential distribution within the Park..	109
Appendix D.	Jackknife results of training gain for each model variable and invasive plant. Higher gain numbers indicate variable that when used in isolation appear to have the most useful information by itself. Blank spaces indicate the variable was not retained in the model.	119
Appendix E.	Soil attributes of Big Bend National Park. Highlighted soil units were considered as having sufficient soil moisture properties to enhance establishment of invasive plants.	120
Appendix F.	Possible combinations of model input variables that may constitute high risk areas. Only combinations with model values $\geq$ to 450 are provided...	122
Appendix G.	Possible combinations of model input variables that may constitute moderate risk areas. Only combinations with model values from 270-499 are provided	143
Appendix H.	Efficiency, occupancy, and hits metric values derived from SAMPLE (Smith and Nichols 2006) used to evaluate each sampling design for the target invasive plant species in Big Bend National Park. Definitions of each metric are provided in Table 5-2. Sample size (n) represents the number of locations used in the simulations	152

Preface

We started this research with a hypothesis that we could predict potential habitat for invasive plant species in the Chihuahuan Desert using remotely sensed data. There has been considerable work on modeling invasive plant populations using GIS and remotely sensed data in mesic environments. However, arid environments host significant challenges in discriminating landscape features, especially when target populations occur in sparse populations. As such, there has been little research in modeling potential distributions of invasive plants in the Chihuahuan Desert. In this report, we present a “proof-of-concept” approach that we believe is a reasonable and successful approach to model potential invasive species habitat. We believe this approach could aid resource managers in developing early detection protocols and prioritizing survey, monitoring, and control efforts for invasive plants in Big Bend National Park.

The contents of this report are presented in chapters. Scientific names of animal or plants used in each chapter are presented the first time they are used in each chapter. Each chapter builds on information from previous chapters. For example, in Chapter 1 we provide a summary of the effects of invasive species in natural areas and describe how GIS and remotely sensed data can provide valuable information for early detection protocols. In Chapter 2, we describe our target invasive plants used in predictive models and risk assessments. The predictive habitat models described in Chapter 3 relies on information provided in Chapters 1 and 2. Likewise, our risk assessments presented in Chapter 4 relies on information from Chapter 1, 2 and 3. Chapter 5 provides some guidance for selecting field sampling designs for our target species. We relied on our predictive models from Chapter 3 and risk models from Chapter 4 for computer simulation. As such, we encourage readers to become familiar with the entire report. Some of the information in this report is unavoidably technical given the extensive reliance on remote sensing analyses and technology.

Acknowledgments

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Chapter 1

Invasive Species Issues

The introduction, establishment, and spread of nonnative, invasive species has received heightened attention because of their associated negative effects on global biodiversity (Stohlgren et al. 1999). The term “invasive species” defined by Executive Order 13112, refers to an “alien species whose introduction does or is likely to cause economic or environmental harm or harm to human health” (U.S. Department of State 1999). Although only about 1 out of 1000 introduced species become a pest (Williamson and Fitter 1996), those that do can have significant ecological and economic impacts. Once established, invasive species have the ability to displace native plant and animal species (including threatened and endangered species), disrupt nutrient and fire cycles, and alter the character of the community by enhancing additional invasions (Cox 1999, Olson 1999, DeLoach et al. 2000, Zavaleta et al. 2001, Osborn et al. 2002). Few communities are impenetrable to invasion by nonnative species, and communities differ in their susceptibility to invasion (Sakai et al. 2001). Environmental disturbance can promote the establishment of nonnative plant species by temporarily eliminating native competitors, and/or increasing resource availability (Stohlgren et al. 1998).

The economic impact caused by invasive species is difficult to estimate because their cumulative impacts are poorly understood. Pimentel et al. (2000) estimates that approximately 50,000 species have been introduced to the United States (U.S.), and have caused losses of approximately \$137 billion per year. Invasive species in the U.S. are thought to be spreading at about 1,862 ha (4,600 acres) per day or approximately 700,000 ha (1.7 million acres) each year

(Asher and Harmon 1995, Pimentel et al. 2000). The Southwest Strategy (a natural resource conservation and management effort by federal, state, tribal and local governments) reported spotted knapweed (*Centaurea maculosa*) infestations reduced winter elk forage by 50-90% and cost an estimated \$42 million annually in direct and secondary economic impacts in Montana (<http://www.swstrategy.org>). Leafy spurge (*Euphorbia esula*) has had a similar economic impact in Montana, Wyoming, North Dakota, and South Dakota.

Invasive Species on National Park Lands

The National Park Service (NPS) has recognized the negative effects of invasive species on Park lands. Exotic plants infest approximately 1.1 million ha (2.6 million acres) in national parks, reducing the natural diversity (National Park Service 2005). The National Park Service established Exotic Plant Management Teams (EPMTs) to combat and control exotic plants (Dress and Johnston 2001). Each EPMT focuses on multiple parks within a broad geographic area.

Big Bend National Park (BIBE) is in the Chihuahuan Desert/Southern Shortgrass Prairie EPMT region. BIBE was established in June 1944 to preserve a portion of the Chihuahuan Desert ecosystem (NPCA 2003). The park hosts an array of Chihuahuan Desert shrubland, grassland, montane woodland, riparian, and wetland areas. BIBE has over 300 perennial springs which are important to plant and wildlife communities. Although the exact number of invasive plant species on BIBE is unknown, park resources are potentially threatened by over 200 invasive plant species that have been identified in the

region of BIBE (<http://cswgcin.nbii.gov/>; Appendix A).

The community dynamics in BIBE demonstrate how invasive species have reduced the biotic integrity the Chihuahuan Desert. The National Parks Conservation Association (NPCA 2003) assessed BIBE in terms of natural resources, cultural resources, and stewardship capacity. The assessment rated the overall condition of natural resources at BIBE as 62 out of 100. This rating was based on evaluation of more than 120 elements describing environmental quality, biotic health, and ecosystem integrity. The NPCA concluded that the low score was a reflection of historic land-use practices, pollution, and threats to biodiversity from invasive species. The introduction of nonnative animals, such as feral hogs (*Sus scrofa*), and nonnative plants, such as tamarisk (*Tamarix* spp.), Johnson grass (*Sorghum halepense*), Russian thistle (*Salsola kali*), and buffelgrass (*Cenchrus ciliaris*) have degraded natural habitats and reduced biodiversity (NPCA 2003). Further, throughout the western U.S., plants such as cheatgrass (*Bromus tectorum*), Lehmann's lovegrass (*Eragrostis lehmanniana*), and tamarisk have increased the frequency of high intensity fire, reinforcing invasive species competitive advantage over native plants and reducing habitat for native wildlife (Cox 1999, Pimentel et al. 2000). Developing methods to detect invasive species soon after introduction could result in substantial economic savings and circumvent negative ecological impacts.

Early Detection Needs

Eradication of large (> 100 ha) invasive plant patches is typically costly and unsuccessful. Eradication efforts are most successful for infestations < 1 ha (Rejmanek and Pitcairn 2002). Successful eradication or control efforts are dependent on detection of new

populations, or in terms of control, when an existing population begins to spread. Early detection methods allow for proactive and adaptive management and ultimately positive invasive species management investments (Rejmanek 2000, Smith et al. 1999).

The key to early detection and eradication is obtaining knowledge about 1) the current presence or absence of a given plant species in or near an area of interest (e.g., management area, Park, forest, county, or state), 2) vectors and pathways of dispersal, and 3) the likelihood of the plant establishing or spreading inside of the area of interest. This knowledge can then be used to plan a rapid response to remove or control the plant.

Gaining this knowledge for large areas can be complicated. Jurisdiction boundaries often limit search areas and increase the potential to miss existing infestations near boundaries. High costs associated with ground reconnaissance may preclude resource managers from surveying all their managed lands to determine invasive species presence. Invasive plant populations often exhibit a lag period between the introduction and subsequent colonization of new areas (Hobbs and Humphries 1995, Binggeli 2001). This lag time may require frequent visits to an area to detect infestations, yet frequent visits over large areas are often not fiscally possible. These factors lead to undetected plants that may rapidly spread once environmental conditions become favorable.

Predictive modeling can provide resource managers with a tool for early detection of invasive plants that allows for large area evaluations and incorporates potential time lags. A cost-effective predictive modeling approach to identify the occurrence of invasive species is to combine the use of remotely sensed and Geographic Information System (GIS) data with knowledge of species

ecology and environmental tolerances (Everitt et al. 1996, Osborn et al. 2002, Goslee et al. 2003, Parker-Williams and Hunt 2004). Once potential areas of invasive species occurrence are predicted, ground reconnaissance can be more effectively used for verification and control. Areas predicted as potential habitat that are not occupied, may be areas for frequent monitoring as undetected plant propagules may exist in the area.

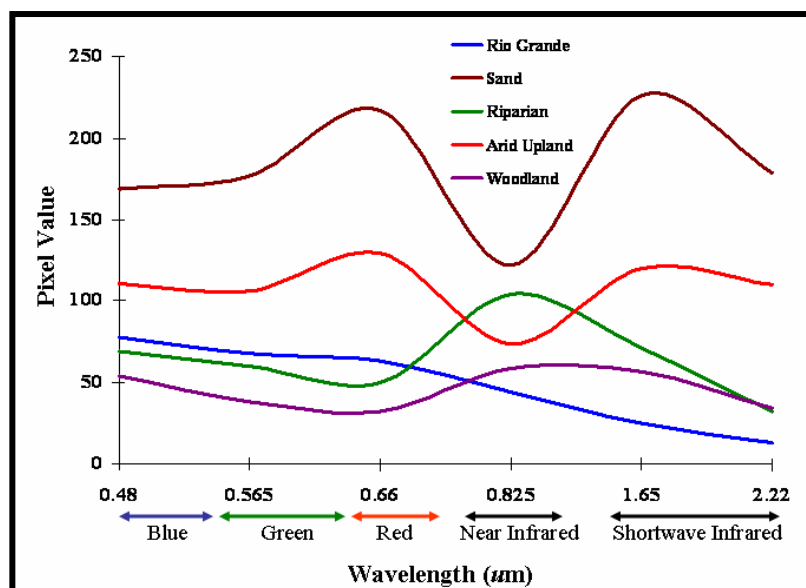
Remotely Sensed and GIS Data for Invasive Plant Early Detection

Remote sensing is a means to describe characteristics of an area without physically sampling the area. Remote sensors can be mounted on a satellite, plane, or other airborne structure. Sensors measure the electromagnetic energy reflected from an object or area on the earth's surface. The guiding principle is that different objects (e.g., soils, plants, buildings, water) reflect and absorb light differently at varying wavelengths. Graphically plotting the amount of radiation reflected at a given wavelength provides a unique signature for an object, especially if there is sufficient spectral resolution to distinguish its spectrum from

those of other objects (Fig. 1-1). Reflectance of clear water is typically low, with higher reflectance values in the blue end of the spectrum which decreases as the wavelength increases. Vegetation reflectance is typically low in both the blue and red regions of the spectrum, due to absorption by chlorophyll (Jensen 2000). Reflectance values peak at the green region of the electromagnetic spectrum. In the near infrared (NIR) region, reflectance is much higher than that in the visible bands due to leaf cellular structure. Therefore, vegetation can be identified by the high NIR but generally low visible reflectance (Jensen 2000). Spectral reflectance curves can be used to discriminate between vegetation types or plant species.

Many GIS datasets are created from remotely sensed data. For example, digital elevation models (DEMs) are derived from space or aircraft borne sensors. Most GIS software packages can calculate slope and aspect from a DEM. Remotely sensed data can also be used to augment GIS datasets by allowing visual interpretation of images for roads, waterways, fence lines, buildings, and other features. These features can be readily digitized and placed into a GIS environment.

Figure 1-1 (Right). Spectral reflectance curves derived from Landsat 7 ETM+ for vegetation types in Big Bend National Park.



Remotely sensed and GIS data have been widely used to detect and model invasive plant distributions (Table 1-1). For example, Advanced Very High Resolution Radiometer (AVHRR), Landsat Thematic Mapper (TM), Landsat Enhanced Thematic Mapper Plus (ETM+), IKONOS, QuickBird, air photos, and hyperspectral sensors have all been used to detect and model invasive plants or their habitats. The efficacy of remote sensing data for detecting invasive plants or associated habitat is a function of the sensors' spatial and

spectral (bandwidth) resolution, the sensors' repeat cycle, and plant phenological characteristics. Hyperspectral sensors are perhaps the most helpful group of remote sensors for detecting small populations of invasive plants. These sensors sample the electromagnetic spectrum in narrow continuous increments which allows for improved identification of species. However, analyses using hyperspectral data are encumbered by large data volumes and computing limitations.

Table 1-1. Invasive plants where remote sensing and GIS data aided in the detection and modeling of the plants distribution. This list is not inclusive of all plants modeled with remote sensing and GIS data.

Plant	Authors
Babysbreath (<i>Gypsophila paniculata</i>)	Lass et al. (2005)
Broom snakeweed (<i>Gutierrezia sarothrae</i>)	Peters et al. (1992)
Cheatgrass (<i>Bromus tectorum</i>)	Bradley and Mustard (2005)
Dyers woad (<i>Isatis tinctoria</i>)	Dewey et al. (1991)
Giant hogweed (<i>Heracleum mantegazzianum</i>)	Mullerova et al. (2005)
Giant reed (<i>Arundo donax</i>)	DiPietro (2002)
Haleakala silversword (<i>Argyroxiphium sandwicense</i>)	Landenberger et al. (2003)
Horse tamarind (<i>Leucaena leucocephala</i>)	Tsai et al. (2005)
Leafy spurge (<i>Euphorbia esula</i>)	Everitt et al. (1995) Anderson et al. (1996) Lawrence et al. (2006) Parker-Williams and Hunt (2002) Parker-Williams and Hunt (2004) Glen et al. (2005)
Mimosa (<i>Mimosa pigra</i>)	Lonsdale (1993)
Saltcedar (<i>Tamarix ramosissima</i>)	Everitt et al. (1992) Everitt et al. (1996)
Scotch thistle (<i>Onopordum</i> spp.)	McGowen et al. (2001)
Serrated tussock (<i>Nassella trichotoma</i>)	McGowen et al. (2001)
Spotted knapweed (<i>Centaurea maculosa</i>)	Lawrence et al. (2006) Lass et al. (2002) Goslee et al. (2003) Lass et al. (2005)
Yellow starthistle (<i>Centaurea solstitialis</i>)	Lass et al. (1996) Lass and Thill (2000) Shafii et al. (2003)

Our ability to differentiate invasive species via remotely sensed data is increased when plants have unique phenological stages relative to the surrounding landscape (Hobbs 1990, McGowen et al. 2001, Chudamani et al. 2004). Some invasive plants flower or green-up at a different time than the surrounding vegetation. Multiple image dates may allow for detecting these phenological differences. For example, remote sensing data that coincided specifically to flowering events aided in the detection of yellow starthistle (*Centaurea solstitialis*) (Lass et al. 1996), and leafy spurge (Anderson et al. 1996, Parker-Williams and Hunt 2002). Likewise, Peters et al. (1992) found that broom snakeweed (*Gutierrezia sarothrae*) could be differentiated from grassland species because of its distinct phenological characteristics.

While remotely sensed data can detect plant species and populations, some researchers have combined remotely sensed data with other spatial datasets to enhance predictive models. Anderson et al. (1996) used aerial photo interpretation, roads, trails, hydrographic data, slope, and aspect to model leafy spurge in Theodore Roosevelt National Park. The authors noted that using GIS and remote sensing data together proved to be a powerful combination of tools. Yellow starthistle was predicted using landscape variables such as elevation, slope, aspect, and Landsat land use classifications (Shafii et al. 2003).

Remotely sensed data can also be used to assess attributes associated with invasive plant presence. For example, many invasive plants are associated with fragmented or degraded landscapes (Sakai et al. 2001). Tanser and Palmer (1999) used a moving standard deviation filter passed over Landsat TM imagery as a measure of landscape heterogeneity. Other environmental variables that are associated with landscape

heterogeneity and appear to have high potential for remote sensing include total vegetation cover, relative proportion of grass and shrub cover, and organic soil cover (Warren and Hutchinson 1984, Schmidt and Karnieli 2000).

The use of GIS and remotely sensed data can provide efficient methods to identify some invasive species populations (Goslee et al. 2003). Predictive models may display 1) probabilities of occurrence (derived from logistic General Linear Modelling [GLM] analyses), 2) the most probable abundance (derived from ordinal GLM analyses), 3) predicted occurrence (based on non-probabilistic metrics), or 4) the most probable entity (from hierarchical analyses) (Guisan and Zimmermann 2000). These outputs allow for prioritization of sites for ground visits that will aid in early detection protocols and risk analyses.

Research Objectives

The goal of this research was to evaluate the efficacy of using remotely sensed and GIS data as tools to support early detection of invasive plants. Early detection protocols are paramount in developing conservation strategies that restrict or eradicate invasive plants in Big Bend National Park. Our proof-of-concept approach was to first create inductive models of potential invasive species habitat based on known plant occurrences in the Park. We also created spatial models of distributional pathways in the Park, and conducted a risk assessment that allows for prioritizing areas for conservation efforts.

Objectives include:

- 1) Identify and prioritize a list of invasive plant species that occur within Big Bend National Park (Chapter 2);

- 2) Create spatially explicit models of predicted distributions of targeted species using remotely sensed and GIS data, and assess models by quantifying the accuracy of the predicted habitat distributions with known occurrences (Chapter 3);
- 3) Identify and spatially model distribution vectors and pathways for the selected invasive species to conduct an assessment of risks posed by invasive species to the Park's biodiversity (Chapter 4); and
- 4) Conduct computer simulations of common field sampling designs to evaluate efficient ground survey methods to detect invasive species in BIBE (Chapter 5), using known plant occurrences, predicted habitat models, and risk assessment models created in previous objectives.

Study Area Description

Big Bend National Park is located in the northern Chihuahuan Desert in southwest Texas (Fig 1-2) and is the 15th largest National Park. The Park encompasses roughly 320,000 ha ($\approx$ 790,700 ac) with elevation ranging from 518 m (1,700 feet) at the Rio Grande, to 2,388 m (7,835 ft) at Emory Peak in the Chisos Mountains. The Park is characterized by hot summers (average temperature is 27°C [80° F]) and cool winters (average temperature is 3°C [51° F]), yet temperatures can range over 37°C (100 ° F) in the summer and less than -17°C (0° F) in the winter (Cochran and Rives 1985). Mean annual precipitation in the Park is approximately 33 cm (13 in), of which 75% falls from April to September as locally heavy thunderstorms (Cochran and Rives 1985). The Chisos Mountains average 48 cm (19 in)

of rainfall annually (Ernst et al. 2003). Thunderstorms occur on average 35 days per year, and most occur in spring. The prevailing wind direction is from the north or northwest (Cochran and Rives 1985). Big Bend National Park has three major environmental zones: the Chihuahuan Desert, the Rio Grande riparian corridor, and the Chisos Mountains (National Park Service 1983). The Chihuahuan Desert is the largest desert in North America (Beck and Gibbens 1999, Dinerstein et al. 2001) and is among the most biologically diverse desert ecoregions worldwide (Ricklefs and Schluter 1993). This vast and unique arid region supports a wide variety of habitats including grasslands, scrublands, shrub communities, and yucca woodlands as well as mixed conifer forests and oak woodlands at higher elevations (Gehlbach 1993). Creosotebush (*Larrea tridentata*) and lechuguilla (*Agave lechuguilla*) are the dominate plant species in the Chihuahuan Desert community type, occupying approximately 72% of the Park (Plumb 1987).

The Chihuahuan Desert has undergone considerable transition in vegetation communities since the mid-1800s, shifting from perennial grassland to shrubland (Buffington and Herbel 1965, Allred 1996, Beck and Gibbens 1999). Factors such as climatic changes, diversion of surface water for agricultural purposes, livestock grazing, erosion, introduction of exotic plant and animal species, and increased urban development have profoundly and negatively impacted the natural processes of the Chihuahuan Desert (Beck and Gibbens 1999).

The Rio Grande defines 118 miles of the Park's southern boundary. Riparian vegetation along the Rio Grande is dominated by saltcedar, mesquite (*Prosopis* spp.), cottonwood (*Populus deltoides*), willow (*Salix* spp.), tree tobacco (*Nicotiana glauca*),

Bermuda grass (*Cynodon dactylon*), and giant reed (*Arundo donax*).

The Chisos Mountains are located near the center of the Park and are considered to be the southern most mountains within the United States (Wauer 1996). The mountains were formed by volcanic activity and are characterized by high peaks, vertical, weathered cliffs, and pinnacles enclosed by pinyon-juniper-oak woodlands. Dominant woodland vegetation includes alligator juniper (*Juniperus deppeana*), red-berry juniper (*J. pinchotii*), weeping juniper (*J. flaccida*), gray oak (*Quercus grisea*), Emory oak (*Q. emoryi*), Grave's oak (*Q. gravesii*) and Chisos oak (*Q. graciliformis*). Chisos oak is endemic to the

Chisos Mountains. Mexican pinyon (*Pinus cembroides*), Ponderosa pine (*P. ponderosa*), quaking aspen (*Populus tremuloides*), and Arizona cypress (*Cupressus arizonica*) are also found in the Chisos Mountains in isolated patches.

Soils in BIBE have been influenced by the diverse geology, landforms, relief, and climate in the northern Chihuahuan Desert. Major soil units in BIBE include Chamberino-Chilicotel-Upton, Lajitas-rock outcrop, and the Lozier-rock outcrop-Marical, which together comprise approximately 83% of soils of BIBE (Cochran and Rives 1985) (Appendix B). These major soil types can be found across the uplands, hills, and mountains of BIBE. Most

of these soils are cobbly to very gravelly, and characterized as “desert pavement” on uplands. Less than 15% of BIBE has loamy soil types with moderate to high soil moisture holding capacity (Appendix B). Coarse-rocky soils are typically well drained and have relatively low soil water holding capacity (Cochran and Rives 1985). In BIBE, available soil water holding capacities range from 0.02 in of water per 1.0 inch of soil (0.19 cm water/cm soil) in rocky areas to 0.21 in of water per 1.0 inch of soil (0.2 cm water /cm soil) in loamy soil types (Cochran and Rives 1985). These arid soils limit the types of plants that can germinate and become established and influence the distribution of native and nonnative vegetation.

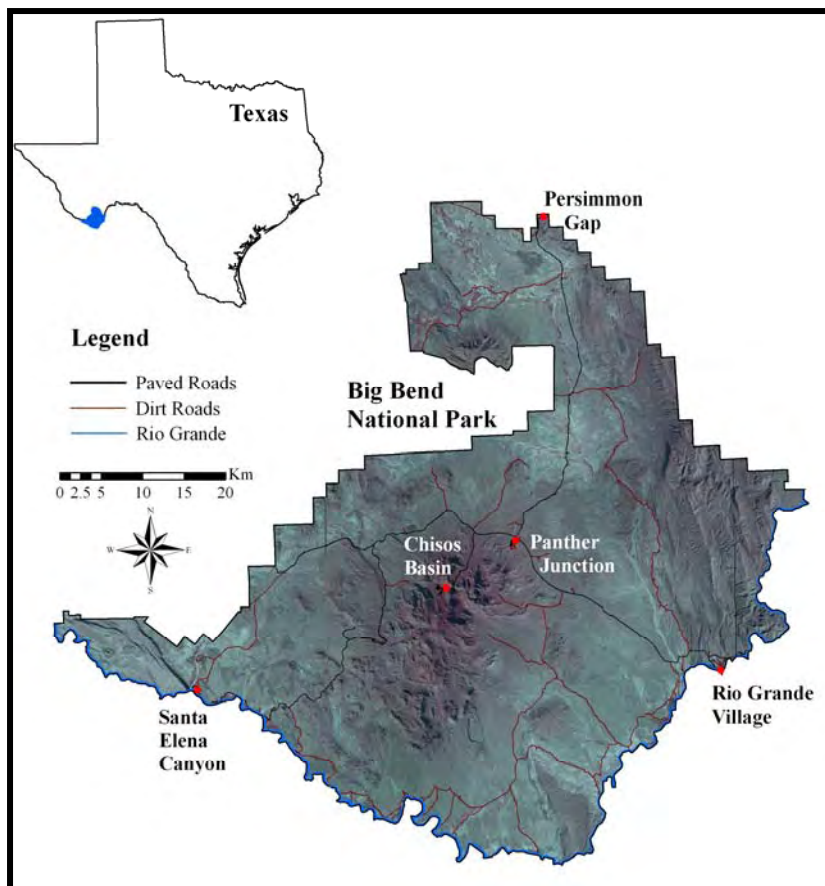


Figure 1-2. Big Bend National Park in southwestern Texas was used to test the efficacy of using remotely sensed data for the early detection of invasive plants in the Chihuahuan Desert.

Chapter 2

Invasive Species in Big Bend National Park

Invasive species are often deterrents to natural resources management. Invasive plant management is expensive, labor-intensive, and can be a long-term endeavor. Resource managers attempt to locate and eradicate new invasions before they threaten native biodiversity and ecological processes (Stohlgren et al. 1999). Once plants become established and begin to actively spread, management strategies become limited to expensive control efforts. Actively spreading species may alter ecological processes and enhance additional invasions (Cox 1999, Deloach et al. 2000, Zavaleta et al. 2001).

Increases in trans-oceanic commerce, terrestrial transportation systems, and varying land use practices has ensured that many plants have either accidentally or deliberately been introduced into places that are well beyond their native range (Moody and Mack 1988). Over 65 exotic plant species have been documented in Big Bend National Park (BIBE) in 1998 (BIBE 1998; Table 2-1). Further, in the region around the Park, the National Biological Information Infrastructure reports over 200 invasive or exotic plants (<http://cswgcin.nbii.gov/>; Appendix A) that could potentially influence the Park's environmental quality, biotic health, and ecosystem integrity (NPCA 2003).

The potential number of invasive species has led resource managers to explore predictive theories about attributes that make habitats susceptible to invasion (Rejmanek and Richardson 1996). The invasion potential of an introduced species may be considered by examining its life history characteristics,

geographic distribution, and ecological distribution. Traits that favor invasive plant establishment include: 1) habitat generalist or wide environmental tolerances, 2) allelopathy (e.g., saltcedar [*Tamarix* spp.]), 3) rapid reproduction and growth (above or below ground), 4) multiple reproductive strategies (vegetative and seed production) (e.g., giant reed [*Arundo donax*], buffelgrass [*Pennisetum ciliare*]), 5) high leaf area index, 6) small seed mass (e.g., *Pinus* spp.), 7) long flowering period or continuous seed production for as long as growing conditions permit (e.g., saltcedar), 8) unique or special strategies to disperse propagules (e.g., Russian thistle [*Salsola kali*]), and 9) the ability to regenerate from severed rootstocks (e.g., saltcedar) (Baker 1965, Goodwin et al. 1991, Rejmanek and Richardson 1996, Williamson 1996, Rejmanek 2000, Lloret et al. 2005). Habitat factors that are thought to increase the ability of plants to invade include: 1) low intensities of competition, 2) altered disturbance regimes, 3) environmental stress or disturbance, 4) previous history of invasion, 5) adequate soil and water resources, and 5) high frequency of introduced propagules, either from existing populations nearby, or through vectors and pathways (Alpert et al. 2000, Rejmanek 2000).

Unfortunately, useful predictive theories have proven elusive, because traits that consistently predict invasiveness vary by geographic location, species, and habitat (Alpert et al. 2000). Nevertheless, predictive models can provide a useful framework for prioritizing conservation efforts to prevent new introductions or control existing infestations.

Table 2-1. Exotic plant species documented in Big Bend National Park in 1998 (BIBE 1998).

Scientific Name	Common Name(s)
<i>Agrostis semiverticillata</i>	Bentgrass, Waterbent
<i>Agrostis stolonifera</i>	Creeping bentgrass
<i>Arundo donax</i>	Giant reed
<i>Asparagus officinalis</i>	Asparagus, Garden asparagus
<i>Bothriochloa ischaemum</i>	King ranch bluestem, Yellow bluestem
<i>Bromus tectorum</i>	Cheatgrass, Downy brome
<i>Caesalpinia gilliesii</i>	Bird of paradise
<i>Carex planostachys</i>	Cedar sedge
<i>Centaurea melitensis</i>	Malta starthistle, Maltese star thistle
<i>Cerastium glomeratum</i>	Sticky chickweed
<i>Chenopodium album</i>	Lambsquarters
<i>Chenopodium ambrosioides</i>	Mexican tea
<i>Chenopodium murale</i>	Nettle-leaf goosefoot
<i>Ciclospermum leptophyllum</i>	Fir-leaved celery
<i>Corispermum nitidum</i>	Neat bug seed
<i>Cynodon dactylon</i>	Bermuda grass
<i>Cyperus esculentus</i>	Chufa, Chufa flatsedge, Yellow nutgrass
<i>Digitaria ciliaris</i>	Southern crabgrass
<i>Digitaria sanguinalis</i>	Large crabgrass
<i>Echinochloa colona</i>	Jungle rice, Watergrass
<i>Echinochloa crusgalli</i>	Barnyard-grass
<i>Eragrostis barrelieri</i>	Mediterranean lovegrass
<i>Eragrostis cilianensis</i>	Candy grass, Lovegrass, Stink grass
<i>Eragrostis lehmanniana</i>	Lehmann's lovegrass
<i>Eruca sativa</i>	Arrugula, Rocket
<i>Ficus carica</i>	Common fig, Edible fig
<i>Hemarthria altissima</i>	African jointgrass
<i>Juncus bufonius</i>	Toad rush
<i>Marrubium vulgare</i>	Horehound
<i>Medicago polymorpha</i>	Bur clover, Toothed bur-clover
<i>Medicago sativa</i>	Alfalfa
<i>Melilotus albus</i>	White sweetclover, White melilot
<i>Melilotus indicus</i>	Annual yellow sweetclover, Indian sweet-clover
<i>Melilotus officinalis</i>	Yellow sweet-clover
<i>Mentha spicata</i>	Bush mint, Spearmint
<i>Morus alba</i>	Mulberry, White mulberry
<i>Nerium oleander</i>	Oleander
<i>Nicotiana glauca</i>	Tree tobacco
<i>Oplismenus hirtellus</i>	Bristle basketgrass
<i>Panicum antidotale</i>	Blue panicgrass, Blue panicum
<i>Panicum lanuginosum</i>	Panic grass

Table 2-1 Cont.

Scientific Name	Common Name(s)
<i>Parkinsonia aculeata</i>	Jerusalem thorn
<i>Paspalum dilatatum</i>	Dallas grass
<i>Pennisetum ciliare</i>	Buffelgrass
<i>Petunia parviflora</i>	Wild petunia
<i>Physalis philadelphica</i>	Mexican ground-cherry, Mexican groundcherry
<i>Polygonum convolvulus</i>	Black bindweed, Climbing buckwheat, Wild buckwheat
<i>Polygonum lapathifolium</i>	Curlytop knotweed
<i>Polygonum persicaria</i>	Lady's-thumb, Smartweed, Spotted knotweed
<i>Polypogon monspeliensis</i>	Annual rabbitsfoot grass, Rabbitfoot beardgrass,
<i>Rumex crispus</i>	Curley dock, Narrowleaf dock
<i>Salsola kali</i>	Russian thistle, Tumbleweed
<i>Sanvitalia procumbens</i>	Mexican creeping zinnia
<i>Setaria adhaerens</i>	Burr bristlegrass
<i>Setaria viridis</i>	Bottle grass, Green bristlegrass, Green foxtail
<i>Sisymbrium irio</i>	London rocket, Rocketmustard
<i>Solanum sisymbriifolium</i>	Sticky nightshade
<i>Sonchus asper</i>	Perennial sowthistle, Prickly sowthistle, Spiny sowthistle
<i>Sonchus oleraceus</i>	Annual sowthistle, Common sowthistle
<i>Sorghum bicolor</i>	Black amber, Broomcorn, Sorghum
<i>Sorghum halepense</i>	Johnson grass
<i>Tamarix aphylla</i>	Athel, Saltcedar, Tamarisk
<i>Tamarix chinensis</i>	China tamarisk, Chinese saltcedar
<i>Tamarix ramosissima</i>	Saltcedar, Tamarisk
<i>Tragus berteronianus</i>	Spiked burr grass
<i>Tribulus terrestris</i>	Bullhead, Caltrop, Goathead, Mexican sandbur, Puncture vine
<i>Ulmus pumila</i>	Chinese elm, Siberian elm

Prioritizing Invasive Species for Modeling Efforts

In 1998, Park staff evaluated the potential impacts of seven invasive plant species that were known to occur in the Park (BIBE 1998). This evaluation was primarily for invasive species that occurred in habitats outside of the Rio Grande corridor, although several invasive species also occurred in the corridor. Species evaluated were Bermuda grass (*Cynodon dactylon*), Lehmann's lovegrass (*Eragrostis lehmanniana*), horehound (*Marrubium vulgare*), buffelgrass, Russian thistle, Johnson grass (*Sorghum halepense*),

and saltcedar. They used an analytical approach based on Hiebert and Stubbendieck (1993), and Rutledge and McLendon (1996) to rank species in terms of their potential to threaten the park's biotic resources. The ranking system used numerical scores of three indices of invasiveness: 1) the ability of the plant to become a problem based on the biological and ecological ability of the species to invade new sites, 2) ability to control establishment and spread, accounting for current number and size of invasive plant patches, and 3) the potential impacts to biotic resources in the Park (BIBE 1998). This evaluation indicated that two species with the greatest negative impacts on park resources

were saltcedar and buffelgrass. Bermuda grass, Lehmann's lovegrass, and Johnson grass also yielded high potential impacts in BIBE. Giant reed was a species that was noted to be an unlikely threat outside of the Rio Grande corridor, but was included on a watch list (BIBE 1998).

We selected invasive plant species for remote sensing and GIS spatial analyses based on the evaluation and ranking (summarized previously: BIBE 1998), and through

consultation with Park Service staff. Eight species were chosen for spatial analyses (Table 2-2). All of these species were Park Service priority species, had been or were currently undergoing evaluation within BIBE, and had known spatial locations of occurrence in the Park. These species are a concern because they colonize disturbed areas quickly, outcompete native species, and alter natural processes that further promote invasive species invasions and establishment.

Table 2-2. Invasive species selected for remote sensing and GIS spatial analyses in Big Bend National Park.

Common Name	Scientific Name	Ecological Consequences
Giant Reed	<i>Arundo donax</i>	Displaces native species and forms dense stands along waterways. Stands create flood-control problems, increase fire hazard, and reduce biodiversity and habitat for wildlife.
Bermuda grass	<i>Cynodon dactylon</i>	Aggressive colonizer of disturbed sites and reduces native vegetation diversity.
Lehmann's lovegrass	<i>Eragrostis lehmanniana</i>	Aggressively spreads into desert and grassland communities where it excludes native plants, especially after disturbance. This plant alters fire frequency and intensity, and reduces biodiversity.
Horehound	<i>Marrubium vulgare</i>	Colonizes disturbed sites and reduces native biodiversity.
Buffelgrass	<i>Pennisetum ciliare</i>	Colonizes disturbed sites and reduces native grass biodiversity by altering fire frequency and intensity, crowds out native grasses and competes for limited resources.
Russian thistle	<i>Salsola kali</i>	Pioneer species in disturbed sites and prolific seed producer, reduce yield and quality of crops, depletes soil moisture, threatens native plant biodiversity, creates hazards along roads and right-of-ways, and spreads wildfire rapidly.
Johnson grass	<i>Sorghum halepense</i>	Aggressive colonizer on disturbed sites and can out compete native vegetation by crowding and by extracting water from lower soil profiles. This grass may impact native communities through allelopathy.
Saltcedar	<i>Tamarix</i> sp.	Aggressive competitor for water resources where it lowers stream flows and water tables, increases soil salinity, and displaces native species. This plant is fire-adapted and increases the frequency, intensity and effect of fires and floods.

Species Characteristics

Our objectives were to model potential invasive species habitat and evaluate the risk of occurrence and spread in BIBE. Life history information is helpful in determining the general range and environmental characteristics required for the establishment and spread of a particular species, and aids in determining appropriate spatial data needed to model invasions. Modeling endeavors that are removed from the underlying ecology of the organism may not be effective (Belovsky et al. 2004). As such, we present brief species profiles for each target species. We describe the origin of each species, provide a general description, and indicate vectors, pathways, and habitat associations that are useful in modeling potential distributions in BIBE.

Giant Reed

General Description

Giant reed is native to the Mediterranean coastal area (Bell 1997). It has been cultivated throughout Asia, southern Europe and northern Africa, and widely planted in the Americas (Perdue 1958, Bell 1997). Giant reed is thought to be first introduced into the U.S. in 1820 near Los Angeles, California, for erosion control along drainage canals (Hoshovsky 1987). Currently, giant reed has become naturalized throughout the southern U.S.

Giant reed is a tall perennial riparian plant with heights from 2-8 m (6.5-26 ft) topped by very large feathery, plume-like inflorescences (Bell 1997, Dudley 2000). Leaves are elongate, 5-8 cm (2.0-3.2 in) at the base and taper to a fine point. Leaf bases are cordate and the ligule is membranous and minutely ciliate (Perdue 1958). Giant reed culms are hollow, divided by partitions at the nodes, and can reach 1-4 cm (0.4-1.6 in) in diameter. Giant reed produces flowers from long plume-like panicles during August and September.



Giant reed can grow > 6 ft in height in BIBE. Photo taken from Jewels Camp and provided by NMCFWRU.

Spikelets are several-flowered with smaller upper florets (Perdue 1958, Felger 1990). Once established, giant reed can form large, continuous, entangled root masses. These root masses can be more than 1 m (3 ft) thick (Bell 1997).

Vectors and Pathways

Although giant reed will produce copious amounts of seed that can travel great distances by wind and water, the viability, dormancy, germination, and seedling establishment are poorly understood (Perdue 1958, Hoshovsky 1987, McWilliams 2004). New invasions are primarily due to translocation of rhizomes. Rhizomes form in dense mats that are broken up by scouring events or other physical disturbances (Bell 1997, Hughes and Mickey 1993). Invasions can occur several miles "downstream" along the riverbank from rhizome sources or in some cases slightly upland due to flooding events. Giant reed responds quickly to fire, re-sprouting faster than many native riparian species (Bell 1997).



Giant reed replacing native mesquite along the Rio Grande in Big Bend National Park. Photo provided by NMCFWRU.

Habitat Associations

Giant reed appears to be a soil generalist, and can flourish in all types of soils from heavy clays to loose sands and gravel. It produces most vigorous growth in well-drained soils where abundant moisture is available (Perdue 1958). Giant reed can grow as fast as 5 cm (2 in) a day in favorable conditions (Perdue 1958).

In the southwestern U.S., giant reed is associated with a variety of riparian and wetland habitat types, floodplains, plains, and arroyos (Dick-Peddie 1993). Giant reed requires substantial water resources for initial development, but can persist for several years in drought once established (Perdue 1958, Bell 1997). In riparian areas in the Chihuahuan Desert, giant reed occurs with saltcedar, and often displaces native willows (*Chilopsis* and *Salix* spp.), honey mesquite (*Prosopis glandulosa*), and common reed (*Phragmites australis*) (Hendrickson and Johnston 1986, Dick-Peddie 1993). In BIBE, large and well established stands of giant reed can be found along the Rio Grande (Hughes and Mickey 1993, Tracy and DeLoach 1998).

Bermuda Grass

General Description

Bermuda grass, native of Africa, was first introduced to the U.S. in 1751 in South Carolina (Northam et al. 1991, Elmore and Cudney 2002). It has since rapidly spread to southern regions of the U.S. from California east to the southeastern states (Holt 1989, Horowitz 1972a). Bermuda grass is commonly used as turf or lawn grass. Cultivars also exist in cool temperate climates of Washington, Idaho, Utah, Colorado, Iowa, Michigan, New York, Massachusetts, and New Hampshire (Belesky et al. 1991, Carey 1995). Bermuda grass is considered to be one of the most serious invasive species in the world (Holm et al. 1977).

Bermuda grass is a long-lived, warm season perennial with short (up to 40 cm; 15.7 in tall), jointed culms. Bermuda grass grows in spring and summer months, and becomes dormant when the weather cools in the fall. It flowers from July to October. Panicles have two to seven digitate spikes, each approximately 2.5-6.4 cm (1.0-2.5 in) long (Carey 1995). Stolons are prostrate with the ground and root at the nodes.



Bermuda grass is also common on the banks of the Rio Grande in Big Bend National Park. Photo provided by NMCFWRU.

Vectors and Pathways

Bermuda grass can reproduce by stolons, rhizomes, and seeds, although it spreads most rapidly by stolons and rhizomes (Carey 1995). Bermuda grass is often used as livestock forage and seeds pass unharmed through livestock. Bermuda grass has deep roots and is capable of extending roots up to 150 cm (59 in) during drought stress (Hay et al. 1991).

Habitat Associations

Bermuda grass grows on a wide range of soils and salinities. Productivity is greater on fertile, sandy to silty soils, or alluvium (Carey 1995). Bermuda grass is not shade tolerant. This plant occurs in rangelands, pastures, fields, riparian areas, and the understory of open woods, forests, and orchards. Bermuda grass requires ample water for establishment and will grow in irrigated areas of any type including roadside runoff areas, springs, seeps, stock tanks, and adjacent to streams and marshes (Burton and Hanna 1985).

Bermuda grass is an early successional grass species. It is an aggressive colonizer of locations subject to frequent disturbances such as grazing, flooding, grading, and fire (Harlan and De Wet 1969, Dong and de Kroon 1994). Once established, this plant will resist scouring on stream banks and can survive being submerged in water for several weeks (Horowitz 1972b).

In BIBE, Bermuda grass was planted in lawns and developed areas in the Rio Grande Village and Cottonwood campgrounds. This species can be considered the dominant grass in many places along the Rio Grande, roadside runoff areas, stock tanks, and along Route 13 between Basin Junction and Croton Springs (BIBE 1998). Bermuda grass has replaced some native associations in the Rio Grande floodplain in BIBE (Boer and Schmidly 1977).



Lehmann's lovegrass occurs in disturbed roadside runoff areas and developed areas in Big Bend National Park. Photo provided by NMCFWRU.

Lehmann's Lovegrass

General Description

Lehmann's lovegrass originates from southern Africa (Martin and Cox 1984, Anable et al. 1992), and has spread aggressively into native grass and grass-shrublands in the southwest. Introduction in the southwest was in 1932 with a series of experiments conducted near Tucson, Arizona (Cox et al. 1988). Starting in the early 1950s, commercial seed growers produced large quantities of Lehmann's lovegrass which was planted from Texas to Arizona to prevent soil erosion and provide livestock forage (Cox et al. 1988, McClaren and Anable 1992). Lehmann's lovegrass is presently in southwestern states ranging from California to Texas (Kartesz 1999).

Lehmann's lovegrass is a warm-season, perennial bunchgrass that reaches heights from 45-61 cm (1.5-2.0 ft) (Humphrey 1970). Plants are mostly erect and form short basal offsets from the base of the stem at nodes. Panicles are somewhat open, and do not form

a compact crown. Leaves are approximately 0.15 cm wide (0.06 in) by 5-15 cm (2-6 in) long (Uchytel 1992).

Vectors and Pathways

Lehmann's lovegrass spread by seed and stolons. It is a prolific seed producer, and can maintain a reserve seed supply that will persist for several years in the soil and quickly take advantage of any moisture available (Uchytel 1992). Lehmann's lovegrass is also stoloniferous. Stems that come in contact with the ground may root at the nodes. Lehmann's lovegrass is very competitive, and colonizes disturbed sites rapidly (Anable et al. 1992). Seedlings emerge following summer rains on sites disturbed by burning, mowing, or grazing (Roundy et al. 1992). Highly variable moisture regimes favor establishment because a portion of seeds remain dormant even if favorable conditions exist (Abbott and Roundy 2003). This allows for subsequent establishment after first attempts have perished due to desiccation. Established plants form nearly pure stands after several growing seasons (Humphrey 1994). New invasions are limited to 6-10 m (20-33 ft) adjacent to existing stands unless seeds are transported by livestock waste or vehicle use in contaminated areas (Anable et al. 1992, Robinett 1992). Seeds can pass unharmed through livestock and this plant is a common



Horehound can be found around Panther Junction and Basin developments in Big Bend National Park. Photo provided by NMCFWRU.

contaminate in feed hay, therefore this plant can easily establish in livestock feeding areas (Anable et al. 1992). Roads and utility right-of-ways, livestock use areas, and fires scars are the primary pathways for spread.

Habitat Associations

Lehmann's lovegrass can establish on most soils in the southwestern U.S. during a wet summer. However, the grass dies out during the next few years unless sites exhibit sandy or sandy loam textured soils (Cox et al. 1988). Elevations of successful Lehmann's lovegrass establishment in the southwestern U.S. range between 2,477 and 5,050 ft (775-1,540 m), with annual rainfall between 11-20 in (28-50 cm) (Cox et al. 1988). This grass does not tolerate low temperatures (Uchytel 1992), which might limit its potential range in the U.S. Lehmann's lovegrass establishes easily in disturbed areas, but can also colonize undisturbed areas that exhibit very low natural canopy cover (Sumrall et al. 1991, Roundy et al. 1992, Ruyle et al. 1988). In the southwestern U.S., it is associated with desert shrub and semi-desert grassland communities (Uchytel 1992).

In BIBE, Lehmann's lovegrass is well established in disturbed roadside run off areas and developed areas along routes 13, 14 and 15 as well as in Panther Junction (BIBE 1998). This plant has not been found in lower elevation areas along the Rio Grande. It does not appear that this plant is spreading into undisturbed communities of Big Bend; however, areas adjacent to established plants could become contaminated if moisture regimes change or disturbances occur.

Horehound

General Description

Horehound is native to northern Africa, Asia, and Europe, and is now common across most of the U.S. (Magness et al. 1971, Simon et al. 1984). This plant is a commercially produced

herb used in cooking, candy, and in cough medicine (Magness et al. 1971). Horehound is an aromatic perennial herb with 25-45 cm (10-18 in) long square stems (Sievers 1930, Simon et al. 1984). Leaves are 2.5-5.0 cm (1-2 in) in length, opposite, with a densely crinkled surface, ovate to near round and resemble mint in appearance (Sievers 1930, Stritzke 1975). Stems and leaves have white woolly pubescence. Flowers are white, occur in clusters on the upper part of the stem, and have a bilabiate corolla. Horehound flowers from June to September (Sievers 1930).

Vectors and Pathways

Little information is available on the spread of horehound. Reproduction is from seed only. Seeds have hooked burrs that are easily transported by attaching to people and animals and can remain viable in the soil for many years (Ainsworth 1999). Seed transportation also occurs from commercial and garden uses.



Buffelgrass emerging along a roadside near Boquillas Canyon in Big Bend National Park. Photo provided by NMCFWRU.

This plant is a common contaminant in feed hay, and therefore can easily establish in livestock feeding areas. This plant species has a slow rate of growth and will rarely persist with competition from native species (Lippai et al. 1996).

Habitat Associations

Horehound can establish in poor infertile soils and is often the first colonizer of a recently eroded area (Lippai et al. 1996). Horehound has wide environmental tolerances, including broad temperature and precipitation ranges (Simon et al. 1984). Horehound occurs in disturbed sites along roadsides, stockyards, fields, pastures, near dwellings, and dry riverbanks in scattered patches (Sievers 1930). In BIBE, horehound occurs in small isolated patches in Panther Junction and Basin developments (BIBE 1998). It can also be found sporadically in horse corrals and trails used by pack animals.

Buffelgrass

General Description

Buffelgrass is native to Africa, Asia, and the Middle East and was introduced into the southwestern U.S. for livestock forage (Holt 1985, Cox et al. 1988, Ibarra et al. 1995). Buffelgrass has been detected across the southern U.S. states and in New York (USDA NRCS 2006, Tu 2002).

Buffelgrass is a warm-season perennial bunchgrass that ranges from 10-150 cm (4-59 in) in height. It actively grows in the summer and fall. Flowering inflorescences are dense, cylindrical, 2-13 cm (0.8-5.0 in) long by 1.0-2.6 cm (0.4-1.0 in) wide. Each inflorescence has a purple, gray, or yellowish color. Inflorescences have 30-50 involucre bracts. Spikelets are clustered into burrs, and are surrounded by numerous bristles (Hickman 1993, Tu 2002).

Vectors and Pathways

Buffelgrass can reproduce either vegetatively or by seed (Bashaw and Johns 1983), although the vegetative rate of spread is slow (USDA NRCS 2006). Buffelgrass is a prolific seed producer and can produce seeds asexually without pollen transfer. Seed spread rate is considered moderate (USDA NRCS 2006), and is aided by bristles on seeds that allow for easy dispersal by wind and animals. Seeds can survive up to three years in the soil and will germinate 10 days after rain events, but seedlings grow very slowly and may be out competed by other plants (Bovey et al. 1984). Buffelgrass grows at a higher density than native grass species, enabling fires to burn at higher intensity than what is normally encountered in sparse desert environments (Martin et al. 1999). Buffelgrass will sprout immediately after a fire resulting in a pure stand after several fires and eliminate all competition in an area (Hamelton and Scifres 1982). Buffelgrass actively spreads along medians and shoulders of highways in the southwest and colonizes areas disturbed by clearing or grazing.

Habitat Associations

Buffelgrass is associated with loam, sandy loam, clay loam, and sandy clay loam soils, and actively spreads by seed in sandy loam soils where sufficient soil moisture is available (Humphreys 1967, Mutz and Scifres 1975). Buffelgrass seedlings gradually lose vigor and die in silt, silt loam, silty clay loam, silt clay and clay soils (Cox et al. 1988). Plants are not known to survive in areas with consistent below freezing temperatures (Hussey and Bashaw 1996).

Vegetation associations for buffelgrass have not been clearly defined. Buffelgrass occurs in semi-desert grasslands and shrublands, disturbed areas, pastures, roadways and utility corridors. This grass is found along roadside runoff areas and developed areas of BIBE. It



A large patch of Russian thistle along the North Rosillos Road in northern BIBE. Photo provided by NMCFWRU.

is common and dense in Rio Grande Village, Boquillas Canyon, and Castelon Route 16 areas (BIBE 1998). Buffelgrass has also been found along route 15 between Study Butte and Croton Springs, and sporadically at Panther Junction, route 11 and 12 (BIBE 1998). It has been found at Chisos Basin Campground at over 1,525 m (5,000 ft) although this is considered to be outside of its normal elevation range.

Russian Thistle

General Description

Russian thistle is native to Eurasia and is believed to have been introduced into North Dakota from Eurasia in the 1870s (Young and Evans 1985). By 1893, it was reported in California, perhaps spread by wind dispersion and aided by railroad shipments of cattle (Shinn 1895).

Another common name for Russian thistle is tumbleweed. Almost everyone recognizes the mature form of Russian thistle, which looks like the skeleton of a normal shrub. Russian thistle is an annual forb that is highly branched and rounded in form. Growth size varies greatly but may be as big as 1.5 m (5 ft) wide (Howard 1992). Stems can be bright green and usually have red or purple stripes.

Leaves are small, spiny-tipped, with inconspicuous flowers in the leaf axils. The small, winged seed is retained in the leaf axils and is dispersed as the plant skeleton rolls across the landscape. The root system consists of a shallow taproot (Allen 1982).

Vectors and Pathways

Russian thistle is prolific seed producer and spreads entirely by seed. Seeds mature in late fall when the plant stem separates from the root (Young 1991). Seeds are carried on dislodged stalks of wind blown plants that can move 400 m (1,312 ft) before spreading the majority of its seed (Stallings et al. 1995). New invasions typically occur in accordance to prevailing wind direction from existing populations. Seeds do not persist in soil beyond one growing season and will not germinate if buried greater than 8 cm (3.1 in) (Wallace et al. 1968). The seed is considered a complete embryo that will grow within 45 minutes of exposure to moisture (Wallace et al. 1968). Seeds will germinate on the surface of very compacted substrates but may not persist (Allen 1982). Seedlings are easily out-competed and crowded by established native plants (Allen 1982). This plant will only persist in areas that are continually disturbed or in areas that are barren and not suitable for other colonizers (Allen 1989).

Habitat Associations

Russian thistle is found on almost all soil types. It is found in unoccupied sites, disturbed areas, fields, overgrazed pastures, roadsides, and grasslands and desert communities (Young and Evans 1972, Young 1991). Within BIBE, Russian thistle can be found on recently disturbed sites or in repeatedly disturbed sites (BIBE 1998). Dense stands have been found at the old dumpsite on Grapevine Hills road, on Route 13, west of Maverick Station along River Road, and in the region north of Buttrill Spring on North Rosillos Road (BIBE 1998).



Johnson grass in a riparian area in northern Big Bend National Park. Photo provided by NMCFWRU.

Johnson Grass

General Description

Johnson grass is native to the Mediterranean region of Europe and Africa (Holm et al. 1977, McWhorter 1989). Johnson grass was apparently first introduced into South Carolina around 1830 for livestock forage, but it rapidly spread across the southern U.S. (Tellman 1998, Howard 2004). Currently, Johnson grass is fairly widespread through the contiguous U.S. (Great Plains Flora Association 1986, Wunderlin 1998).

Johnson grass is a tall (heights may reach 3.7 m or 12 ft) warm-season perennial (Anderson 1961, Radford et al. 1968, Martin and Hutchins 1980, Diggs et al. 1999). Leaves have a prominent midvein. Johnson grass flowers from May to November in the Southwest (Martin and Hutchins 1980, Diggs et al. 1999). Inflorescence ranges from 10-60 cm (4-24 in) with an open panicle. Seeds are approximately 2 mm (0.08 in) in length (Radford et al. 1968) and may have twisted awns that aid in seed dispersal (Diggs et al. 1999). The leaves of Johnson grass respond distinctly to solar radiation and therefore can be distinguished easily from other plants by remote sensing (McWhorter 1989).

Vectors and Pathways

Johnson grass can reproduce through seed production and rhizomes. Wind, water, machinery, and animals disperse Johnson grass seed (Ghersa et al. 1993, Hartzler et al. 1991). Winds during thunderstorms (around 50 km/hr [31mi/hr]) carried Johnson grass seed up to 1.0 km (0.62 mi) from parent plants (Ghersa et al. 1993). Seeds are also dispersed by flowing water. Farming equipment also spreads seeds (Ghersa et al. 1993). Johnson grass seed is a contaminant in hay and commercial seed (Allen 1990). Johnson grass plants begin growing rhizomes in the seedling stage. A single plant may produce 60-90 m (200-300 feet) of rhizomes (McWhorter 1981). Rhizomes are drought-resistant, remaining viable after drying to 40% of initial harvest weight (Anderson et al. 1960).

Characteristics of Johnson grass that aid in its spread include: 1) formation of dense rhizomes that host meristematic tissue responsible for regenerating plants (Anderson et al. 1960), 2) moderate drought resistance (Anderson et al. 1960), 3) salt tolerance (Yang et al. 1990), 4) abundant seed production with seeds that remain viable for 2 to 5 years (Huang and Hsiao 1987, Leguizamon 1986, Allen 1990, Unger et al. 1999), and 5) may produce toxins that are allelopathic (Warwick and Black 1983).

Habitat Associations

Johnson grass is associated with a variety of habitats. It is most common in ecosystems below 1,800 m (6,000 ft) with moist to mesic moisture regimes, especially riparian habitats (Howard 2004). Johnson grass is associated with open habitats and does not persist under closed canopies. Johnson grass can be found in irrigation canals, flood plains, springs, and stock tanks (Bendixen 1988, Monaghan 1979).

Johnson grass is found in various sized patches throughout BIBE in depressions,

ditches, and waterways that have had historic disturbances (BIBE 1998). Large patches exist in Tornillo Flat and along North Rosillos road near Basin Campground. It does not appear to be expanding into healthy native community types that are not subject to disturbance (BIBE 1998).

Saltcedar

General Description

Saltcedar was introduced to the U.S. in the late 1800s from Europe and Asia as an ornamental and for windbreaks and erosion control (DiTomaso 1998). Currently, saltcedar is widespread along water courses, springs, lakes, arroyos, floodplains, roadsides, and residential areas throughout the southern U.S. (Christensen 1962, Swenson and Mullins 1985, DeLoach 1991, Brock 1994), and is spreading into the northern states (USDA NRCS 2006). Saltcedar covers nearly 0.6 million ha (1.5 million ac) of floodplain in 23 states (DiTomaso 1998, Zavaleta 2000). Several species of saltcedar exist in the U.S. and are difficult to distinguish by morphological characteristics in the field



Saltcedar along the Rio Grande in Big Bend National Park. Photo provided by NMCFWRU.

(Allred 2002, DiTomaso 1998). Common species that occur in the desert southwest and BIBE are *T. chinensis* and *T. ramosissima*, but other species are known to occur (DiTomaso 1998, BIBE 1998).

Saltcedar is a perennial woody plant that is typically < 6 m (< 20 ft) tall with numerous basal branches or trunks. The bark on younger branches and saplings is reddish-brown or purplish, and larger trunks are reddish-brown to dark brown or blackish. Leaves are scale-like, < 0.35 mm (< 0.01 in) in length, alternate, sessile or sheathing, with salt-secreting glands. The inflorescence is a panicle of small, perfect flowers, subtended by a small bract. *Tamarix ramosissima* and *T. chinensis* flower in the spring and summer (Allred 2002).

Vectors and Pathways

Saltcedar has short generation times, grows rapidly, and can spread vegetatively and through seed dispersal. This plant produces very small seeds that have small hairs on the apex of the seed coat that aid in wind and water dispersal. Although seeds are short-lived (< 45 days), they are produced continuously during warm temperatures (Frasier and Johnsen 1991, Dudley et al. 2000, Young et al. 2004). Water dispersed seeds are deposited along sandbars, lakeshores, and riverbanks (Brotherson and Field 1987). Reproduction also occurs by translocation of stem or root fragments. This plant re-sprouts robustly from rootstock following stem cutting or fire. Saltcedar has deep roots that are tolerant to extended drought and inundation. It is also a fire-adapted species that burns easily and resprouts quickly (Frasier and Johnsen 1991).

Habitat Associations

Habitat types most susceptible to saltcedar invasion include floodplains, riparian



Saltcedar, giant reed, and Bermuda grass displace native riparian vegetation along the Rio Grande. Photo taken from Loop Camp overlook, Big Bend National Park. Photo provided by NMCFWRU.

communities, seasonal wetlands, lake margins, disturbed and undisturbed waterways, bottomlands, moist rangelands, and other areas where seedlings can be exposed to extended periods of saturated soil (Frasier and Johnsen 1991). Saltcedar has also been planted in residential and commercial areas. Saltcedar will grow in any soil type and can tolerate highly saline and alkaline conditions (DiTomaso 1998). In Utah, the plant often occupies sites with silt loams and silt clay loams that have high organic matter and intermediate moisture (Brotherson and Winkel 1986). Saltcedar leaves accumulate salt on surface glands that will precipitate onto the soil increasing salinity levels and reducing competition from salt intolerant plants (Friederici 1995, Dudley et al. 2000).

Saltcedar was reported to occur in 43% of all water sources (spring, seeps, wells, and stock tanks) in BIBE in 1995 (BIBE 1998). It is a common tree in the Rio Grande corridor, and large trees can be found in Banta Shut-in, Bone Spring, Bee Spring, McKinney Well, Santa Spring, and Solis Spring.

Chapter 3

Invasive Species Predictive Habitat Models

Introduction

Remote Sensing Considerations

Over the last decade, there has been a substantial increase in the use of remotely sensed and GIS data to model invasive species distributions or potential habitats (Chudamani et al. 2004). This increase coincides with improved remote sensors, computer technology, and classification techniques (Lass et al. 2005).

There are a variety of remotely sensed data that may be used to detect invasive plants or potential invasive plant habitats (Table 3-1). Data chosen for individual modeling endeavors depend on several variables including 1) fiscal considerations, 2) goals and objectives, 3) size of management units or area of interest (e.g., park, forest, or region size), 4) distribution and patch sizes of current (known) invasive plants, 5) invasive plant species (species that are phenologically different than the surrounding land cover), 6) spatial and spectral resolution of sensor and repeat cycle, and 7) the availability of computing resources.

The ability of remote sensing data to detect invasive plants or habitats is associated with the sensors' spatial and spectral (bandwidth) resolution and the sensors' repeat cycle. Spatial resolution describes the amount of detail an image contains across a given distance, which is typically a cell size. The ability to "resolve," or describe small details or objects, is one way of describing spatial resolution. Smaller objects are typically better "resolved" or detected with high-resolution images. For example, IKONOS and QuickBird images have spatial resolutions < 5 m (16.4 ft) and are considered high-resolution

images. Landsat sensors have spatial resolutions around 30 m (98.4 ft) and are considered medium-resolution images. Advanced Very High Resolution Radiometer (AVHRR) is considered a low-resolution with a spatial resolution of 1.1 km (0.68 mi).

Spectral resolution describes the amount of data captured across the electromagnetic spectrum. In general terms, this is the number of bands that compose the wavelength (e.g., bandwidths) (Table 3-1). For example, multispectral sensors are sensitive to radiation within a narrow wavelength band, typically hosting ≤ 7 bands across the electromagnetic spectrum. These scanners have been successfully applied to discriminate between broad land cover types. Multispectral sensors include Landsat 7 Enhanced Thematic Mapper Plus (ETM+), SPOT, IKONOS, and QuickBird, among others. Conversely, hyperspectral sensors acquire data in more (10 to several 100) but narrower (from 10 to a few nm wide) spectral bands than multispectral sensors. This precise spectral data allows for capturing finer spectral characteristics (less variability per bandwidth) which yield better identification of objects.

Sensors that yield high spatial resolution data and have hyperspectral capabilities have the highest likelihood of detecting microhabitats or rare plants (Marcus 2002, Marcus et al. 2003, Lass et al. 2005, Lawrence et al. 2006). However, these data also tend to be expensive and their relatively small swath size (ground area of the image) requires extensive computer processing time and storage for analyses of large areas. As such, landscape scale modeling endeavors often compromise bandwidth and spatial resolution considerations.

Table 3-1. Spatial and spectral resolution of remote sensors that may be useful for detecting potential invasive plants or modeling their habitats. This list is not inclusive of all possible remote sensors available.

Sensor	Spatial Resolution	# of Bands	Wavelength Range (um)	Swath	Repeat Path	Comments
Multispectral, Low Spatial Resolution Sensors						These sensors have limited ability to detect invasive plants, but can be used for large areas with invasive plant infestations > 1ha that host > 20% canopy cover.
Advanced Very High Resolution Radiometer (AVHRR)						
	1.1 km	5	0.58 – 12.5	2,399 km	2 per day	
Multispectral, Medium Spatial Resolution Sensors						These sensors are good for small or large areas that contain infestations with > 20% canopy cover, or for plants that have a unique vegetation phenology than the surrounding landscape. Data are reasonably priced with good temporal availability.
Landsat 7 Enhanced Thematic Mapper Plus (ETM +)						
	30-60 m	7	0.45-2.35	185 km	16 days	
SPOT (5)						
	2.5-20 m	4	0.50-1.75	60 - 80 km	26 days	
Multispectral, High Spatial Resolution Sensors						These sensors show much promise for detecting small populations of invasive plants. Their high spatial resolution may allow for detecting plants with low canopy cover. Due to large data volumes and the expensive nature of the data, these data are best suited for small areas of interest.
IKONOS						
	4 m	4	0.45-0.85	23.9 km	Various	
QuickBird						
	2.4 m	4	0.45-0.90	16.5 km	3-7 days	
Hyperspectral Sensors						These sensors are perhaps the most helpful group of remote sensors for detecting small populations of invasive plants because they sample the electromagnetic spectrum in narrow continuous increments which allows for improved identification of species. Due to large data volumes and the expensive nature of the data, these data are best suited for small areas of interest.
NASA Jet Propulsion Laboratory Airborne Visible/Infrared Imaging Spectrometer (AVIRIS)						
	20 m	224	0.40 - 2.45	11 k	By Request	
PROBE-1 Hyperspectral Instrument						
	1 – 10 m	128	0.40 - 2.45	< 1 - 6 km	By Request	

The repeat cycle of the sensor is also a key criterion to consider when selecting imagery. Similar spectral signatures between target plants and the surrounding environment, changes in soil color or moisture, and low plant densities hinder discrimination efforts for invasive plants. However, seasonal differences in plant phenology may help in detecting invasive plants. Some invasive

plants flower or green-up at a different time than the surrounding vegetation. Multiple image dates allow for detecting these phenological differences between target plants and the surrounding landscape. Imagery dates should correspond to critical recognition phases of the target plant (Hobbs 1990, McGowen et al. 2001, Chudamani et al. 2004).

Types of Remotely Sensed Data

Low Spatial Resolution Sensors

Multispectral, low spatial resolution sensors have limited ability to detect individual invasive plants or small populations. The Advanced Very High Resolution Radiometer (AVHRR) sensor is an example. The ground resolution of 1.1 km (0.68 mi) prohibits its ability to detect small invasive plant populations. However, the AVHRR sensor has useful temporal data, with continuous global coverage since 1979. Two daily (morning and afternoon) acquisitions make AVHRR data useful for detecting landscape changes or evaluating degraded landscapes (Eve et al. 1999). Bradley and Mustard (2005) used AVHRR data and time series analyses to estimate cheatgrass (*Bromus tectorum*) area in the Great Basin. Further, large dense patches of broom snakeweed (*Gutierrezia sarothrae*) were detected using this sensor (Peters et al. 1992).

Medium Spatial Resolution Sensors

Multispectral, medium spatial resolution sensors such as Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+), and SPOT have been used extensively to model landscape vegetation types and conduct landscape change analyses. Landsat sensors have varied as technologies have improved. The latest Landsat satellite is Landsat 7, which was launched in 1999. The sensor on Landsat 7, the ETM+, replicates the capabilities of the TM instruments on Landsat 4 and 5. Landsat 7 ETM+ has the same spectral bands as the previous Landsat sensors, but the ETM+ also features a panchromatic band with 15 m (49.2 ft) spatial resolution and a thermal IR channel with 60 m (196.8 ft) spatial resolution. A mechanical failure of the scan line corrector onboard Landsat 7 in May 2003 resulted in present data gaps from this sensor. Images from the SPOT sensor are available at four levels of resolution (2.5, 5, 10, and 20 m; 8.2, 16.4,

32.8, and 65.6 ft). SPOT satellites can also be programmed to revisit a given geographic area at any specific time. This service allows obtaining imagery specifically for a vegetation phenology event.

Landsat TM and ETM+ have been extensively used to model broad vegetation types. In some cases, these sensors can identify individual plant species with unique spectral or temporal characteristics (Parker-Williams and Hunt 2002). Dewey et al. (1991) compared dyers woad (*Isatis tinctoria*) locations with spectral classes created from Landsat 5 TM data in northern Utah. The authors found strong associations between 10 spectral classes and dyers woad locations. They concluded that their remotely sensed predictive model provided resource managers with a powerful tool for estimating potential dyers woad distributions. Several authors have suggested that Landsat TM and ETM+ data are best used for detecting invasive plants that have patch sizes around 0.5 ha (1 ac) or larger (Everitt and DeLoach 1990, Everitt et al. 1992, Anderson et al. 1993). McGowen et al. (2001) utilized Landsat 5 TM to map serrated tussock (*Nassella trichotoma*) and scotch thistle (*Onopordum* spp.) in Australia. Detections were limited to areas with infestations > 20% groundcover. Likewise, cheatgrass in the Great Basin was modeled using Landsat TM and ETM+ data (Bradley and Mustard 2005).

High Spatial Resolution Sensors

Multispectral, high spatial resolution sensors such as IKONOS and QuickBird have < 5 m (16.4 ft) spatial resolution. QuickBird's panchromatic band has a spatial resolution of 60 cm (23.6 in). These high spatial resolution sensors show promise for detecting individual species and capturing plant phenological state (Asner and Warner 2003, Turner et al. 2003, Wang et al. 2004), and mapping shallow aquatic habitats (Mumby and Edwards 2002).

Tsai et al. (2005) used QuickBird imagery to accurately map the spatial extent of the invasive horse tamarind (*Leucaena leucocephala*) in southern Taiwan. IKONOS and QuickBird are commercial products by Space Imaging and Digital Globe, respectively. IKONOS and QuickBird also have relatively small swath sizes (around 20 km; 12.4 mi); thus, analyses of large areas require extensive computer processing time and storage.

The oldest remote sensing method is aerial photography (Sabins 1987, Lillesand and Kiefer 1994). Historically, the use of aerial photographs was limited to small areas because of the high cost of data acquisition (Lass et al. 2005). Advances in digital aerial photography have improved both the spectral and spatial resolution. Digital cameras can be attached to a variety of aircraft providing greater flexibility with resolution and timing. There are also a variety of commercial developers that specialize in obtaining aerial photographs and videography (Lass et al. 2005).

Aerial photographs or videography have been found to be useful in detecting saltcedar (*Tamarix ramosissima*) (Everitt et al. 1992; 1996) and leafy spurge (Everitt et al. 1995, Anderson et al. 1996). Further, aerial photographs were used as a tool to estimate the rate of spread for giant hogweed (*Heracleum mantegazzianum*) in the Czech Republic (Mullerova et al. 2005) and for mimosa (*Mimosa pigra*) in northern Australia (Lonsdale 1993). Color digital aerial photos flown in a helicopter 100 m (328 ft) above ground level were used to census haleakala silversword (*Argyroxiphium sandwicense*) within a remote volcano crater in Hawaii (Landenberger et al. 2003). This method allowed the authors to survey large remote areas in less time than ground based surveys.

Hyperspectral Sensors

Hyperspectral sensors partition the electromagnetic spectrum into narrow continuous increments allowing for improved identification of small populations of invasive plants. Sensors are attached to a variety of aircraft which provides flexibility in spatial resolution. There are many hyperspectral sensors available from both governmental and commercial use. For example, NASA Jet Propulsion Laboratory Airborne Visible/Infrared Imaging Spectrometer (AVIRIS) has a 20 m (65.6 ft) resolution with 224 spectral bands that measure light from 0.4 to 2.5 μm in 0.01 μm increments. As such, each 20 m pixel in the image has 224 spectral attributes of data. Likewise, the Hyperion sensor has 220 spectral bands (0.4 to 2.5 μm), with a 30 m (98.4 ft) resolution, and the Probe-1 sensor has 128 spectral bands. Spatial resolution of the Probe-1 sensor varies between 1.0 and 10 km (0.62- mi) depending on height of the aircraft.

Several invasive plants have been detected with the hyperspectral sensors. DiPietro (2002) used AVIRIS data to discriminate giant reed (*Arundo donax*) from riparian vegetation in California. Likewise, AVIRIS data was used to estimate leafy spurge canopy cover and distribution in Wyoming (Parker-Williams and Hunt 2002; 2004). Lawrence et al. (2006) experienced high accuracy in detecting spotted knapweed (*Centaurea maculosa*) and leafy spurge (*Euphorbia esula*) using the Probe-1 sensor. Other hyperspectral sensors have been used to detect spotted knapweed (Lass et al. 2002; 2005), yellow starthistle (*Centaurea solstitialis*) (Lass and Thill 2000), and babysbreath (*Gypsophila paniculata*) (Lass et al. 2005). Small infestations of leafy spurge were identified in southeastern Idaho using HyMap hyperspectral data (Glen et al. 2005).

Analytical Approaches to Data Modeling

Statistical approaches to constructing species distribution models (SDMs) allow enhanced predictive power over deductive approaches. Species distribution models are fundamentally statistical representations relating the likelihood of a species occurrence to a set of predictor variables. Statistical analyses attempt to discriminate between response values (e.g., species presence and absence) using a set of environmental predictors. When the environmental predictors host spatial information, the resultant model can be extrapolated across the extent of digital data. This process has been facilitated by GIS and remotely sensed data, allowing assessment of the distribution of resources over large and even inaccessible areas (Guisan and Zimmermann 2000).

Many analytical approaches have been applied to determine statistical relationships between species and predictor variables (Guisan and Zimmermann 2000, Elith 2002). Examples of analytical methods commonly used in species distribution models include generalized linear models (GLMs) (McCullagh and Nelder 1989), generalized additive models (GAMs) (Hastie and Tibshirani 1990, Yee and Mitchell 1991), environmental envelopes (Busby 1986), classification trees (Breiman et al. 1984, De'ath and Fabricius 2000, Liaw and Wiener 2002, Lawrence et al. 2006), and genetic algorithms (Stockwell and Peters 1999). There is no one analytical method that works for all scenarios. The appropriate analytical method for constructing predicted habitat models is a function of goals and objectives, and the type and structure of the data. For example, GLMs are a class of statistical models that includes analysis of variance, regression, and logistic regression and have been widely used in species distribution modeling (Elith 2002). GLMs require data on species presence (or abundance) and absence. Often land

managers lack species absence data or adequate information on species detection probabilities. A common sampling error is implying that non-detection represents species' absences (Mackenzie et al. 2002, Weidong and Swihart 2004). Errors in presence/absence classifications can greatly skew wildlife habitat relationship models (Weidong and Swihart 2004).

Recently, there is growing interest in modeling efforts using presence-only data, e.g., datasets containing only observations of the organisms with no data on where the species was absent (Elith 2002, Arriaga et al. 2004, Graham et al. 2004, Argaez et al. 2005, Pearce and Boyce 2006, Phillips et al. 2006). These techniques are valuable when absence data are either unreliable or unavailable, or when absence data is too costly to gather.

A relatively new presence-only spatial modeling approach is the use of maximum entropy. The principle of maximum entropy estimates the target probability distribution that has the broadest distribution compatible (uniform distribution) with the information available, while recognizing that the available information is incomplete (Jaynes 1957, Phillips et al. 2004). Essentially, maximum entropy assigns positive weight to every possibility that is not ruled out by the initial data. This provides an unbiased representation of the dataset (Jaynes 1957).

An example maximum entropy analytical program for spatial datasets is MaxEnt (Phillips et al. 2006). MaxEnt is a generative approach verses a discriminative approach towards model building. Models are based on the probability of inputs given a positive response, verses GLM which models the probability of a response given inputs. MaxEnt uses pixels with known species occurrence records to constitute sample points. Remotely sensed and GIS datasets can

provide the environmental variables measured at each sample point. Analysis output includes a probabilistic interpretation, grading from least to most suitable habitat conditions.

In this Chapter, we present predictive habitat models for giant reed (*Arundo donax*), Bermuda grass (*Cynodon dactylon*), Lehmann's lovegrass (*Eragrostis lehmanniana*), horehound (*Marrubium vulgare*), buffelgrass (*Pennisetum ciliare*), Russian thistle (*Salsola kali*), Johnson grass (*Sorghum halepense*), and saltcedar (see Chapter 2) in Big Bend National Park (BIBE). We also present a predictive habitat model of all eight target species combined to investigate areas where management efforts could impact multiple species.

Methods

Target Species Occurrences

Big Bend National Park staff provided spatial data on the occurrences of the target invasive plants in the park. This dataset included present and historic (areas where eradication and remediation efforts were applied) plant locations from opportunistic sightings and planned roadside surveys from 2001 to present.

We conducted additional ground surveys in June of 2006 throughout BIBE to ascertain additional locations of target species populations. We used a systematic sampling approach to optimize our chance of detecting additional invasive plant populations. We used known locations of existing invasive plant populations to concentrate ground survey efforts. We also systematically surveyed roads and trails throughout BIBE. Arroyos, springs, and cross-country surveys were also conducted. Once a targeted species was identified,

Global Positioning System (GPS) locations (UTM Zone 13, GRS 1980 Spheroid and NAD 83 Datum) and digital photographs were collected. Spatial data provided by BIBE and from our ground survey efforts were consolidated for predictive modeling.

Spatial Datasets

Since the goal of this research was to evaluate the efficacy of using remotely sensed data to develop predictive models that could be used as early detection tools for invasive plants, we developed our predictive habitat models based on remotely sensed spectral data. We used Landsat 7 ETM+ data for this study because of the large size (roughly 320,000 ha; 790,734 ac) subjected to analyses. The cost associated with purchasing high resolution or hyperspectral data for all of BIBE were beyond the fiscal allocation of this project. Further, some of the target invasive plants had unique vegetation phenology or distinct habitat associations. We used multiple dates/seasons (Table 3-2) of imagery to account for dynamic habitat conditions that occurs in the study area due to discrete rainfall events. Multiple dates and seasons also provided a means to capture phenological differences among the targeted species and the surrounding landscape. We acquired Fall 1999, Summer and Fall 2000, Spring and Fall 2001, Spring 2002, and Spring 2003 Landsat 7 ETM+ data from America View

Table 3-2. Dates and seasons of imagery used to model potential invasive plant habitat in Big Bend National Park.

Year	Season	Scene 3140	Scene 3040
1999	Fall	September 30	September 23
2000	Summer	May 27	July 23
2000	Fall	October 02	November 28
2001	Spring	April 28	March 20
2001	Fall	October 21	October 14
2002	Spring	March 30	April 8
2003	Spring	January 28	March 26

(<http://www.americaview.org>) and the Texas View Remote Sensing Consortium (<http://www.texasview.org>). Data was obtained pre-processed to Level 1-G, which includes radiometric and geometric correction. This pre-processing is part of the USGS National Land Archive Production System. Two Landsat scenes were required for complete coverage of BIBE: 3140 and 3040, although most of BIBE is within scene 3040 (Fig. 3-1). Imagery was acquired in UTM zone 13, WGS 84 projection and was reprojected to UTM Zone 13, GRS 1980 Spheroid and NAD 83 Datum. Image mosaic was performed in Erdas Imagine 9.0 using feather blending of overlapping regions. Results were clipped to BIBE boundary provided by the Park. Each band in each mosaic (one per year/season) was extracted into Generic ASCII raster format for data analyses needed by MaxEnt software.

Predictive Habitat Models

MaxEnt software version 2.0 (<http://www.cs.princeton.edu/~schapire/maxent/>) was used to create predictive habitat models for each target species. Known species occurrences were used for the sample locations, and the Landsat 7 ETM+ datasets provided the environmental variables measured at each sample point. A description of how the program functions can be found in the software tutorial, help section, and from Phillips et al. (2004; 2006).

We used the default settings for the regularization multiplier, iterations, convergence threshold, and the number of background points (Fig. 3-2). A random sample of 10,000 background pixels was used to represent the variety of environmental conditions present in the data. The optimum distribution was then computed using the union of the background pixels and the pixels from plant occurrence locations. We randomly withheld 10% of the occurrence

points for each target species to assess model performance.

MaxEnt uses the principle of maximum entropy to estimate the target probability distribution that has the broadest distribution compatible with the information available (Phillips et al. 2004). Outputs include a probability surface with values ranging from 0 to 100 and tabular and graphical representations of model performance and variable contribution.

Variable Selection

We used analyses provided in MaxEnt to guide model variable selection. MaxEnt uses a jackknife approach to evaluate which variables are most important in the model. Each variable is excluded, and a model created with the remaining variables. Next, a model is created using each variable in isolation. The contribution of each variable can be compared in relation to the general model using a measurement of gain. The gain is a measure of the likelihood of the sample locations. For example, a gain of 3.0 means

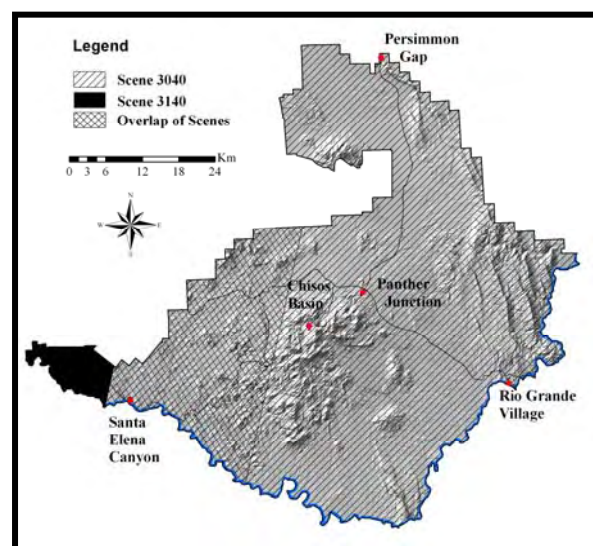


Figure 3-1. Two Landsat 7 ETM+ scenes (3040 and 3041) were conjoined to provide complete coverage of Big Bend National Park.

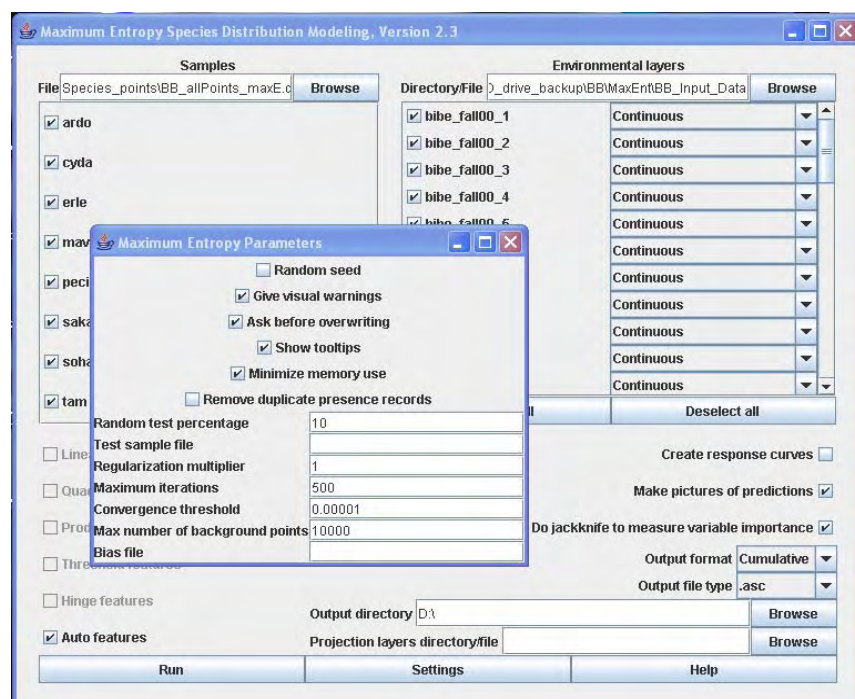


Figure 3-2. (Left) MaxEnt program settings used to estimate potential invasive plant habitats in Big Bend National Park.

that the average sample likelihood is $\exp(3.0) \approx 20$ times higher than that of a random background pixel (Phillips et al. 2007). Gain starts at zero (uniform distribution), and increases as the probabilities of the sample locations increase. In addition, MaxEnt creates response curves that evaluate the contribution of a variable in relation to the mean of all other variables for the occurrence locations. Using these procedures, we removed variables that appeared to provide little contribution to model predictions.

Model Performance

Robust model performance procedures typically employ independent data, i.e., data not used in developing the predictive model. Model performance can be assessed based on threshold-dependent metrics and threshold-independent metrics. Threshold-dependent metrics require a known threshold to classify a response as presence/absence, or suitable/not suitable. These metrics are often summarized in a confusion matrix (Table 3-3). The matrix is populated by tallying the number of observations that fall into each of the four

possible categories. Percent agreement and disagreement (error of commission and omission) are often summarized or further analyzed to evaluate the association between observed and predicted.

Alternatively, threshold-independent metrics are not based on the selection of a specified threshold for classifying the predicted observation into binomial outcomes (e.g., 0/1, present/absent, or suitable/not suitable). Instead, model performance is evaluated across the continuum of thresholds from 0 to 1.0. Receiver operating characteristic (ROC) curves are a threshold-independent metric used for model assessments. ROC analyses are also independent of species prevalence (Pearce and Ferrier 2000). ROC curves are created by fitting a smooth curve to plots of sensitivity (true positive occurrences) and specificity (false positive occurrences) across a range of thresholds (Table 3-3). The area under the ROC curve (AUC) provides a summary measure of the model's discrimination ability, i.e., ability to differentiate suitable from non-suitable

Table 3-3. Confusion matrix to derive measurements of model performance.

Predicted	Actual	
	Present	Absent
Present	a	b
Absent	c	d

Omission (False Negative Rate) = $c/(a+c)$

Commission (False Positive Rate) = $b/(b+d)$

Sensitivity = $a/(a+c) = 1 - \text{Omission Rate}$

Specificity = $d/(b+d) = 1 - \text{Commission Rate}$

habitats (Phillips et al. 2006). AUC values range from 0.5-1.0, with values between 0.5-0.7 indicating low discriminate ability, values between 0.7-0.9 indicating moderate discriminate ability, and values > 0.9 indicating high discriminate ability (Pearce and Ferrier 2000, Manel et al. 2001).

MaxEnt evaluates model performance by testing if a model performed significantly better than random (Phillips et al. 2006). This approach, considered threshold-dependent, employed a binomial test (Wilcoxon signed-rank test) based on omission and predicted area. Model performance is evaluated using extrinsic omission rate (fraction of the test localities that fall into pixels not predicted as suitable) and the proportion of all the pixels that are predicted as suitable habitat. MaxEnt output yields a variety of threshold-dependent values. Users can choose a threshold value based on their objectives (Fieldings and Bell 1997). For example, an issue may require greater emphasis be placed on the ability to accurately predict species presence (specificity) rather than species absence (specificity). As such, a threshold weighted towards specificity would be selected.

We selected a threshold that provided an equal tradeoff between the test data sensitivity and specificity. Values below this threshold were considered unsuitable habitat and values above the threshold were considered to have a high probability of providing adequate habitat for the targeted species. Selection of threshold values allowed for the calculation of suitable habitat area, and was used in risk assessments (See Chapter 4).

We also evaluated model performance using threshold-independent ROC curves (Phillips et al. 2006). The AUC value provides a single measure of model performance, and can be interpreted as the likelihood that habitat quality in the predictive model is correctly classified at a random positive site and a random negative site (Phillips et al. 2004). We present AUC values accompanied by their 95% Confidence Intervals.

Results and Discussion

Sample Locations

We spent approximately 400 hrs systematically searching roads, trails, campgrounds, rivers, drainages, and developed areas for invasive plants in June 2006 (Fig 3-3). Combining this survey effort and plants documented by BIBE, a total of 1,464 invasive plant locations were documented for the eight species of interest (Table 3-4; Appendix C). The number of occurrence points used to generate individual species models ranged from 26-627. Small sample sizes (< 50 occurrences) occurred with giant reed, horehound, and Russian thistle. As a result, the number of samples withheld (10%) to evaluate model performance for these three species was ≤ 3 . The influence of small sample sizes on model performance is poorly understood. However, Phillips et al. (2004) recommended 50-100 samples for optimal MaxEnt models.

There were 388 invasive plant locations that hosted multiple species. These locations occurred throughout the Park, but were especially prevalent along the Rio Grande corridor and developed areas (e.g., saltcedar, giant reed, and Bermuda grass). Additional survey effort would provide additional target plant or plant population locations in BIBE.

Performance of Predictive Models

The performance of predictive models produced by MaxEnt can be qualitatively (visual inspection) and quantitatively (AUC values) assessed. ROC curves allow for a visual interpretation of a model's performance. Good predictive models yield a high sensitivity (true positive occurrences) rate while the specificity (false positive occurrence) rate is low. As such, good predictive models display ROC curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Poor models display ROC curves near the diagonal, random prediction line. Likewise, visual interpretation of the omission versus predicted area provides insight to model performance. An optimal model would have an omission rate close to the predicted omission (diagonal line).

Visual interpretation of ROC and omission

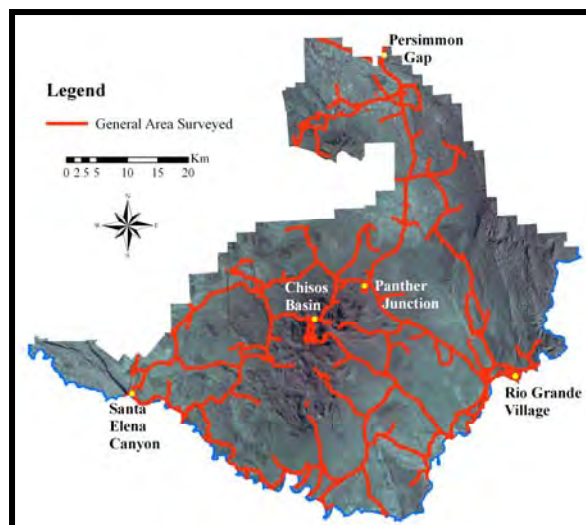


Figure 3-3. General area surveyed (red) for target invasive plants in Big Bend National Park in June 2006.

curves in Figures 3-4 to 3-12 indicates all predictive plant models performed better than a random prediction, with the exception of Russian thistle (Fig. 3-9). Training AUC values ranged from 0.86 to 0.99, while test AUC values ranged from 0.77 to 0.99 (Table 3-5). Confidence intervals (95%) built around test AUC values indicate models did not vary across discrimination classes (low moderate, or high), with the exception of Russian thistle (Fig. 3-13).

Table 3-4. Number of georeferenced target invasive plant occurrences (individual plants or populations) used to construct predictive habitat models in Big Bend National Park.

Common Name	Scientific Name	Number of Known Occurrences	Number of Training Samples	Number of Test Samples	Number of Predictive Variables Retained in Model
Giant Reed	<i>Arundo donax</i>	31	28	3	20
Bermuda grass	<i>Cynodon dactylon</i>	117	106	11	34
Lehmann's lovegrass	<i>Eragrostis lehmanniana</i>	253	228	25	34
Horehound	<i>Marrubium vulgare</i>	26	21	2	16
Buffelgrass	<i>Pennisetum ciliare</i>	627	565	62	37
Russian thistle	<i>Salsola kali</i>	38	35	3	22
Johnson grass	<i>Sorghum halepense</i>	147	133	14	35
Saltcedar	<i>Tamarix</i> spp.	225	203	22	36
All 8 plants (duplicate locations removed)		1076	969	107	36

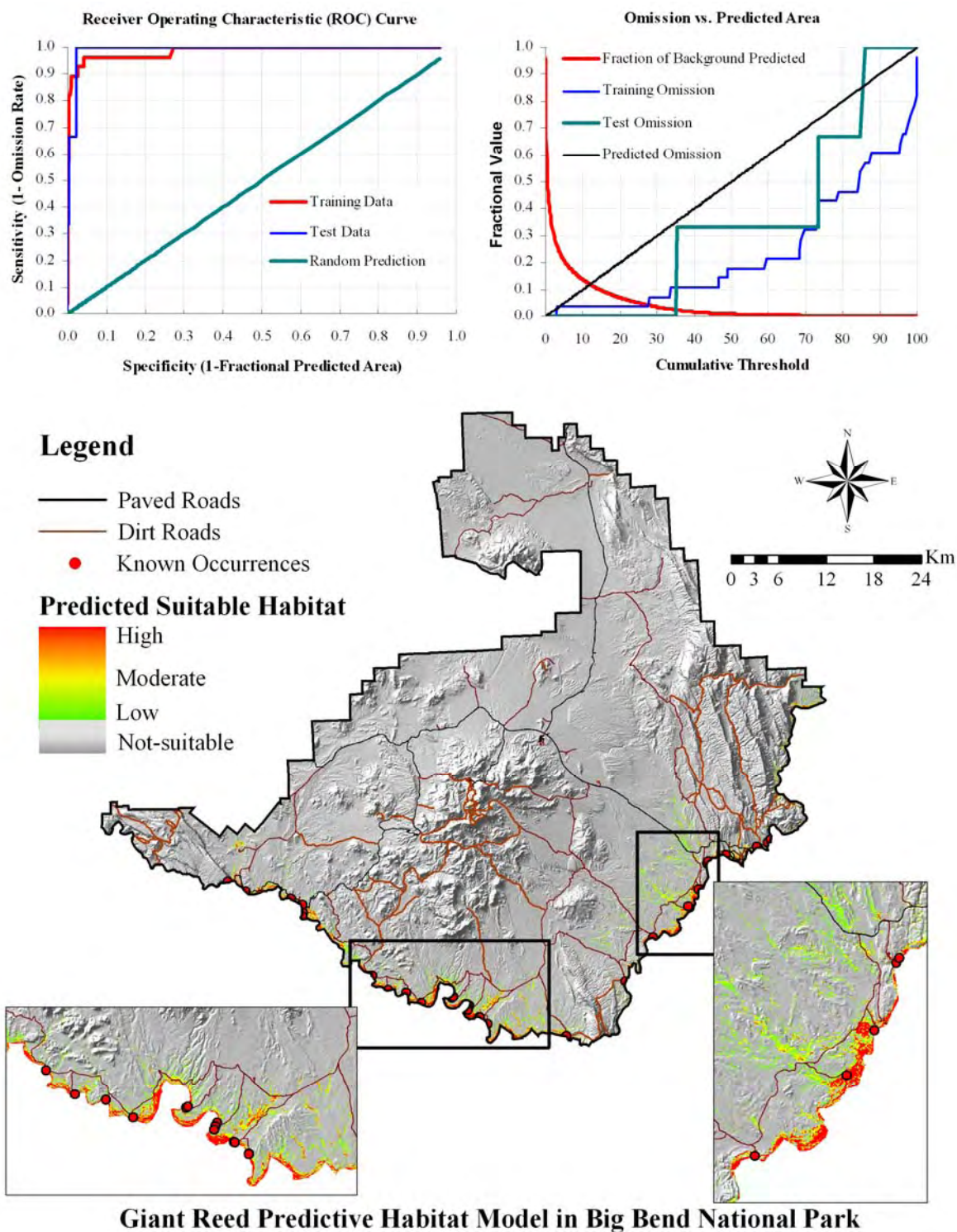
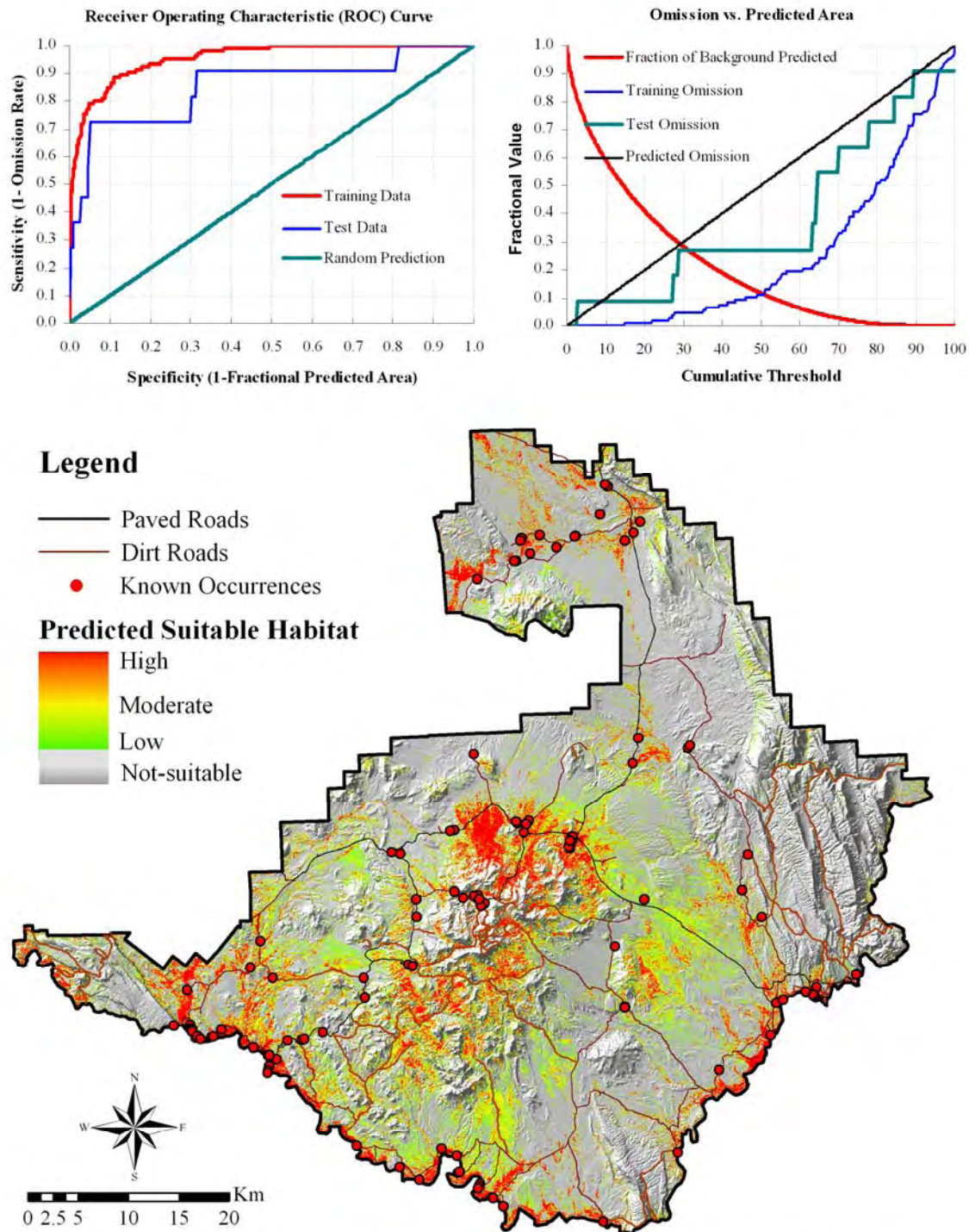
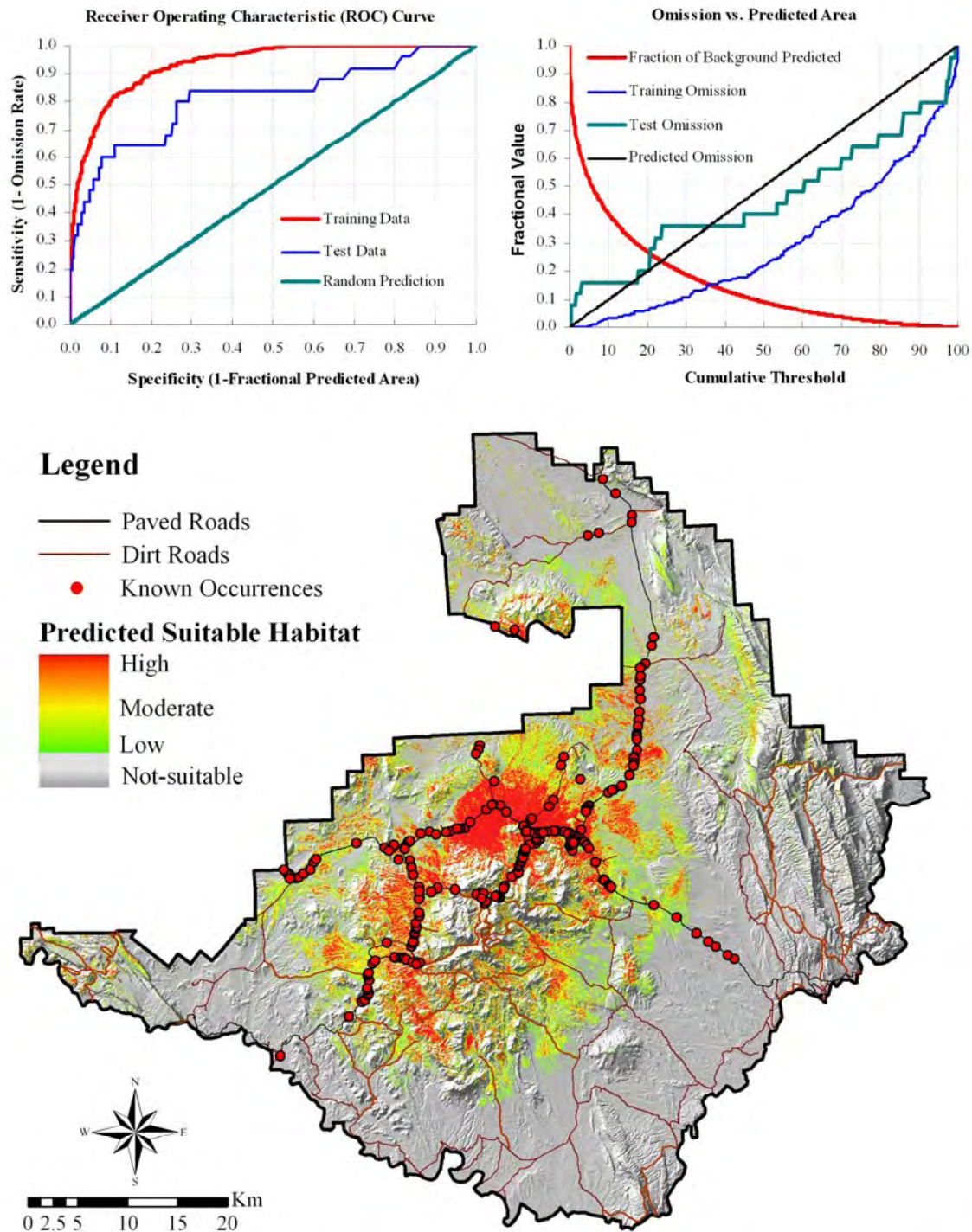


Figure 3-4. Potential distribution of giant reed in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



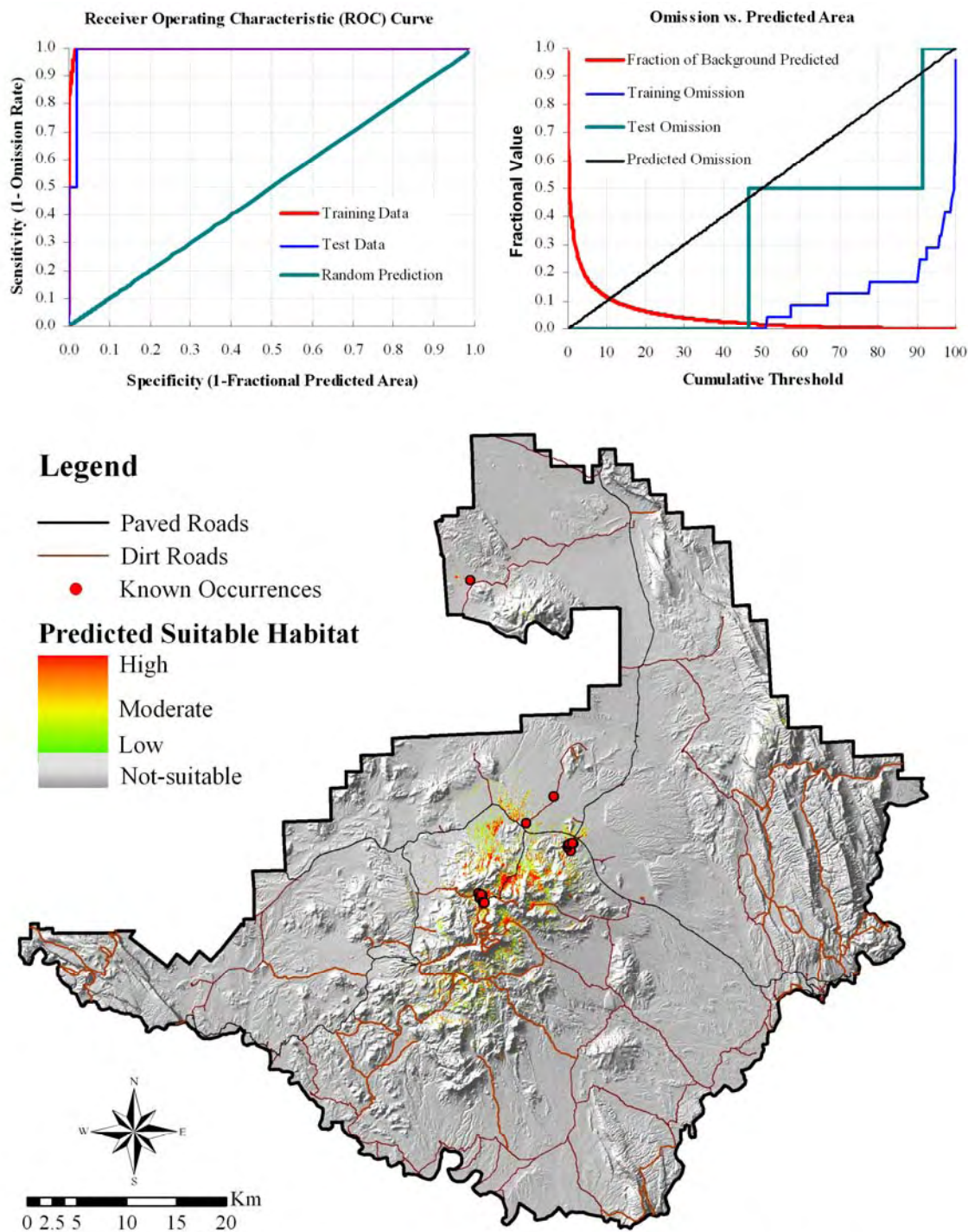
Bermuda Grass Predictive Habitat Model in Big Bend National Park

Figure 3-5. Potential distribution of Bermuda grass in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission versus Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



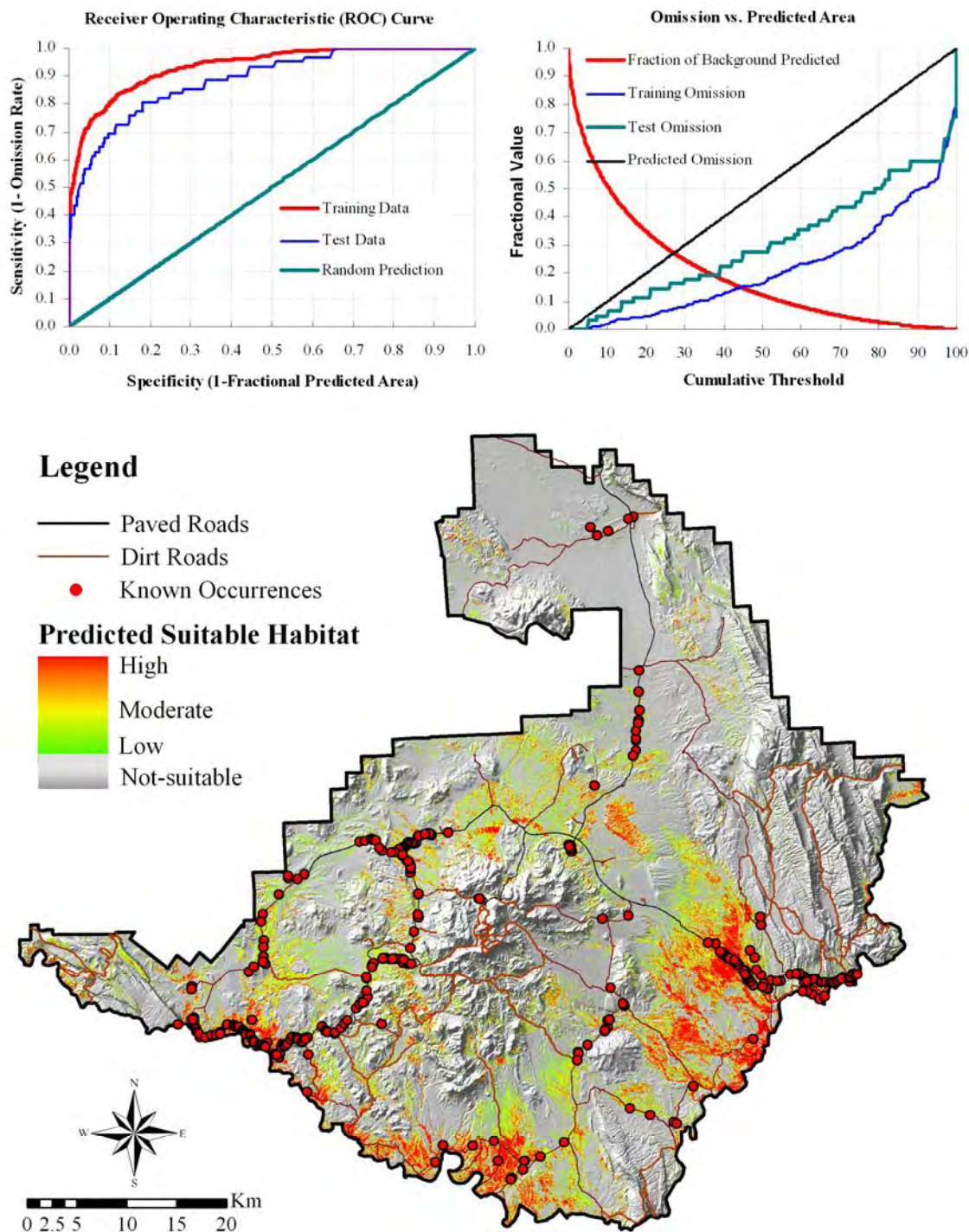
Lehmann's Lovegrass Predictive Habitat Model in Big Bend National Park

Figure 3-6. Potential distribution of Lehmann's lovegrass in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission versus Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



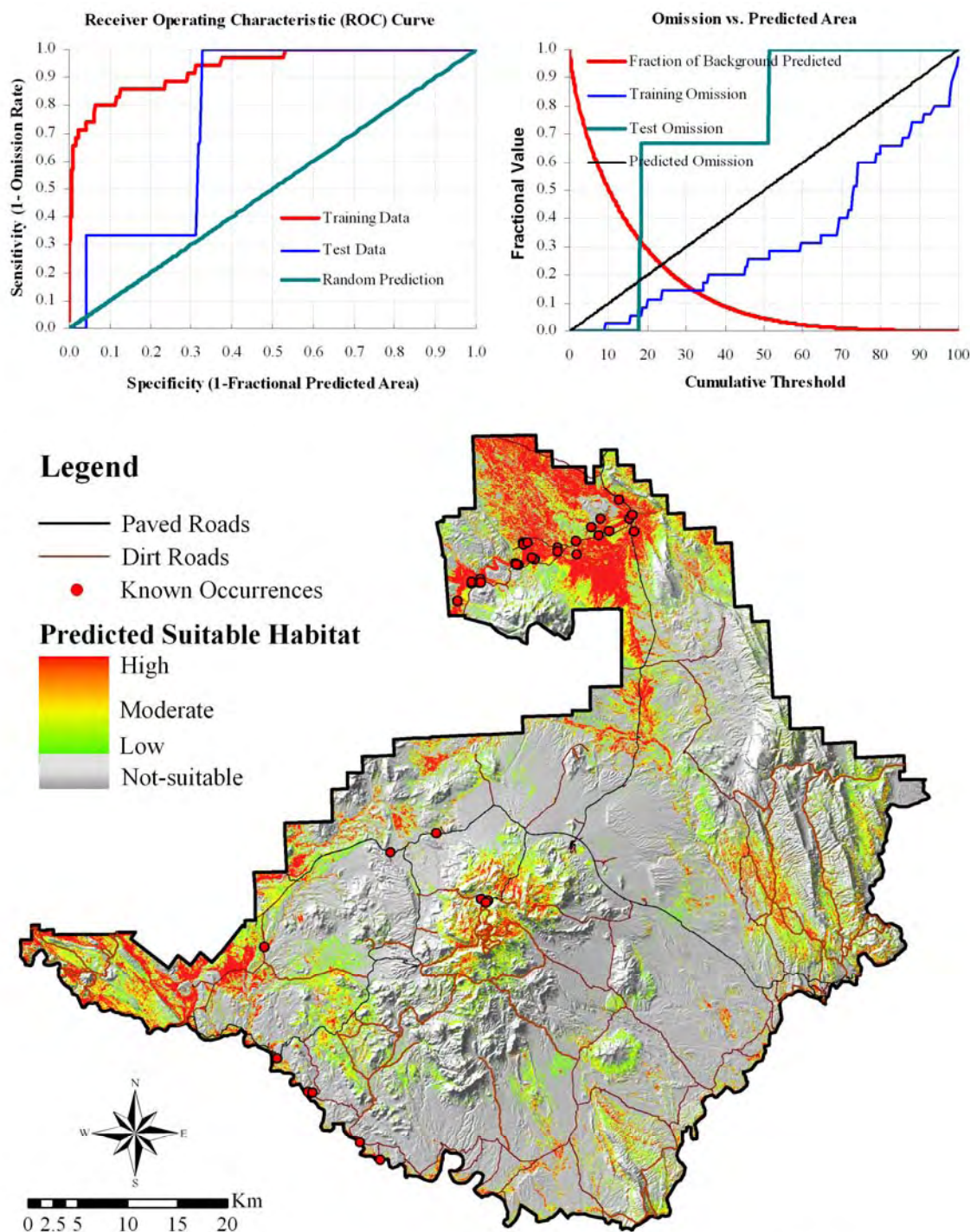
Horehound Predictive Habitat Model in Big Bend National Park

Figure 3-7. Potential distribution of horehound in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



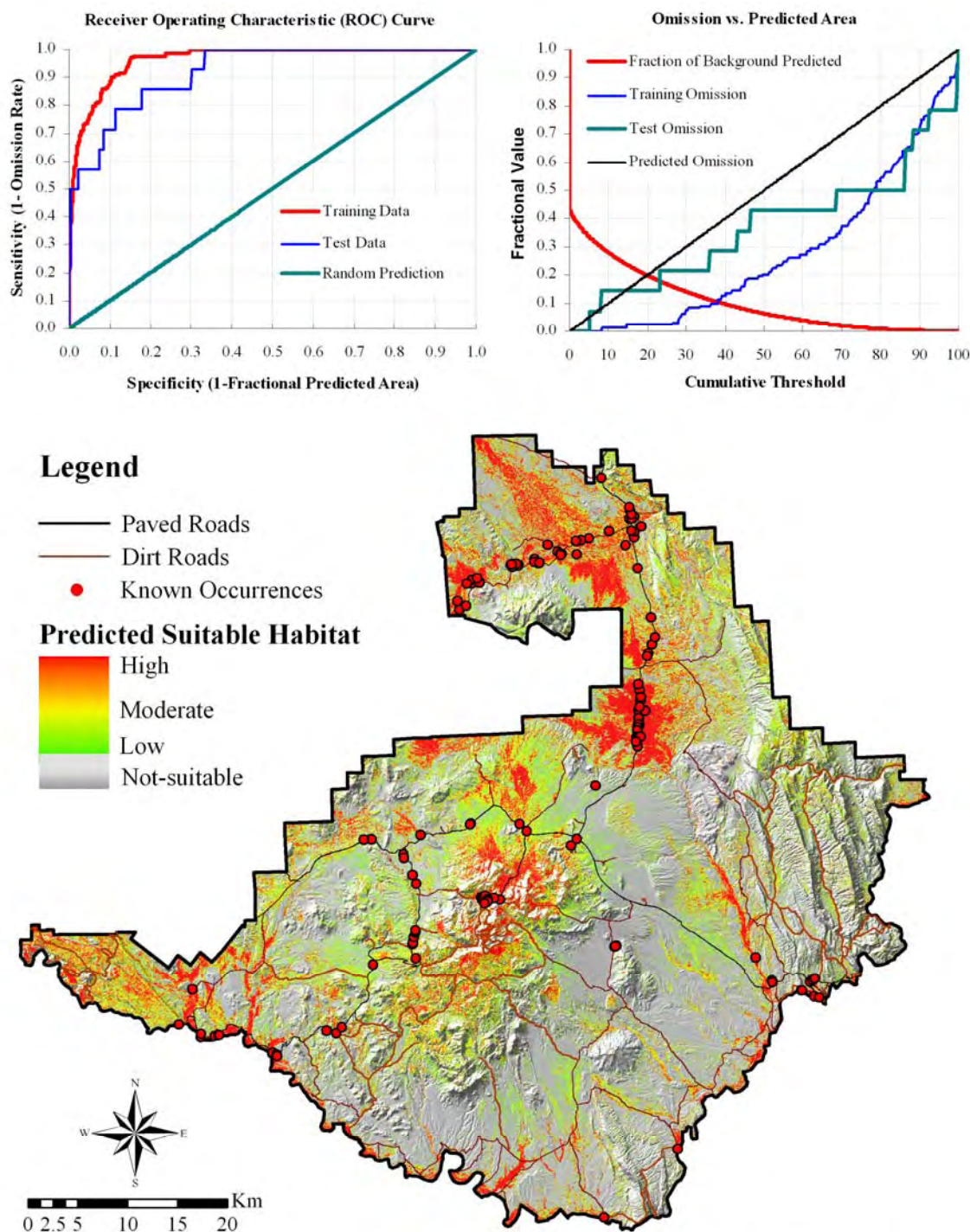
Buffelgrass Predictive Habitat Model in Big Bend National Park

Figure 3-8. Potential distribution of buffelgrass in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



Russian Thistle Predictive Habitat Model in Big Bend National Park

Figure 3-9. Potential distribution of Russian thistle in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



Johnson Grass Predictive Habitat Model in Big Bend National Park

Figure 3-10. Potential distribution of Johnson grass in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).

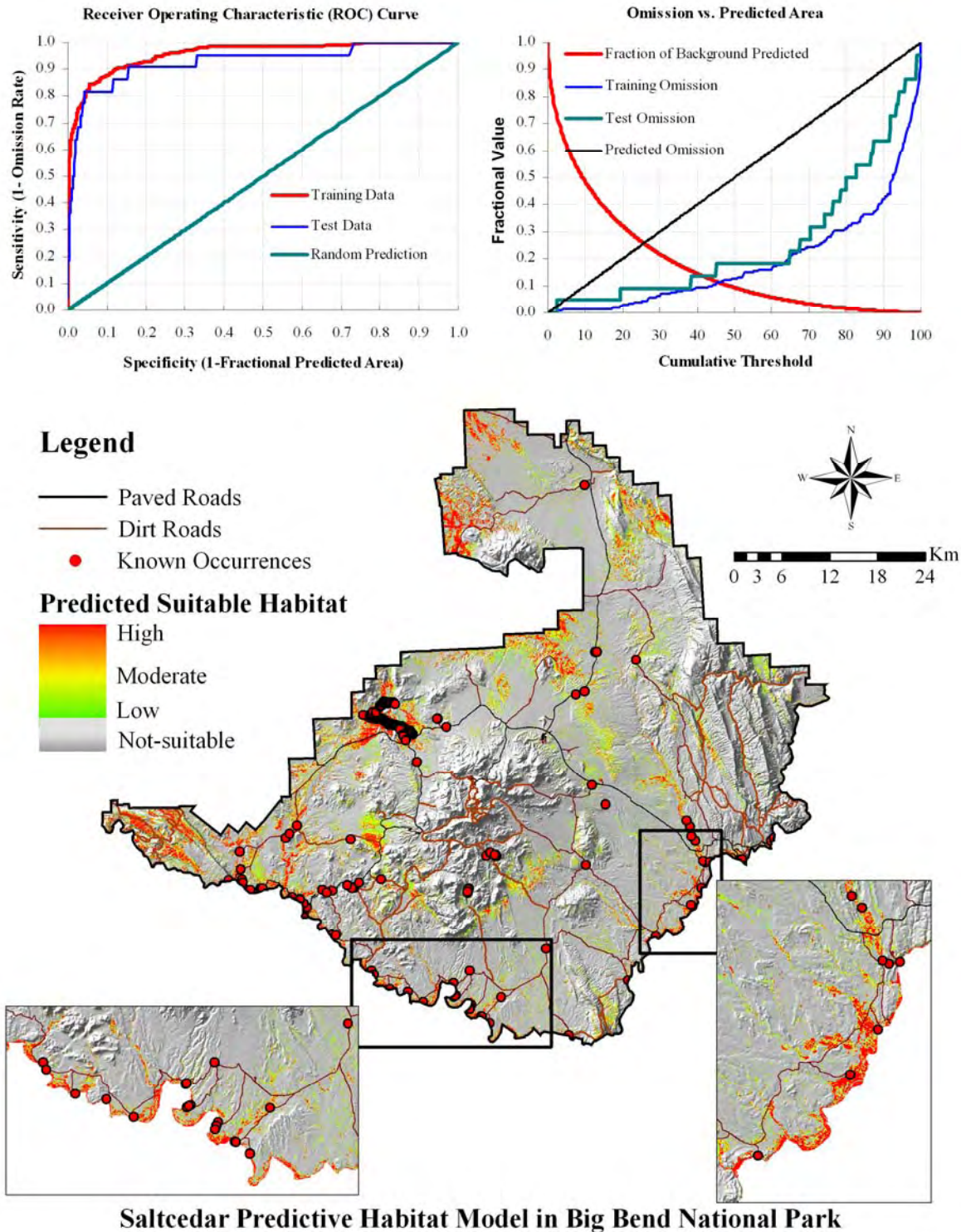
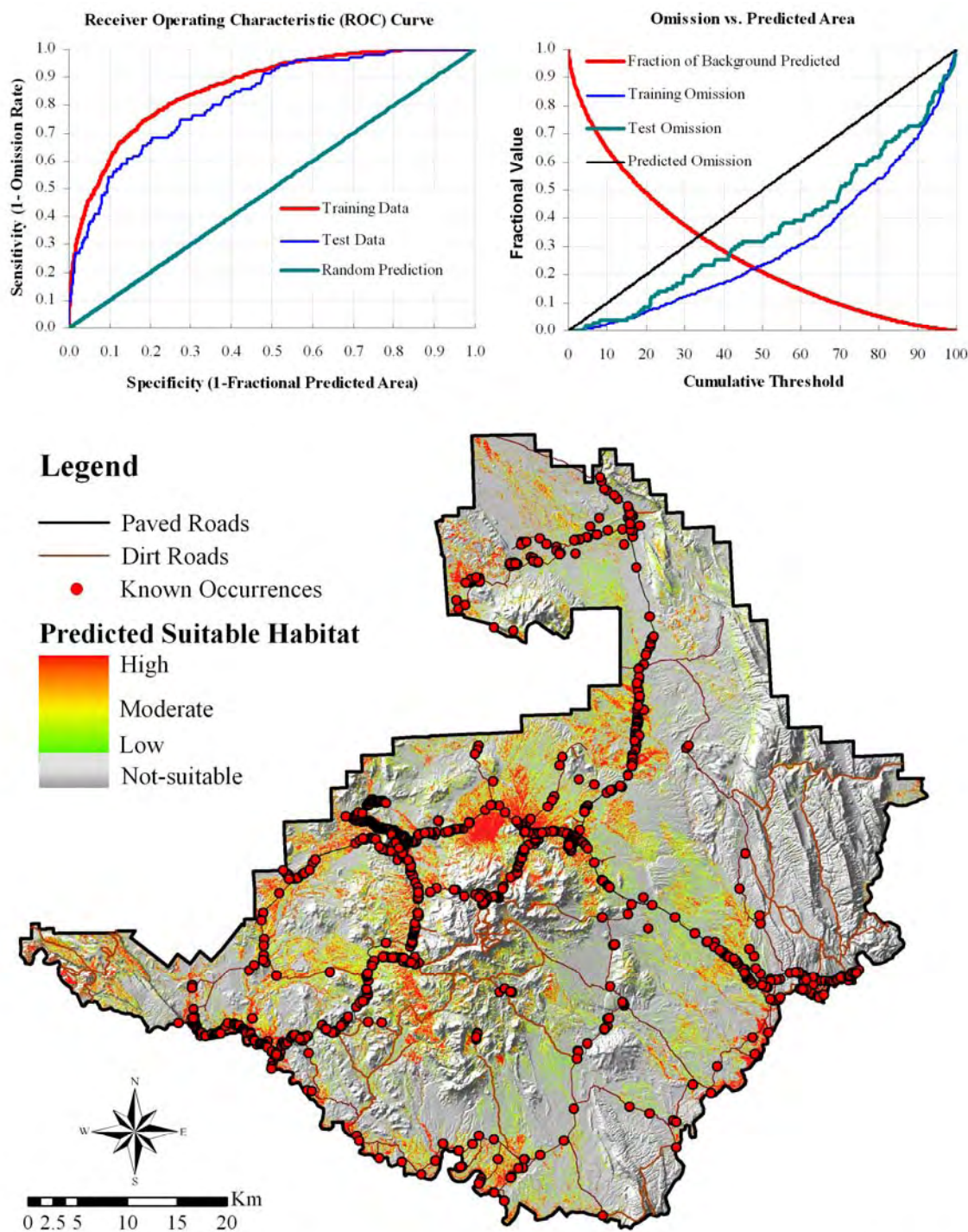


Figure 3-11. Potential distribution of saltcedar in BIBE modeled using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission verses Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).



Invasive Habitats (modeled from all 8 species) in Big Bend National Park

Figure 3-12. Potential distribution of invasive habitats modeled from all eight target species in BIBE using maximum entropy of presence locations. ROC curve visual interpretation: good predictive models have curves that rise steeply at the origin, and then level off at a sensitivity value near 1.0. Omission versus Predicted Area interpretation: an optimal model would have an omission curve close to the predicted omission (diagonal line).

High Discriminate Models

Horehound and giant reed models yielded high test AUC values, narrow 95% confidence intervals (0.99 ± 0.02), high model gain values (3.4) and low errors of omission ($\approx 0\%$; $P < 0.01$) (Table 3-5, Fig. 3-13). Model AUC and gain values indicate that the likelihood that a random positive horehound or giant reed occurrence and a random negative location were accurately predicted was 99%, and the average sample likelihood was $\exp(3.4) \approx 30$ times higher than that of a random background pixel. However, model assessment values for these species may not be robust due to the low number of test samples (Table 3-4). Additional samples should be collected to provide additional insight to model performance. Nevertheless, our models indicate that useful predictive models may be created with < 50 occurrence locations.

Likewise, AUC estimates and 95% confidence intervals for the Johnson grass and saltcedar models yielded high AUC values ($\approx 0.92 \pm 0.02$) (Fig. 3-13). These two plant species had moderate errors of omission ($\approx 15\%$; $P < 0.01$)

(Table 3-5). In BIBE, these two plants can be found in large patch sizes that are readily distinguishable from the surrounding landscape. Giant reed and saltcedar often commingle along the Rio Grande. These attributes likely helped in the successful modeling of Johnson grass, saltcedar, and giant reed.

Moderate Discriminate Models

Model test AUC values indicated moderate discrimination for buffelgrass (0.88 ± 0.01), Bermuda grass (0.85 ± 0.05), and Lehmann's lovegrass (0.81 ± 0.02) (Fig. 3-13). Errors of omission were high for Bermuda grass (27%) and Lehmann's lovegrass (28%), but were moderate from buffelgrass (19%) (Table 3-5). In BIBE, these plants are fairly widespread and can be found in most soil types (Cox et al. 1988, Carey 1995). These plants readily colonize disturbed areas and establish easily in roadside run-off areas or developed areas.

We observed similar levels of discrimination when combining all plant locations into one predictive model (Fig. 3-12). The combined

Table 3-5. Performance measurements for invasive species predicted habitat models in Big Bend National Park.

Common Name	Threshold Independent				Threshold Dependent			
	Training Samples		Test Samples		Equal Test Sensitivity and Specificity			
	AUC ¹	Gain ²	AUC ¹	SD	Cumulative Threshold	Training	Test	P-value
						Omission Rate	Omission Rate	
Giant Reed	0.987	3.37	0.991	0.006	35.23	0.11	0.00	1.28E-05
Bermuda grass	0.954	1.47	0.854	0.072	31.08	0.05	0.27	2.20E-03
Lehmann's lovegrass	0.934	1.10	0.811	0.052	20.68	0.07	0.28	2.25E-06
Horehound	0.998	3.41	0.992	0.006	46.74	0.00	0.00	3.28E-04
Buffelgrass	0.935	1.07	0.888	0.022	37.15	0.11	0.19	1.26E-34
Russian thistle	0.935	1.73	0.773	0.076	17.80	0.06	0.33	0.25
Johnson grass	0.965	1.67	0.920	0.029	23.09	0.02	0.14	6.52E-08
Saltcedar	0.957	1.41	0.929	0.035	41.30	0.09	0.14	3.70E-14
All 8 plants	0.862	0.49	0.822	0.019	41.54	0.18	0.27	7.99E-27

¹ AUC = Area under the curve derived from receiver operating characteristic (ROC) curves for each plant.

² Regularized model gain derived from training samples.

target plant predictive model resulted in a test AUC value of 0.82 ± 0.004 , and a 27% error of omission (Table 3-5, Fig. 3-13).

Combining data from all target plants likely created a habitat generalist dataset which may have reduced our ability to differentiate suitable habitats (resulting in a moderate AUC value) or detect vegetation phenology differences using the maximum entropy analytical approach. However, the larger sample size resulted in greater model precision (low AUC standard deviation), meaning that repeated modeling effort would likely result in a similar AUC value.

Additional research is needed to investigate the effects of combining species attributes (spectral signatures in our case) using the maximum entropy presence-only modeling approach. The ability to discriminate buffelgrass, Bermuda grass, Lehmann's lovegrass, and all species combined models may be improved by including more specific habitat association variables in the modeling endeavor (e.g., disturbed areas) or analytical

approaches that utilize plant presence and absence data.

Low Discriminate Model

Our Russian thistle model yielded a moderate AUC value (0.77) with a 95% confidence interval ranging from 0.58-0.96 (Fig. 3-13), effectively reducing the interpretation of the model. This model also resulted in a high error of omission (33%), and was the only model with a statistically insignificant Wilcoxon signed-rank p-value (Table 3-5). Several factors likely attributed to this poor discriminate ability of this model. Russian thistle is an annual that can be found on most soil types in disturbed areas, fields, overgrazed pastures, roadsides, and desert communities (Young and Evans 1972, Young 1991). This plant can be found as large clumps or individual plants. These factors, compounded by the low number of occurrences used in the modeling process, likely reduced our ability to discriminate Russian thistle from surrounding vegetation

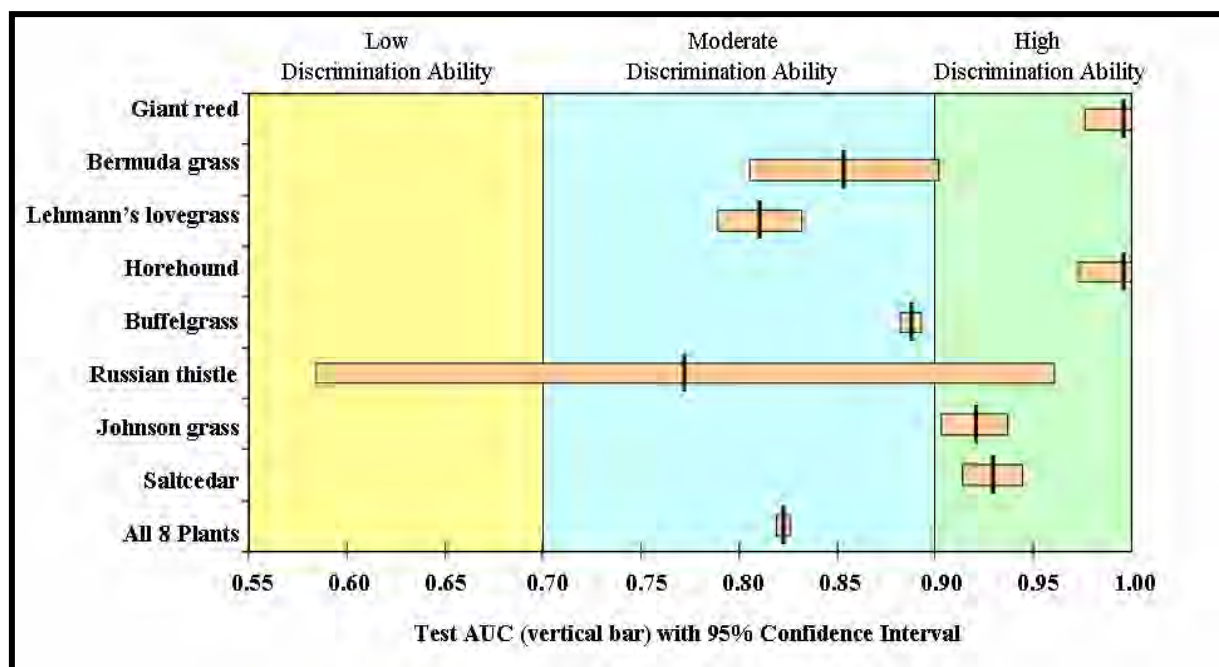


Figure 3-13. Invasive plant model discrimination classes (Pearce and Ferrier 2000, Manel et al. 2001) based on test AUC and 95% confidence intervals.

communities. This model may also be improved by including additional samples in the modeling endeavor or by including more specific habitat association variables (e.g., frequently disturbed areas), or using presence/absence modeling approaches.

Model Variable Selection

A model that incorporates the largest number of variables possible is considered to have the most flexibility in fitting the observed data (White 2001). However, a large number of input variables increases the amount of variability and decreases the precision of the model, effectively decreasing our ability of accurately predicting suitable habitat. Conversely, models with few parameters typically are more precise because all the data are being used to estimate the parameters. But, these models have more bias associated with them. In other words, as the number of variables increase, bias is reduced but precision is lost (Franklin et al. 2001). Ideally, we would like to have the most parsimonious model possible, e.g., a model that provides a balance between the extremes of having too few parameters (under-fitting) and models that have too many parameter (over-fitting) (Burnham and Anderson 1992).

Our predictive habitat models used three seasons of spectral data across five years (Fall 1999, Summer and Fall 2000, Spring and Fall

2001, Spring 2002, and Spring 2003), or 42 possible spectral classes. We used analyses provided in MaxEnt to guide us toward parsimonious models. We removed variables that appeared to provide little contribution to model predictions. Even with these procedures, the number of variables retained in each model was relatively large and varied between 15 and 35 spectral classes (Table 3-4). All seasons and all years were used in the eight target species models (Appendix D). However, Spring 2001 (Band 5), Spring 2002, (Band 5), Fall 2001 (Band 4), and Summer 2000 (Band 4 and 6) had the greatest contribution to all the species predictive models (based on the gain of each variable in the models). Likewise, elevation was an important model input variable for all species except Johnson grass and Russian thistle. Conversely, spectral characteristics of the Fall 2000 and Spring 2003 dataset did not contribute much to the predictive models.

Our results indicate a moderate relationship between the number of variables selected and sample size and the regularized training gain derived from training samples (Fig. 3-14). Essentially, larger sample sizes (number of plant occurrences) yielded greater number of input variables retained in the predictive models. Further, the greater number of variables in the models reduced the likelihood that the model to differentiate plant

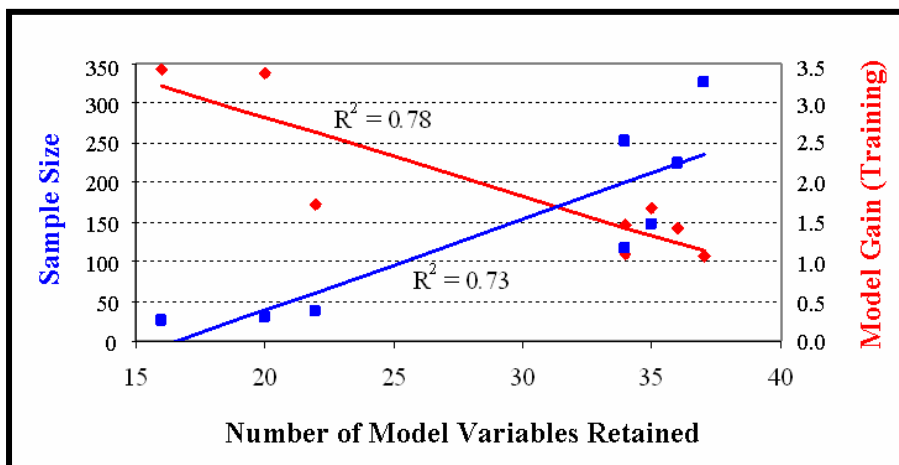


Figure 3-14 (Left). Relationship between sample size, gain derived from training samples and the number of model variables retained in the eight invasive plant predictive habitat models.

occurrences from random background pixels (regularized training gain) (Fig. 3-14).

These results likely reflect MaxEnt attempt to estimate the plant's broadest probability distribution given an increase in variability associated with greater numbers of sample locations and the increased variability associated with greater number of model variables. These results also support the need to seek parsimonious models. Phillips et al. (2004) noted that MaxEnt models with a larger number of variables tend to overfit small training sets, but they give more accurate predictions for large training sets. Although our smaller training sets yielded smaller number of model variables, our giant reed, horehound, and Russian thistle models may still be overfit due to their low number of occurrence locations. Additional field sampling in BIBE would likely produce additional locations of these plants that might alleviate concerns related to overfitting. However, we recommend additional model selection processes including correlation analyses, and evaluating multiple models using Akaike's Information Criterion (AIC) (Burnham and Anderson 1992, 2002). The creation and evaluation of multiple models are often warranted because natural landscapes and associated species habitat relationships are complex (McComb et al. 2002). When multiple models are considered, the model yielding the smallest AIC value can be considered the most parsimonious representation of the data since AIC metrics provide a trade off between the fit of the model and the number of parameters (White 2001).

Amount of Predicted Suitable Habitats

MaxEnt outputs include a continuous surface with values ranging from 0-100. Larger values indicate areas with greater likelihoods of suitable habitat conditions. We anticipate resource managers will initially focus their

attention on areas with high likelihoods of suitable habitats, followed by lower likelihood values. Additionally, threshold values that partition the data between suitable and unsuitable habitat areas may be chosen based on specific management objectives. We used a threshold that provided an equal tradeoff between sensitivity and specificity (Table 3-5). Values greater than the cumulative threshold indicate suitable habitats, while thresholds less than the cumulative threshold indicate areas unsuitable for the target plant. The selection of a threshold allowed us to estimate the amount of suitable habitat for each target plant species in BIBE.

Our results indicate that giant reed and horehound may have the most limited potential distribution in BIBE, with approximately 2% of the park surface modeled as potential habitat (Table 3-6). Modeled potential habitat for giant reed was fairly restricted to the Rio Grande corridor where large stands are well established (Fig. 3-4), which is supported by Hughes and Mickey (1993), and Tracy and DeLoach (1998). Horehound potential habitat is primarily around developed areas near the Chisos Mountains and small patches in the northern part of the park near the Rosillos Mountains (Fig. 3-7).

Approximately 14% of BIBE was modeled suitable for saltcedar (Table 3-6; Fig. 3-11). Large stands of saltcedar occur in high suitable habitats along the Rio Grande, but smaller stands are fairly common in arroyos and springs that have adequate seasonal water for establishment. Saltcedar has been a species of concern for BIBE for many years. Control efforts have been employed in water sources throughout the Park (BIBE 1998).

Potential suitable habitat for the remaining target plants was widespread and encompassed between 20-40% of the Park

Table 3-6. Amount of predicted suitable habitat in BIBE for each target invasive plant species estimated using the 'Equal Test Sensitivity and Specificity' threshold from MaxEnt.

Common Name	Not-suitable Habitat			Suitable Habitat		
	Hectares	Acres	Percent	Hectares	Acres	Percent
Giant Reed	311,553	769,861	97.7	7,326	18,103	2.3
Bermuda grass	233,080	575,951	73.1	85,799	212,013	26.9
Lehmann's lovegrass	237,811	586,762	74.6	81,068	200,322	25.4
Horehound	313,455	774,561	98.3	5,425	13,405	1.7
Buffelgrass	256,477	633,766	80.4	62,402	154,198	19.6
Russian thistle	222,365	549,474	69.7	96,514	238,490	30.3
Johnson grass	191,493	473,188	60.1	127,386	314,776	39.9
Saltcedar	273,987	677,034	85.9	44,892	110,930	14.1
All 8 plants	233,493	576,972	73.2	85,386	210,993	26.8

(Table 3-6). Johnson grass had the largest predicted amount of suitable habitat in BIBE (127,386 ha [314,776 ac]; 40%). Large stands of Johnson grass occur along the North Rosillos Road (Fig. 3-10). Further, Johnson grass can be found in the northern and southern Tornillo Creek drainage.

We estimated approximately 85,799 ha (212,013 ac; 27%) of suitable area for Bermuda grass in BIBE (Table 3-6, Fig. 3-5). Bermuda grass is well established in BIBE especially in developed areas, campgrounds, and along the Rio Grande where it has displaced native vegetation (See Chapter 2). Large patches of suitable habitat occur north of the Chisos Mountains.

Buffelgrass is a potentially increasing problem for BIBE. Approximately 20% of BIBE was predicted to have suitable conditions for this plant (Table 3-6). Large stands of buffelgrass occur in the southern part of the Park, but this grass can be found throughout BIBE (Fig. 3-8). Further, Arriaga et al. (2004) noted that buffelgrass in northern Mexico could invade desert scrub and mesquite woodlands.

Predictive Model Considerations

Favorable spatially explicit predictive models incorporate: 1) invasive species distribution data, 2) population rates, 3) factors influencing the number of propagules, 4) dispersal modes, 5) landscape structure, 6) ecological processes, and 7) statistically explained patterns (Moody and Mack 1988, Higgins et al. 1996, Higgins and Richardson 1999, Wadsworth et al. 2000, Sakai et al. 2001). Unfortunately, the ecology of many invasive plant species is poorly understood or has been inadequately investigated to include all these factors in spatial modeling endeavors.

We used a maximum entropy approach for our modeling endeavors. This approach estimates the broadest target probability distribution which should have minimized errors of omission. From conservation perspective, errors of omission are less acceptable than errors of commission (Shrader-Frechetter and McCoy 1993). For example, if the predictive model concluded an area was suitable, and the species was not present (commission), then the area would be a good candidate to monitor for potential expansion of the species.

Our invasive plant predictive models were created using Landsat 7 ETM+ spectral information derived from georeferenced locations in BIBE. We employed multi-temporal datasets to capture unique vegetation phenology and to help differentiate invasive plant populations or their habitats from the surrounding landscape. Based on model performance, our proof-of-concept approach worked well for giant reed, horehound, Johnson grass, and saltcedar, and moderately well for buffelgrass, Bermuda grass, and Lehmann's lovegrass. The efficacy of using remotely sensed data to model the potential distribution likely was influenced by Landsat 7 ETM+ spatial resolution, invasive patch size, habitat generality, and similar environmental conditions surrounding invasive plant populations. Modeling approaches that employ presence/absence analyses may help to differentiate these communities. Further, integrating other spatial information (e.g., disturbances, and soil moisture regimes), high spatial resolution data, or greater sample sizes may improve model performance.

Conservation Implications

Remotely sensed images have a number of features which make them ideal for predicting invasive species in large landscapes. Remote sensing technologies look beyond the human view in the electromagnetic spectrum, which allows better detection of vegetation. Remote sensors allow for regional analyses that would be cost-prohibitive using ground-based visits. Regional analyses allow for prioritizing ground reconnaissance visits to survey, control, or eradicate potential invasive plant populations. Further, the ease of securing temporal data allows for repeat analyses and change detection.

Our ability to detect invasive plants using remotely sensed data has increased with

improved sensors, computer technology, and classification techniques (Lass et al. 2005). Although integrating remotely sensed data with other spatial datasets enhances our abilities to model invasive plants, detecting small or sparse plant populations is still hampered by spatial and spectral resolution, and our ability to analyze large datasets. The optimal remote sensing data, or combination of data, would have characteristics of hyperspectral sensors and high spatial resolution sensors. While hyperspectral data facilitates detection of individual plants, hyperspectral data has approximately 75 times greater data volume than an equivalent area using Landsat ETM+ (Thenkabail et al. 2004). Likewise, multispectral, high spatial resolution sensors (e.g., IKONOS or QuickBird) also show promise in detecting invasive plants with spatial resolutions < 5 m (16.4 ft). These sensors are also encumbered by large data volumes over large areas. The new challenge is to develop methods which integrate the required spectral resolution with the ideal spatial resolution, and are efficient with the high-dimensional datasets for large area analyses. Remote sensing datasets also come with fiscal and technical expertise considerations. Higher spectral and spatial resolution data are substantially more expensive than multispectral, medium spatial resolution sensors, and require greater technical expertise for image processing.

Since spatial data on invasive plant species absences are often not available for land managers, reliable modeling approaches using presence-only datasets could become powerful tools for land managers. We found MaxEnt to be a user friendly free-ware program, although documentation on the interpretation of the program's output is somewhat limited. Unfortunately, there has been limited use of this procedure in species distribution modeling. Additional research is needed to evaluate the full potential of maximum

entropy and MaxEnt in developing species predictive models. Similar presence-only data modeling procedures such as Genetic Algorithm for Rule-set Prediction (GARP) have been widely used in modeling species distributions (Stockwell and Peters 1999, Stockman et al. 2006). However, Phillips et al. (2006) evaluated MaxEnt outputs versus GARP outputs, and concluded that MaxEnt models outperformed GARP models.

National Parks need predictive models that can help in setting priorities for control of

invasive species and predict the potential of future invasions. Remotely sensed data can aid in the development of spatially explicit predictive habitat models and estimates of distributional vectors and pathways (see Chapter 4). This information provides land managers with early detection tools, a means to evaluate current and future control needs, and a means to prioritize conservation efforts. Early detection methods increase our ability to eradicate invasive plants and reduce cost of control (Rejmanek and Pitcairn 2002).



Chapter 4

Invasive Species Risk Assessment

Introduction

Few communities are impenetrable to invasion by nonnative species, and communities differ in their susceptibility to invasion (Sakai et al. 2001). Repeated introductions increase the chances of establishment (Sakai et al. 2001, Perrings et al. 2002). However, the variability associated with species traits, climatic events, dispersal mechanisms, and landscape characteristics prohibits a set of conditions or traits that can be universally applied to accurately characterize all successful invasions (Alpert et al. 2000, Tackenberg 2003).

Invasive species spread and establishment is partially influenced by population growth, dispersal mode, landscape structure or suitability, and the number and frequency of introduction locations (related to propagule pressure) (Moody and Mack 1988, Higgins and Richardson 1999). Spatial analyses of these factors may provide some understanding of localities to concentrate survey efforts to find new species, or expanding populations of existing species. This understanding is central to early detection protocols.

Risk assessment procedures can assist natural resource managers in prioritizing areas for conservation or monitoring efforts. Thus, risk assessments can be powerful tools for early detection protocols. Ecological risk assessments are processes that evaluate the likelihood that adverse ecological effects may occur as the result of exposure to one or more stressors (EPA 1998). Principle elements of risk assessments include 1) defining the problem, 2) analyzing exposure and effects of exposure, and 3) risk characterization (EPA 1998).

In terms of invasive plant species, areas at risk can be estimated by incorporating a broad array of information describing factors that may influence their distribution (Allen et al. 2006). Exposure analysis for invasive species may incorporate the quantity, timing, frequency, duration, vectors, pathways, and susceptibility of populations exposed (Anderson et al. 2004a). Invasive species effects analyses involve estimating the probability and severity of economic, health, or environmental consequences of an exposure to invasive species (see Chapter 1).

Risk characterization integrates information from the problem statement, exposure and effects analyses into an overall conclusion about risk that is informative for conservation efforts (Anderson et al. 2004a). Risk characterizations may range from qualitative judgments to quantitative probabilities (EPA 1998). Although quantitative risk assessments are desirable, quantitation of risks is often not possible. In these instances, qualitative conclusions (and associated uncertainties) are more desirable than not conducting risk assessments (EPA 1998).

There are several approaches to modeling risk of invasions. For example, neutral landscape models, which evaluate flows through spatially heterogeneous landscapes, were used to assess the risk of invasions in fragmented landscapes (With 2004). Landis (2004) described a relative risk model that incorporated a system of numerical ranks and weighting of factors that may influence the distribution of invasive species. The Landis (2004) approach used spatial relationships of the locations of species introductions, migration paths (potential vectors and

pathways), and the habitat structure or suitability. Vectors refer to the mechanism of plant introduction, while pathways refer to the route taken. Examples of vectors include wind, water, and animals (Sakai et al. 2001). Examples of pathways include roads, trails, and waterways.

In many parks and other natural areas, roads and waterways are perhaps the pathway of most concern. These pathways enhance species invasions by acting as dispersal corridors, providing suitable habitat, and containing reservoirs of propagules (Parendes and Jones 2000). Disturbance along roads by vehicle traffic and maintenance activity (e.g., road grading, ditch clearing, and trimming of overhanging vegetation) creates suitable habitats for species introductions and provides a source of repeated introductions (Gelbard and Belnap 2003). Waterway disturbances occur from floods and associated transport of sediment.

Many authors have analytically modeled potential vectors, especially seed dispersal by wind. Schurr et al. (2005) created a mechanistic model for secondary seed dispersal by wind (the wind-driven movement of seeds along the ground surface). The authors found a relationship between seed size and dispersal, but noted that the model tended to underestimate dispersal rates. Tackenberg (2003) also modeled seed dispersal by wind and found that long-distance dispersal was primarily influenced by weather conditions that yielded thermal turbulence and convective updrafts. Tackenberg (2003) noted that the inclusion of topography in estimating dispersal rates is important, even in landscapes that exhibit only small differences in elevation and slight slopes. In addition, Campbell et al. (2002) simulated landscape scale invasions of plants that use rivers to transport propagules. Higgins et al. (2001) created a spatially explicit simulation model based on plant

recruitment, dispersal, mortality, and disturbance for two plant species which invaded South Africa's shrub lands.

Remote sensing and GIS datasets can provide the necessary spatial datasets to model potential habitats, vectors, and pathways that allows for landscape scale risk assessments. GIS datasets for invasive species pathways are readily available. Many parks and natural areas already have road, trail, and waterway GIS layers. Further, road and hydrologic layers are available from a variety of GIS data clearinghouses, federal agencies, and private companies. Digital line graphs (DLGs) (digital vector data derived from USGS maps), Digital Raster Graphic (DRGs) (scanned digital images of USGS topographic quadrangles), and National Hydrography Dataset (NHD) can be downloaded from the USGS website, <http://edc.usgs.gov>. Spatial datasets on potential vectors are not readily available, and would likely have to be created specifically for the target species and the area of interest.

Methods

Our risk analysis approach was similar to Landis (2004) in that we incorporated a system of numerical ranks and weighting of spatial factors that may influence the distribution of our target invasive plant species. We use data layers of vectors and pathways identified in Chapter 2, locations of known invasive species populations (to represent species introduction points), and predictions of suitable habitat for each species for our risk assessments (Chapter 3).

We discuss the effects of invasive species presence in natural areas in Chapter 1. In terms of our risk assessments, we assume landscapes with invasive species presence would have negative economic, health, and environmental consequences regardless of the

plant species. Thus, we did not conduct a more detailed effects analysis or weight our plant species differently.

Transmission vectors and pathways were very similar for our target invasive plant species (Table 4-1; see Chapter 2). Propagule dispersal by wind, water, and human transportation were the primary vectors identified. Likewise, waterways, roads,

campgrounds, and other human development areas tended to be the common pathways for our target invasive plant species. As such, we created spatial data layers to represent these common vectors and associated pathways for BIBE (Table 4-1). We used six spatial variables for our risk analyses: 1) occurrence locations, 2) habitat suitability, 3) disturbance, 4) hydrologic features, 5) wind direction, and 6) fire history.

Table 4-1. Risk analysis datasets used to estimate vectors and pathways for target invasive plants in Big Bend National Park.

Vectors	Pathways	Risk Analysis Dataset
Seed or vegetative dispersal by wind	Downwind from known populations (land or water)	Wind Direction Directed polygon extending downwind from known locations
Seed or vegetative dispersal by water	Rivers and other waterways	Hydrologic Features Weighted areas of rivers and waterways
Seed transportation may occur from commercial and garden uses	Roads and human development	Disturbance Weighted distance to roads and dwellings
Seed and/or vegetative reproduction	Lateral movement from existing populations	Occurrence Locations Weighted distance to known plant locations
Potentially transported by vehicles or other anthropomorphic use	Roads, trails, campgrounds, and human development	Disturbance Weighted distance to roads, campgrounds, and dwellings
Potential seed contaminant in feed hay, thus potentially dispersed livestock or animals	Campgrounds and stock trails	Disturbance Weighted distance to campground and trails
	Colonizes disturbed areas, (e.g., grazing, flooding, grading, fire)	Disturbance and Fire History Evidence of historic fire and weighted distance to roads, trails, campgrounds, and dwellings
	Suitable habitats	Habitat Suitability Weighted habitat suitability categories
	Moist soils near roadside runoff areas, springs, seeps, stock tanks, and adjacent to streams and marshes	Hydrologic Features and Disturbance Weighted areas of soil moisture properties and distance from roads

We used three general numeric ranks (250, 150, and 50) to represent each variable's relative importance in the establishment and spread of our target invasive plant species (Table 4-2). These numeric ranks are not supported by quantified data, and thus should be evaluated with empirical data and updated as information becomes available. Likewise, these numeric ranks may not be appropriate for other study areas or species.

Numeric Ranks of Risk Variables

High Ranked Variables

The probability of invasion success increases with greater propagule pressure or number of introduction attempts and the presence of suitable habitat (Rejmanek 2000). Thus, we considered areas with existing invasive plants and areas predicted to host suitable habitats as the two main characteristics that influence the establishment and spread of our target invasive plant species in BIBE. Inherently, existing plant populations provide a steady propagule source for neighboring areas. Further, areas that host favorable environmental conditions for target plants are more likely to have successful invasions (Rejmanek 2000). Thus, we gave both variables a high numeric rank (250) which represents approximately 28% of the total weighted scale (Table 4-2). When these two variables spatially co-exist, their combined rank (added together) would account for greater than half the total possible score in the risk models.

Moderate Ranked Variables

Disturbance and hydrologic features are also important considerations for invasion success. The spread of many invasive plants is heavily influenced by human activity and associated disturbances (Hobbs and Huenneke 1992, Rejmanek 2000). Altered natural disturbance cycles reduce native plant communities and favor introduced species that are pre-adapted to the new conditions. As such, areas that

Table 4-2. Numeric ranks given to risk assessment variables to represent each variable's importance in the establishment and spread of our target invasive plant species in Big Bend National Park.

Risk Variable	Numeric Rank	Rank Importance
Occurrence Location	250	High
Habitat Suitability	250	High
Disturbance	150	Moderate
Hydrologic Features	150	Moderate
Wind Direction	50	Low
Fire History	50	Low

concentrate human activity (e.g., roads, trails, campgrounds, and developed areas) increase the likelihood of successful plant invasions. Likewise, hydrological features can greatly influence the likelihood of successful plant invasions, especially in desert environments (Brooks and Pyke 2001). Water can serve as a vector (e.g., seed dispersal by flowing water), pathway (e.g., propagule dispersal down river), or an attribute of habitat suitability (e.g., soil moisture availability). Therefore, we gave the disturbance and hydrologic variables a moderate numeric rank of 150, which represents approximately 17% of the total weighted scale (Table 4-2).

Low Ranked Variables

The remaining two variables, wind direction and fire history, were considered modifiers or enhancers to invasiveness for the other risk assessment variables. For example, areas downwind of existing invasive plant populations would be at higher risk than areas upwind due to increase propagule pressure.

Likewise, many invasive plants prosper in post-fire landscapes, partly due to temporary increases in the availability of soil nutrients and the elimination of competition of limited resources (Brooks and Pyke 2001). The

increase in invasive grasses post-fire dominance may further promote recurrent fire that continues to convert native to nonnative communities (Humphrey 1974, Brooks and Pyke 2001). Because of these enhancer qualities, we provided these two variables with a numeric rank of 50, which represents approximately 6% of the total weighted scale (Table 4-2).

Numeric Weights Within Risk Variables

Each of the six variables described above were given numeric weights based on the variable characteristic (e.g., degree of habitat suitability) or distance interval from a given feature. We expanded (buffered) many of the spatial variables used in the risk analyses to represent the potential influence of the variable on the landscape. For example, spatial data layers of roads would be spatially limited to the width of the road. Yet, the influence of a road to the invasion process is greater than the width of the road (Gelbard and Belnap 2003).

Determining an appropriate influence distance for each variable without quantified data is difficult. In BIBE, the actual distance influenced by a variable was not known and designing a study to quantify influence distances was beyond the scope of our project. Therefore, we created buffer distances that seemed logical for our plant species in our study area. These buffer distances should be evaluated and updated as information becomes available. Likewise, these buffer distances may not be appropriate for other study areas or species.

We chose our buffer distances based on several considerations. Our first consideration was based on information provided in the literature regarding area of influences around features, e.g., seed dispersal distance or human activity around different roads or campgrounds. We also considered the

influence of the scale of our datasets. For example, since we used 30 m (98.4 ft) raster datasets, creating a 1 m (3.3 ft) buffer around a feature would be meaningless and inconsistent with the scale of the overall project. Further, our buffer distances consider the precision of our datasets. For example, GPS locations may have between 3-90 m (9.8-295.3 ft) spatial error rate, depending on the quality of the equipment and the availability of satellites, among other influential factors. The use of appropriate buffers ensures that our inferences represent locations of the features within a reasonable degree of data precision.

We provided numeric weights to each variable characteristic or a distance interval (Table 4-3). These numeric weights represent a relative importance within each variable to the establishment and spread of our target invasive plant species. These weights are not supported by quantified data, and thus should be updated as information about each variable characteristic becomes available. Likewise, these numeric weights may not be appropriate for other study areas or species.

Occurrence Locations

We created five circular 50 m (164 ft) intervals around all target invasive plant species coordinates (250 m; 820 ft radius to the outer buffer) through a simple buffering operation in ArcGIS 9.2. Animals or people may inadvertently transport propagules from these populations in any direction. The actual distance that propagules may travel is unknown, but assumed to be variable by species, locations, habitat, etc. Therefore these distance intervals should be considered conservative estimates. Each interval was given a numeric weight between 50 and 250 (Table 4-3). Intervals closer to the plant coordinate were provided higher weights due to increased likelihood of lateral expansion of the established population through seed dispersal or vegetative reproduction.

Table 4-3. Numeric weights given to variable characteristics used to estimate areas at risk of invasion in Big Bend National Park.

Risk Variable	Weight	Data Characteristic
Occurrence Location		
	250	< 50 m of plant occurrence
	200	50-100 m of plant occurrence
	150	100-150 m of plant occurrence
	100	150-200 m of plant occurrence
	50	200-250 m of plant occurrence
	1	> 250 m of plant occurrence
Habitat Suitability		
	250	High suitable habitat
	200	Moderate suitable habitat
	150	Low suitable habitat
	1	Not-suitable habitat
Disturbance (Paved roads, dirt roads, hiking trails, campgrounds, and human development)		
	150	< 50 m of paved road, campground or developed area
	100	50-100 m of road, campground or developed area < 30 m of a dirt road
	50	100-150 m of paved road, campground or developed area < 30 m of a trail
	1	> 150 m of paved road, campground or developed area > 30 m of a dirt road > 30 m of a trail
Hydrologic Features (Rio Grande, springs, ephemeral streams, and soil moisture properties)		
	150	< 100 m of the Rio Grande < 50 m of springs
	100	100-200 m of the Rio Grande 50 - 100 m of springs < 100 m of an ephemeral stream Soil moisture properties that enhances plant establishment
	50	200-300 m of the Rio Grande 100-150 m of springs
	1	> 300 m of the Rio Grande > 150 m of springs > 100 m of an ephemeral stream Not sufficient soil properties to enhance plant establishment
Wind Direction		
	50	< 250 m down wind of plant occurrence
	1	> 250 m downwind or upwind of plant occurrence
Fire History		
	50	Historic fire
	1	No evidence of historic fire

Habitat Suitability

Habitat compatibility has also been recognized as an important component to successful invasions (Rejmanek 2000). As such, we used the predicted habitat suitability models described in Chapter 3 to represent areas of suitable habitat for each species. To differentiate suitable from non-suitable habitat, we selected a threshold that provided an equal tradeoff between the test data sensitivity and specificity (See Chapter 3). Values below this threshold were considered unsuitable habitat and values above the threshold were considered to have adequate habitat for the targeted species. We then divided the distribution of suitable habitat values into equal quantile proportions to identify classes of low, moderate, and high suitable habitat. These classes were provided numeric weights between 150 and 250 (Table 4-3), with higher weights representing higher suitability and thus more likely to host successful invasions.

Disturbance

We created multiple buffers around disturbance factors to better spatially represent the influence these factors have in successful invasions (Table 4-3). We considered paved roads, dirt roads, campsites, hiking trails, and park development areas (e.g., Panther junction visitor center and residential area, Rio Grande Village and day use areas, Basin Center and residential areas, water treatment facilities, and garbage dump) as disturbed areas. Spatial data layers representing these disturbance features were provided by BIBE.

Communities with high levels of disturbance or high levels of human activity have higher risk of invasion (Gelbard and Belnap 2003, Rejmanek 2000). Roads are a major pathway for the spread of invasive plants, and provide an example of both high disturbance and high levels of human activity. Vehicles, roads, and roadside maintenance operations cause

disturbances that create habitats for the establishment of invasive plants (Schmidt 1989, Greenberg et al. 1997, Thrombulak and Frissel 2000, Gelbard and Belnap 2003). Plant communities adjacent to improved roads, (e.g., paved or gravel) typically receive high vehicle traffic and have higher likelihood of being invaded than those adjacent to infrequently used primitive roads (Parendes and Jones 2000). Gelbard and Belnap (2003) found a decline in the presence of exotic species with greater distance from roads. Campgrounds and parking lots serve as a central location for human exploration of the surrounding areas.

As such, we provided higher numeric weights to disturbances and distance intervals that had higher levels of disturbance or human activity. For example, we created three intervals around paved roads, campgrounds and human development areas (Table 4-3). Areas within 50 m (164 ft) of a paved road, campground or development area were given a numeric weight of 150. Areas between 50-100 m (164-328 ft) of these features were provided a numeric weight of 100, and areas between 100-150 m (328-492 ft) of these features were provided a numeric weight of 50. Dirt roads and hiking trails receive smaller amounts of public use and thus have less influence on landscapes. We created a 30 m (98.4 ft) buffer around dirt roads and hiking trails. Dirt roads were provided a numeric weight of 100, while hiking trails were provided a numeric weight of 50 (Table 4-3).

Hydrologic Features

Hydrologic features represent vectors (e.g., water movement), pathways (e.g., rivers), suitable habitats (e.g., soil moisture availability), and sources of potential disturbance (human activities). For example, the Rio Grande provides the only perennial flowing water source in BIBE. Flooding events dislodge and transport propagules

downstream to available moist habitats. Further, there are numerous campgrounds (developed and undeveloped) along the Rio Grande which receive large amounts of public use, and thus associated disturbances. The Rio Grande also serves as an available water source for a variety of wildlife species, which in turn may distribute propagules to uplands or other water sources.

We used the National Hydrology Dataset (NHD) to represent proximity to the Rio Grande, springs, and ephemeral streams in BIBE. We created three 100 m (328 ft) buffers around the Rio Grande and provided numeric weights between 50 and 150 that reflect an increased likelihood of the landscape to support hydrologic vectors, pathways, habitat suitability, and human recreational use (Table 4-3). BIBE has numerous springs throughout the Park that serve as a source of periodical moisture for establishment of plants as well as available water sources for wildlife. They also receive occasional recreational use by the public. As such, invasive plant propagules may be transported by humans or wildlife to these isolated springs, where they can establish and flourish. Thus, we created three 100 m (328 ft) buffers around springs and provided numeric weights between 50 and 150 that reflect an increased likelihood of the landscape to support the spread and establishment of invasive plants (Table 4-3).

Ephemeral streams exist throughout BIBE. These drainages carry water only immediately after periods of rainfall, and thus can provide an important vector and pathway for propagule movement as well as adequate soil moisture regimes for plant establishment. As such, we gave areas within 30 m (98.4 ft) of ephemeral streams a numeric weight of 100 (Table 4-3).

Soil properties such as moisture capacity, may enhance the likelihood of plant establishment,

especially in desert environments. Soil moisture capacity varies among soil types and textures in BIBE. We used soil data layers provided by BIBE and Soil Survey Geographic Database (SSURGO) to estimate areas with soil moisture regimes that may enhance the likelihood of invasive plant establishment. Soil attributes (texture, depth, water capacity, and permeability) described by Cockran and Rives (1985) were linked to the spatial soil data layers. Soils with moderate or high available water capacity were considered to have properties that would enhance the likelihood of plant establishment (Appendix E). These areas were provided a numeric weight of 100 (Table 4-3).

Wind Direction

Although seed dispersal by wind is affected by seed traits, wind conditions, and ground movement barriers, the movement of propagules by wind has been recognized as an important dispersal mechanism for plants (Greene and Johnson 1989, Tackenberg 2003, Schurr et al. 2005). The actual distance dispersal of invasive plant seeds in BIBE is unknown, and it is difficult to generalize because plant propagules released under different wind or landscape conditions yield different dispersal distances (Tackenberg 2003). For example, Russian thistle has been noted to disperse seeds between 0.06-4.07 km (0.04-2.5 mi) in six weeks depending on climate (snow or frozen) and wind condition (Stallings et al. 1995). Johnson grass seed has been recorded to be carried up to 1.0 km (0.62 mi) from parent plants from uplifts during thunderstorms (Ghersa et al. 1993).

Wind velocity in BIBE varies, but the prevailing wind direction is from the northwest to the southeast (Mercer 1999). We modeled potential short-distance wind dispersal by creating an area 250 m downwind of known invasive plant localities. We spatially shifted all invasive plant locations

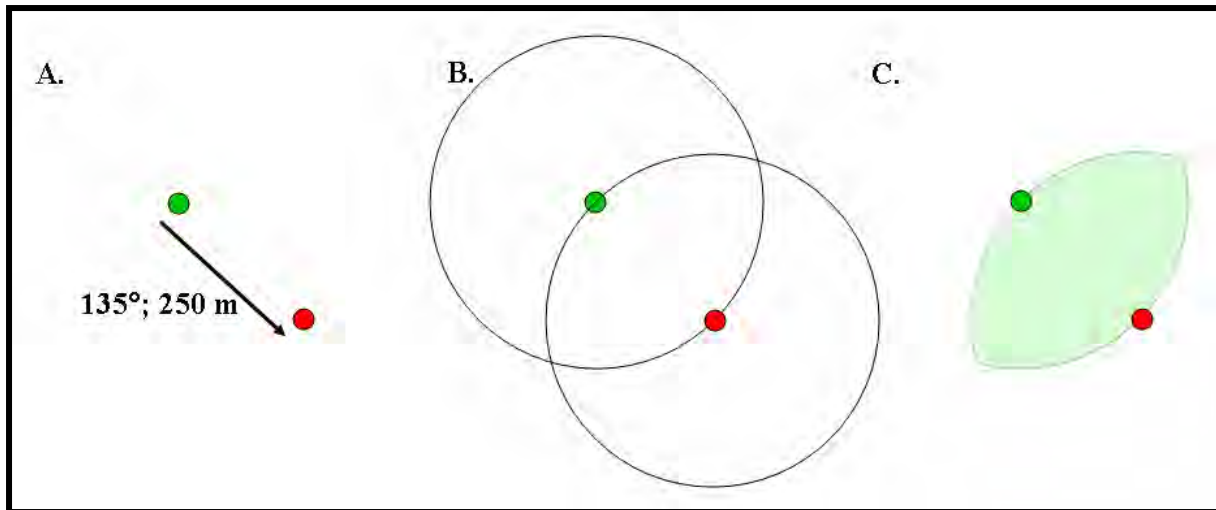


Figure 4-1. Diagram of how wind direction polygons were created: A) occurrence points (green) were shifted 250 m downwind (red), B) points were buffered 250 m, C) overlap area of polygons were considered an area where invasive species establishment is enhanced by short distance wind movement of propagules.

250 m (820 ft), 135° in ArcGIS 9.2 (Fig. 4-1). We then applied a 250 m (820 ft) buffer to the original and shifted locations. We considered the overlap area of the two resulting buffers an area where invasive species establishment is enhanced because of greater likelihood of propagule dispersal by wind. This area resulted in a semi-circle with its apex starting at the location of the target species and extending in an arc outward in the prevailing wind direction (Fig. 4-1). The area within the semi-circle was given a numeric weight of 50.

Fire History

Since many of our invasive plants prosper on post-fire landscapes (especially the invasive grasses), we obtained spatial data on fire history from BIBE. This data layer contained fire history polygons of fires (suppression, prescribed, and fire use) that were over 4 ha (10 ac) from 1948 through March 2006. This data was maintained at the Park level, and polygons were obtained from Mylar overlays, paper USGS maps, and detailed drawings associated with original fire reports. We

provided the fire history polygons a numeric weight of 50.

Building Risk Surfaces

The numerical ranks and weighting allowed us to combine multiple stressors, habitats, and invasive species occurrences into a spatial model that represent areas most at risk of invasion by nonnative species in BIBE. To create the risk surface, we used an additive Boolean overlay that combined the six risk variables into one risk surface. The highest value possible in the risk surface (result of adding the highest value for each variable together) was 900. We grouped risk scores into classes of low, moderate, and high risk (EPA 1998) based on the descriptions of the combinations of risk variables (Appendix F and G). We considered values > 450 to be areas of high risk to invasion by nonnative species; values from 270-449 were areas of moderate risk to invasion, and values < 270 were considered to be areas of low risk to invasion. The amount of area at risk of invasion was summarized within each risk category.

Results and Discussion

Our assessments were designed to evaluate the risk of landscape parcels in BIBE being further invaded or established by our target invasive plants. Our approach yields semi-quantitative, nonprobabilistic, categorical landscape risk models and is similar to other ecological relative risk approaches (Anderson et al. 2004b, Landis 2004). The ranks and weights provided to our risk variables were not based on quantified data, since this type of data is lacking. Therefore, the relative rank of each variable, or variable characteristic, is somewhat arbitrary. Further, it is possible that some variables have multiplicative effects to landscapes versus additive effects. Thus, further investigations into the potential biases associated with our categorical thresholds are warranted.

Categories of Risk

There were 1,536 unique combinations of the six risk assessment variables. We used values > 450 to represent areas that have the highest risk of invasive plant incursion. Approximately 974 (64%) of the possible combinations had values > 450, and an additional 402 (26%) of the possible combinations were categorized as moderate risks (risk values from 270-449). Areas with 50 m (164 ft) of a known occurrence within suitable habitats were typically considered at high risk. In non-suitable habitats the presence of a disturbance or hydrological feature was sufficient to consider the area high risk when it was within 50 m (164 ft) of a known invasive plant occurrence. However, as the distance from known plant occurrences increase, the amount of disturbance or hydrological features needs to increase for the area to be considered at high risk. For example, non-suitable habitats that were 150 m (492 ft) from a known plant occurrence location could still be considered at high risk if there was a moderate degree of disturbance

and the area had hydrological features that would enhance the establishments of invasive plants. Non-suitable habitats that occur between 150-250 m (492-820 ft) from a known plant occurrence locations would require at least a moderate level of disturbance or hydrological features to be considered at high risk. Likewise, areas > 250 m (820 ft) from a known plant occurrence need to have at least low suitable habitat and at least a moderate level of disturbance or hydrological features to be considered at high risk. Appendix F provides all the possible combinations of model input variables that may constitute high risk areas.

Because high risk areas are in close proximity to known occurrences, most high risk areas may already be occupied by invasive plants. As such, areas of moderate risk of invasion are important areas to consider for conservation efforts, and may be focal points for early detection of new invasions. Moderate risk areas typically had at least moderate to low suitable habitat with some degree of disturbance or the presence of hydrological features that enhance the establishment of invasive plants. As the distance from known occurrences increase, the degree of disturbance or hydrological features needs to increase for the area to be considered at moderate risk. For example, low or moderate suitable areas that are > 250 m (820 ft) from a known location may be considered at moderate risk if they have a moderate level of disturbance or hydrological features. Appendix G provides all the possible combinations of model input variables that may constitute moderate risk areas.

Conversely, areas at low risk (risk values < 270) to invasive plant incursion were typically in non-suitable habitats that were > 150 m (492 ft) from known plant occurrences. These areas were either not enhanced, or not greatly enhanced, by disturbance, hydrological

features, or by wind and fire. Given the description of these values, we are confident that our selection of thresholds used to define risk categories was appropriate, and did not greatly bias the interpretation of our results.

Areas at Risk of Invasion

Areas at risk of invasive plant incursion occurred throughout the Park (Fig 4-2 to 4-10). Areas at risk included Panther Junction west to Croton Springs, Chisos Basin, Tornillo Flat near U.S.385, Tornillo Creek near State Highway 118, Rio Grande Village, Boquillas Canyon, drainage between Dominguez Spring and the Rio Grande, riparian habitats from Cottonwood Campground to Santa Elena Canyon, East Corazones Draw, and paved roadways, especially Chisos Basin Road, Rio Grand Village Road, Boquillas Canyon Road, State Highway 118, Ross Maxwell Scenic Drive, and Persimmon Gap Road (U.S. 385).

The amount of BIBE at high risk to invasion by at least one of our invasive plant species varied from approximately 188 ha (465 ac) to 1,957 ha (4,836 ac) (Table 4-4). Assessments using combining all eight target species into one dataset indicated approximately 3,976 ha (9,825 ac) are at high risk of invasion and 25,027 ha (61,843 ac) are at moderate risk in BIBE. Moderate risk landscapes may develop into high risk landscapes if invasive plant

species are found in close proximity, disturbance regimes enhance habitat suitability, or suitable vectors and pathways are developed linking landscapes. Although the acreage at risk for all plants combined represents < 10% of the entire Park, the area at risk is not equally distributed across all habitat types. For example, habitats that host greater human infrastructures (roads, trails, campgrounds, and developed areas), are at greater risk to incursion by invasive plants.

We calculated the proportion of general habitat types (woodland, riparian, grassland, and shrubland) at moderate and high risk within the extent of a general vegetation community map of BIBE produced by Plumb (1991). Unfortunately, this map does not represent the entire Park, and the spatial resolution appears to under-estimate riparian habitats. However, our results indicate that riparian habitats may be 3-times more likely to become invaded than other general habitat types in BIBE (Fig. 4-11). Riparian habitats (mesic and xeric) in BIBE have already been heavily impacted by saltcedar, giant reed, Bermuda grass, and Johnson grass (BIBE 1998). They are also popular recreational areas that concentrate human activity and disturbance. As such, invasive species management efforts may consider prioritizing efforts in riparian habitats.

Table 4-4. Amount of area at risk to invasion by target invasive plants in Big Bend National Park.

Plant	Amount of Area at Risk to Plant Invasion					
	High Risk		Moderate Risk		Low Risk	
	Hectares	Acres	Hectares	Acres	Hectares	Acres
Giant Reed	465	1,149	3,613	8,928	314,801	777,887
Bermuda Grass	1,674	4,137	24,647	60,904	292,558	722,924
Lehmann's Lovegrass	1,811	4,475	34,153	84,394	282,915	699,095
Horehound	188	465	2,181	5,389	316,510	782,110
Buffelgrass	1,957	4,836	16,380	40,476	300,541	742,650
Russian Thistle	2,381	5,884	52,903	130,726	263,595	651,355
Johnson Grass	1,936	4,784	38,869	96,047	278,074	687,133
Saltcedar	1,114	2,753	13,743	33,960	304,023	751,254
All 8 Invasive Plants	3,976	9,825	25,027	61,843	289,875	716,294

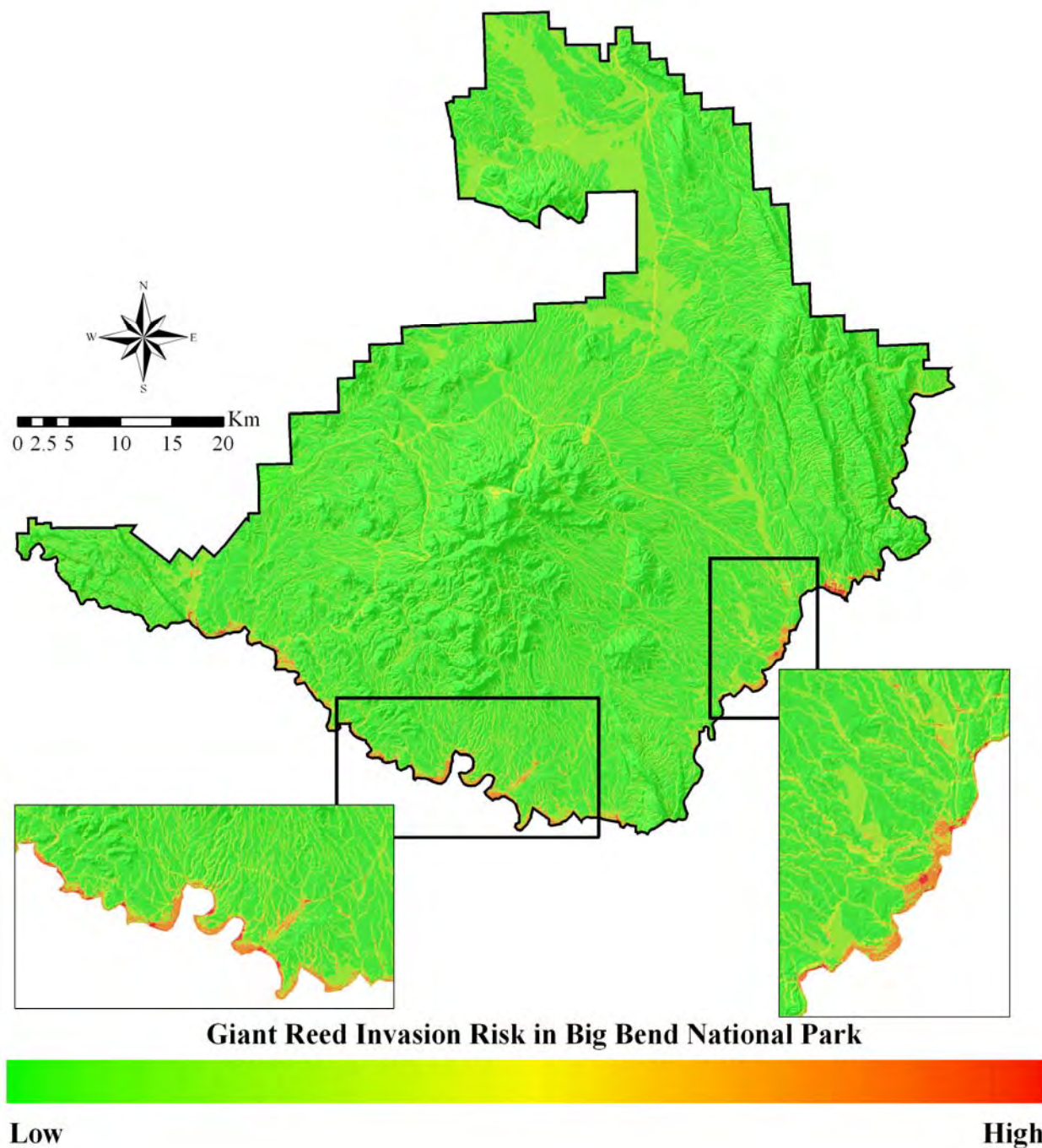


Figure 4-2. Areas at risk of giant reed incursion or spread in Big Bend National Park. The likelihood of giant reed incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian and other habitats may be threatened by giant reed as demonstrated by the moderate or high risk areas (orange to red) in the figure.

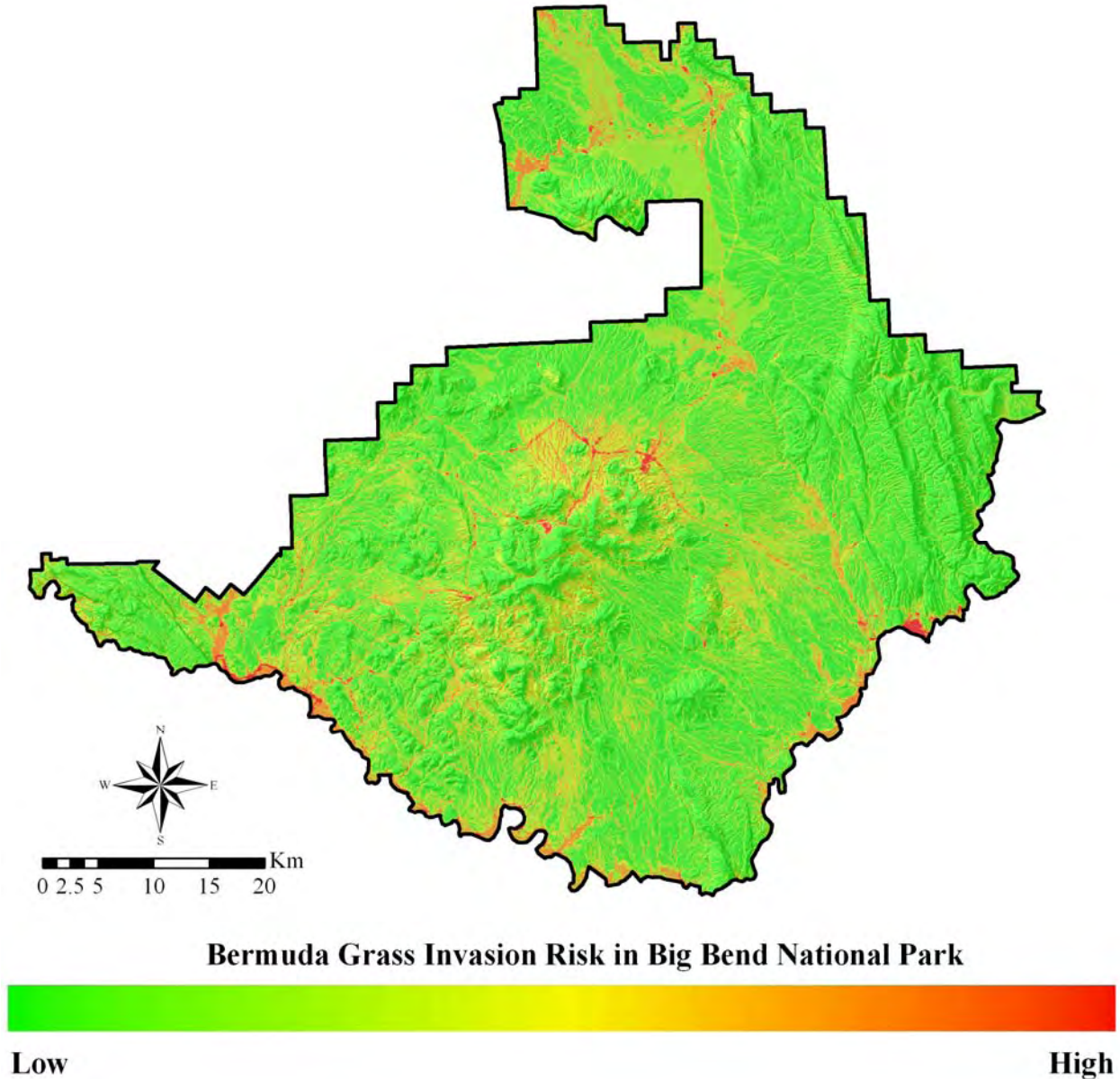


Figure 4-3. Areas at risk of Bermuda grass incursion or spread in Big Bend National Park. The likelihood of Bermuda grass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of rangelands, fields, riparian areas, and understories of open woodlands may be threatened by Bermuda grass as demonstrated by the moderate or high risk areas (orange to red) in the figure.

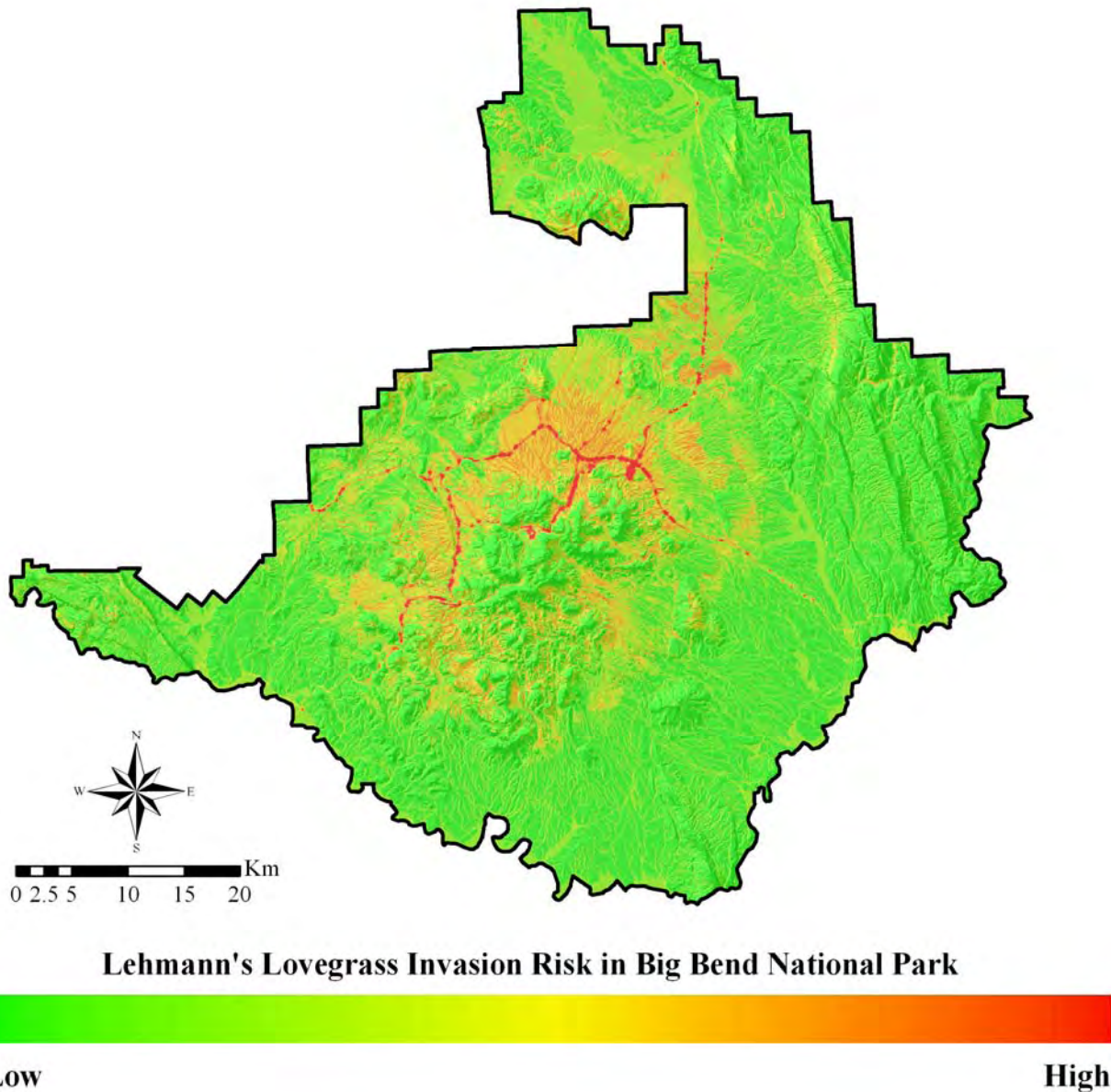


Figure 4-4. Areas at risk of Lehmann's lovegrass incursion or spread in Big Bend National Park. The likelihood of Lehmann's lovegrass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function desert shrubland and semi-desert grassland communities may be threatened by Lehmann's lovegrass as demonstrated by the moderate or high risk areas (orange to red) in the figure.

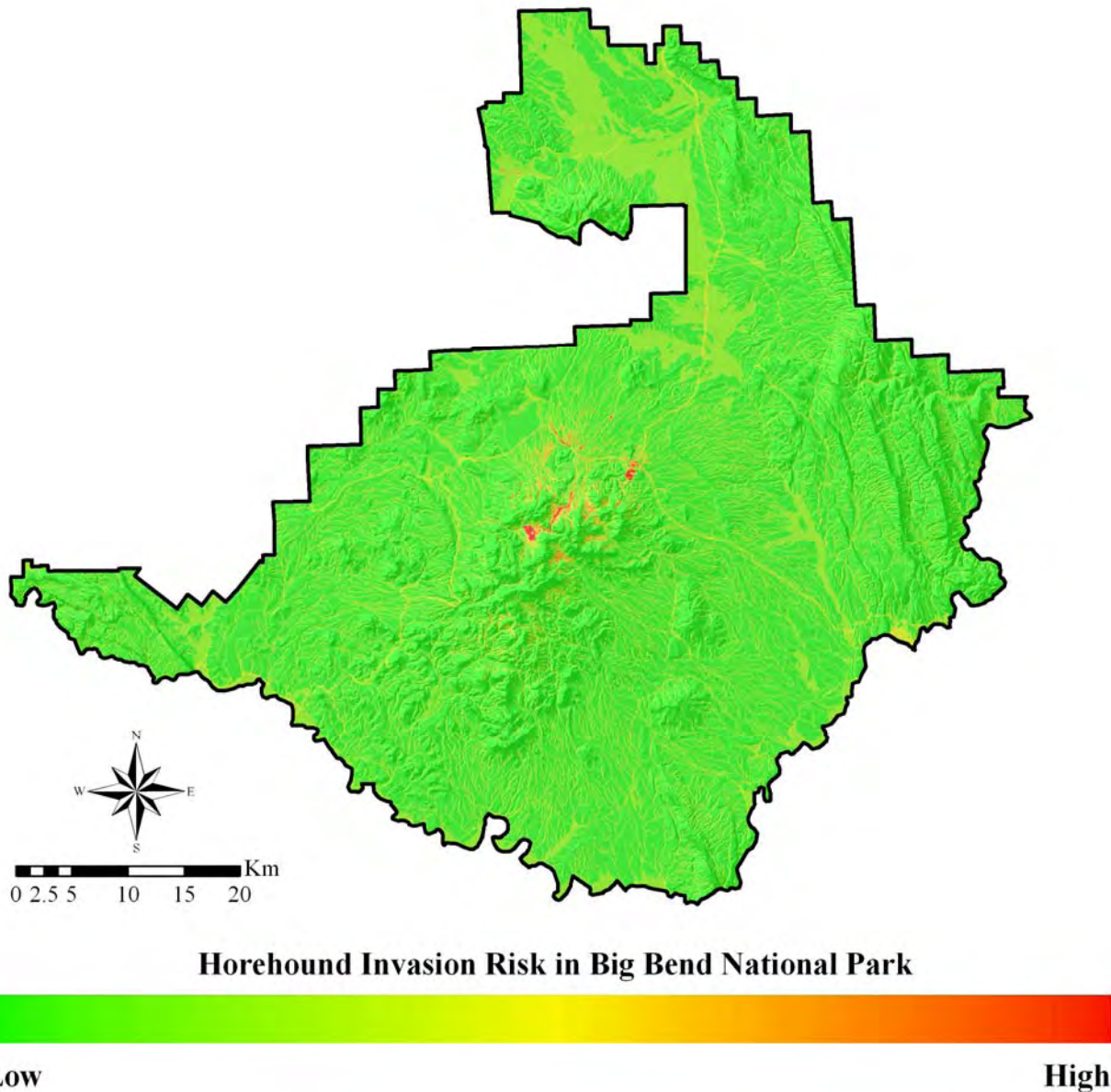


Figure 4-5. Areas at risk of horehound incursion or spread in Big Bend National Park. The likelihood of horehound incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function desert shrubland and semi-desert grassland communities may be threatened by horehound as demonstrated by the moderate or high risk areas (orange to red) in the figure.

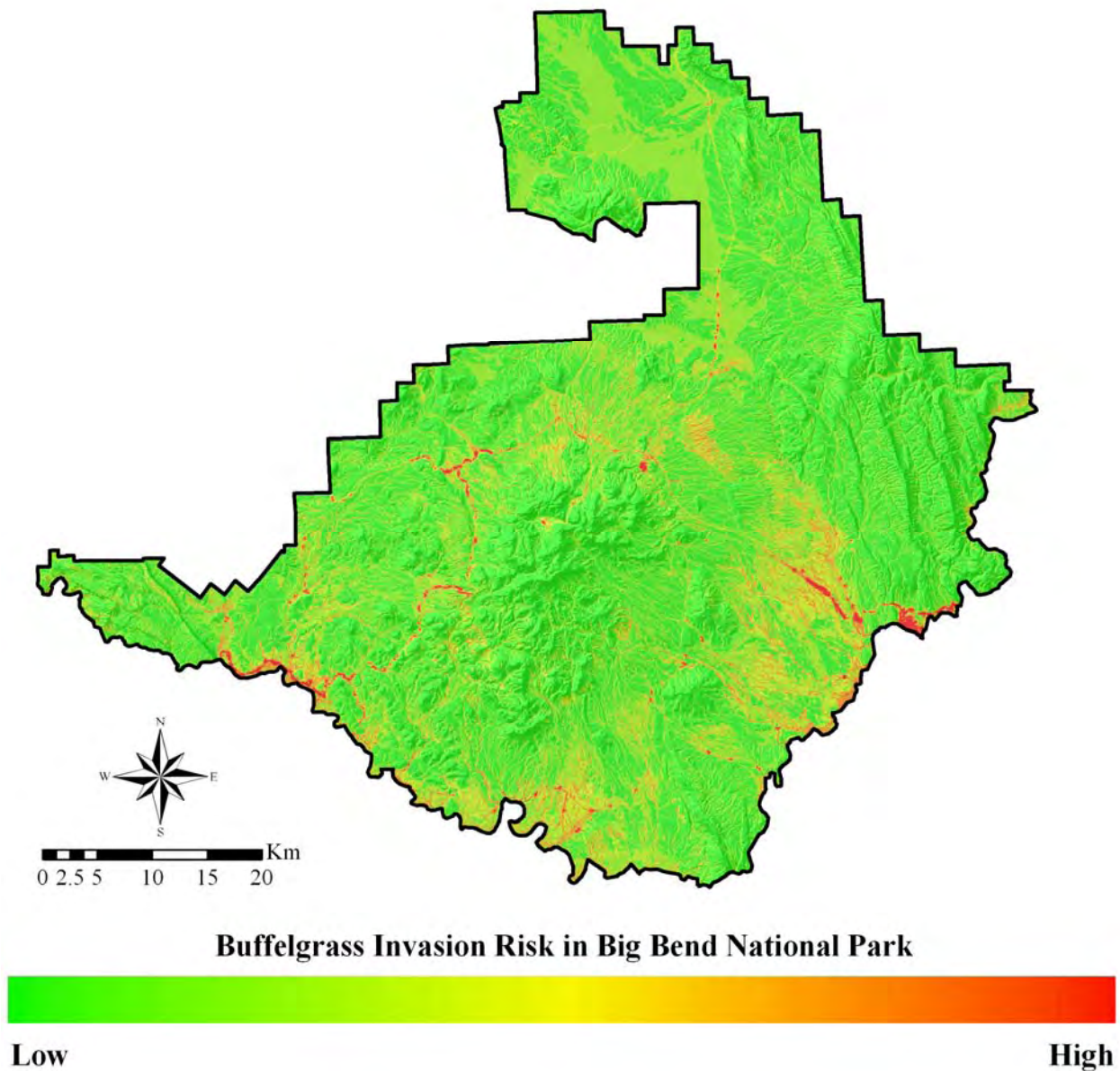


Figure 4-6. Areas at risk of buffelgrass incursion or spread in Big Bend National Park. The likelihood of buffelgrass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of semi-desert grassland communities may be threatened by buffelgrass as demonstrated by the moderate or high risk areas (orange to red) in the figure.

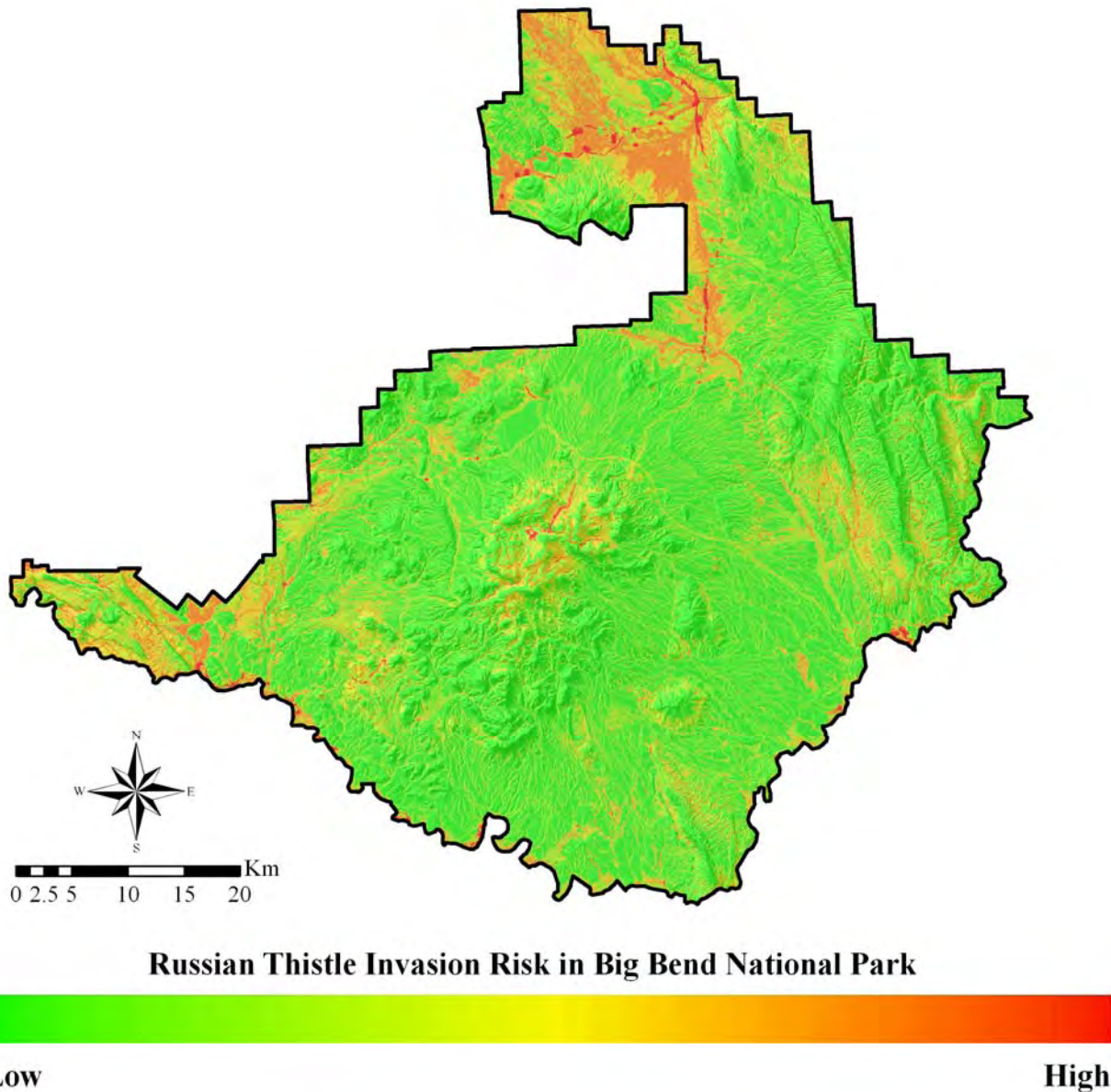


Figure 4-7. Areas at risk of Russian thistle incursion or spread in Big Bend National Park. The likelihood of Russian thistle incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of semi-desert grassland communities may be threatened by Russian thistle as demonstrated by the moderate or high risk areas (orange to red) in the figure.

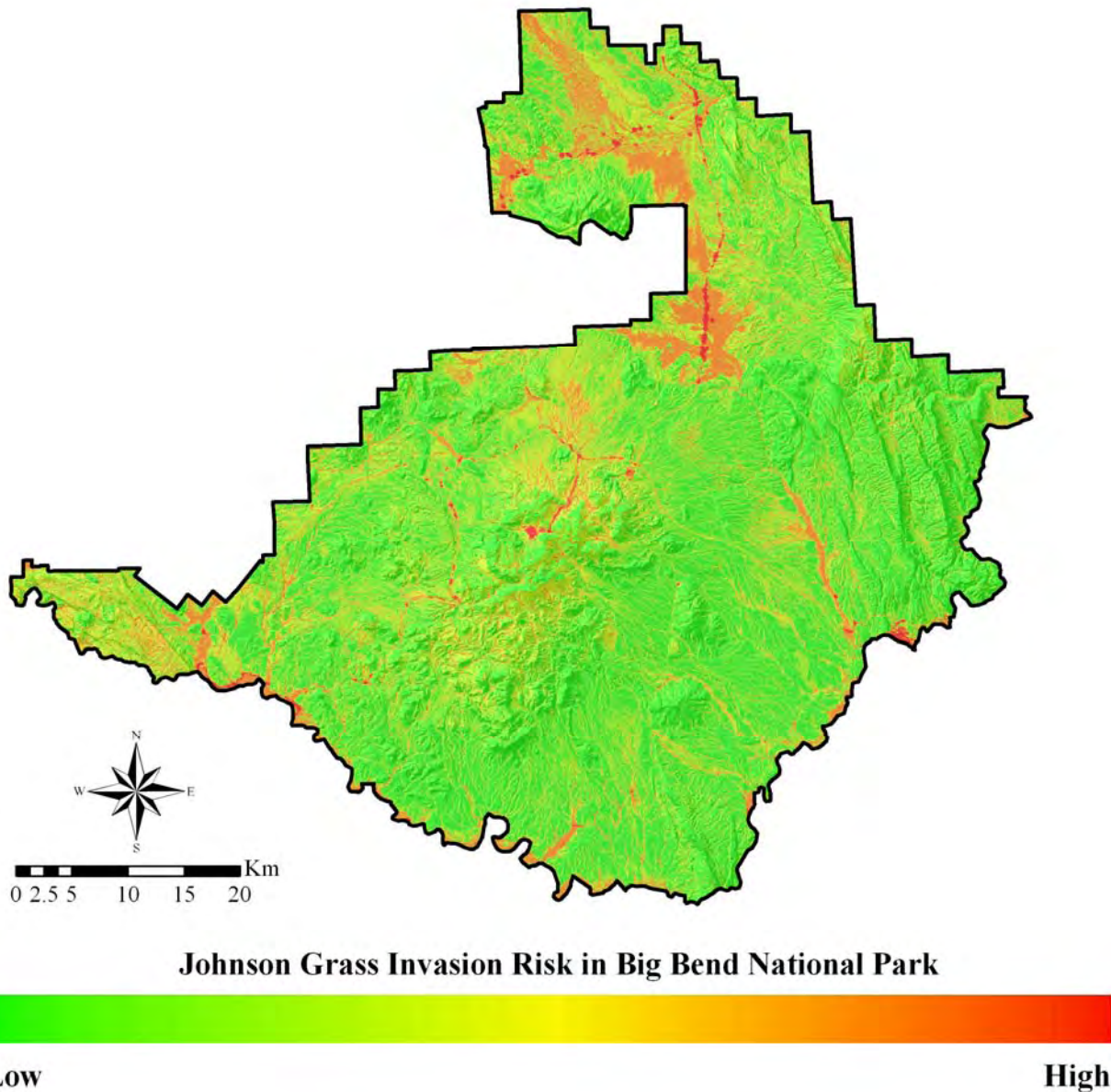


Figure 4-8. Areas at risk of Johnson grass incursion or spread in Big Bend National Park. The likelihood of Johnson grass incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian habitats and springs may be threatened by Johnson grass as demonstrated by the moderate or high risk areas (orange to red) in the figure.

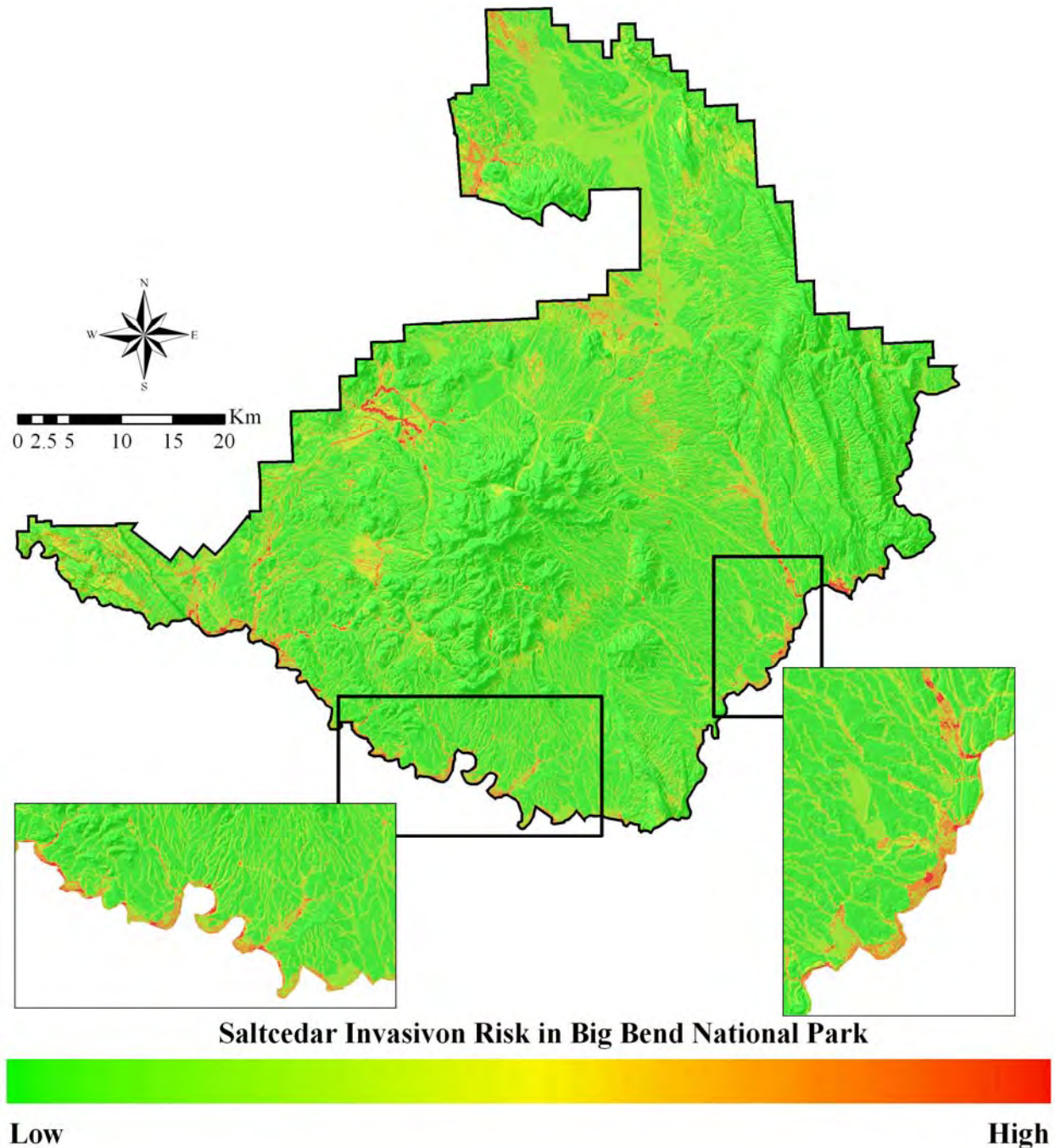


Figure 4-9. Areas at risk of saltcedar incursion or spread in Big Bend National Park. The likelihood of saltcedar incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Ecological stability and function of riparian habitats and springs may be threatened by saltcedar as demonstrated by the moderate or high risk areas (orange to red) in the figure.

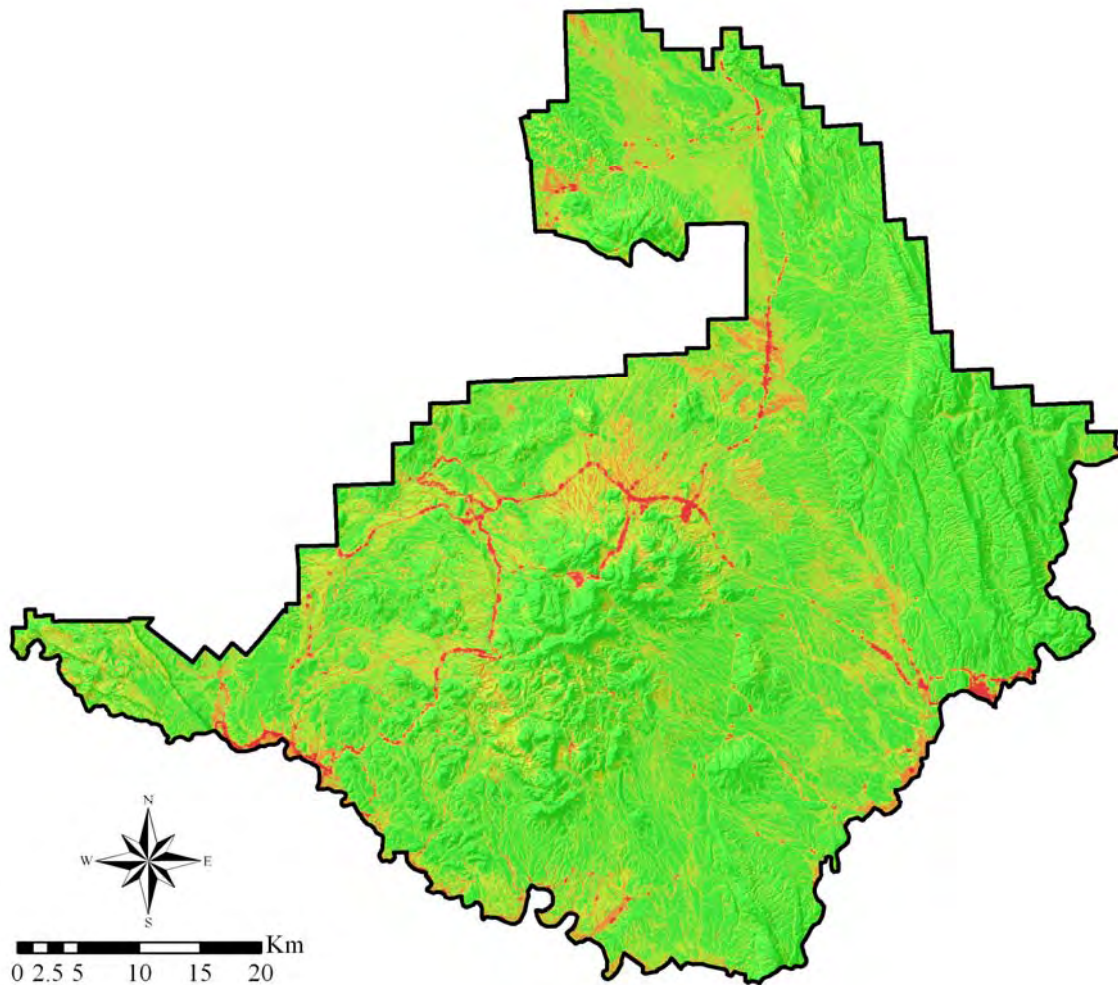
**Plant Invasion Risk (modeled from the 8 target species) in Big Bend National Park**

Figure 4-10. Areas predicted to be at risk by at least one of the eight target invasive plant species in Big Bend National Park. The likelihood of invasive plant incursion was estimated from spatial analyses of current target invasive plant locations, predicted suitable habitats, vectors and pathways, and disturbances. Primary areas at moderate or high risk (orange to red areas) which may suffer modified ecological stability and function include areas immediately proximate to riparian and spring habitats, roads and human activity centers.

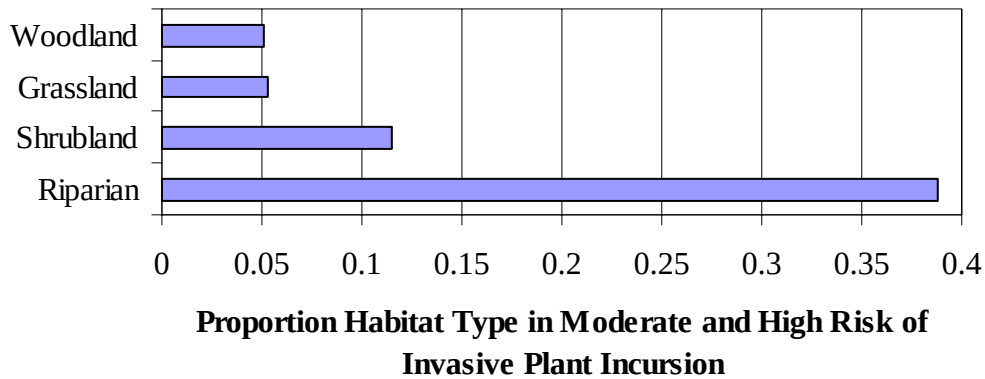


Figure 4-11. Proportion of general habitat types modeled by Plumb (1991) that are in moderate and high risk of invasive plant incursion in Big Bend National Park.

Desert shrubland and grassland habitats in BIBE are potentially threatened by buffelgrass, Bermuda grass, and Lehmann's lovegrass. Park staff conducted invasive plant surveys outside the Rio Grande corridor along roads (paved and dirt), campgrounds, and around developed areas in the park in 1998. They estimated approximately 64 ha (159 ac) of the area surveyed was infested by exotic species (BIBE 1998). Of that total, Johnson grass, Bermuda grass, Lehmann's lovegrass, buffelgrass and Russian thistle accounted for 98% of the acreage infested: Johnson grass was estimated as occupying 28.6 ha (70.8 ac), Bermuda grass 11.5 ha (28.5 ac), Lehmann's lovegrass 9.5 ha (23.5 ac), buffelgrass 8.9 ha (21.9 ac), and Russian thistle 4.5 ha (11.0 ac) (BIBE 1998).

The park's evaluation concluded that saltcedar and buffelgrass had the greatest negative impact on the park's resources, followed by Bermuda grass, Lehmann's lovegrass, and Johnson grass (BIBE 1998; see Chapter 2). Our results indicate that Johnson grass may pose the greatest risk to park resources, in terms of acreage of suitable habitat (Chapter 3) and area at risk, followed Lehmann's lovegrass, Bermuda grass, buffelgrass, and saltcedar (Fig 4-12). Russian thistle may pose

a serious threat as well, however, the predictive ability our model was weak for this species (see Chapter 3). Our analysis differs from the park's evaluation in that we considered the ecological impact of the presence of a species equal across all eight target plants. The eight invasive plant species are aggressive colonizers of disturbed sites and can outcompete native vegetation by crowding and exploiting limited soil resources. These plants also alter fire frequency and intensity which can lead to further invasions. More detail evaluation of ecological impacts, or incorporating costs associated with control efforts, could change the order of plants that pose the most risk to the park's natural resources.

Conservation Implications

Managing plant invasions is challenged by discrepancies in timing when invasions occur and when invasions are observed or recorded (Campbell et al. 2002). Understanding the likelihood of an invasive plant occurrence across large landscapes can aid in establishing early detection protocols which may reduce the severity of invasions. Landscape scale risk assessments provide a powerful tool to help in this endeavor. Invasive species risk

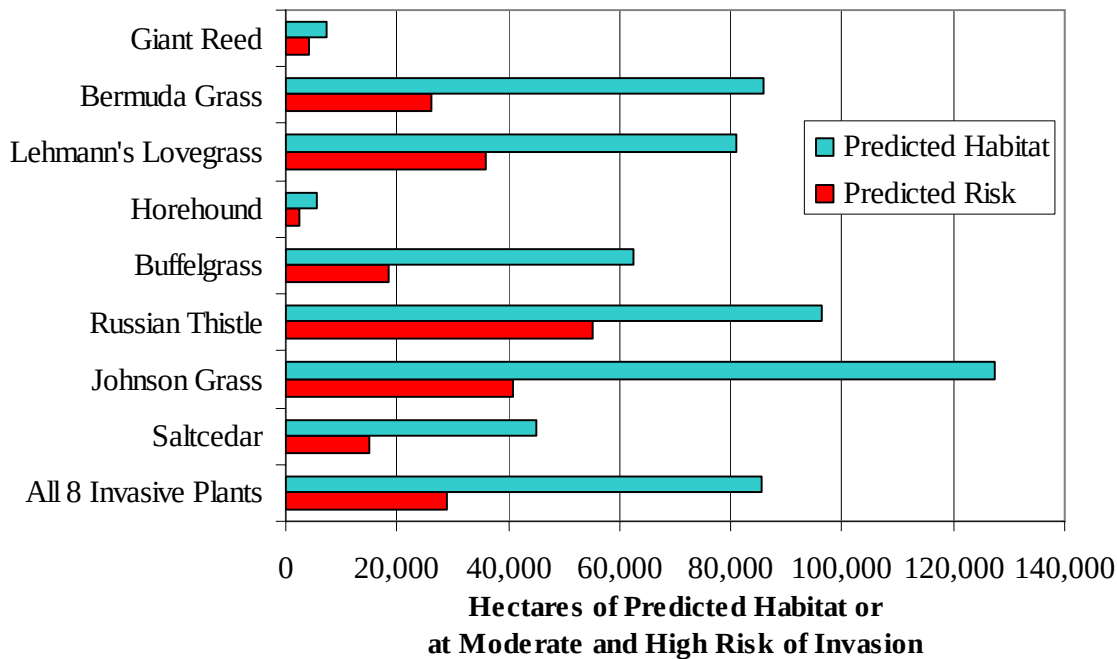


Figure 4-12. Amount of area (hectares) predicted to be suitable habitat (Chapter 3) and predicted to be at moderate or high risk of invasion by our target invasive plants in Big Bend National Park.

assessments synthesize information about invasive species occurrences or likelihood of occurrences and their potential effects on biotic resources to form conclusions that are informative for conservation efforts (EPA 1998, Anderson et al. 2004a).

Our risk models demonstrated an effective approach to assess landscapes for risk to incursion of invasive plants. These models can inform park managers regarding the potential introduction of invasive species, their vectors and pathways, and the allocation of resources for the control of invasive species. Additional regional analyses would provide a greater understanding of potential species that could alter important structural or functional components of Park's ecosystems (EPA 1998). Regional assessments provide insight into the potential pathway that invasive species could enter park boundaries. Further, incorporating demographic parameters into

risk analyses may provide insight into the rates of spread which may provide estimates of when plants will reach critical areas. For plants that are sensitive to habitat patch dynamics, habitat connectivity and fragmentation datasets could also be incorporated into risk analyses.

In addition to prioritizing areas to monitor or control invasive plants, our risk models can be used to evaluate the effects of invasive species occurrences to Threatened and Endangered species (T&E), species of concern, recreational activities, and planned management actions. For example, risk models can be used to evaluate spatial locations of sensitive or T&E species to see if any populations may be affected by invasive species occurrences. Areas at risk would then receive priority for survey, monitoring, or control efforts. When considering areas for management, it may be important to consider

the potential synergistic effects between invasive species impacts and other environmental stressors, especially in regards to ecosystem function and stability. For example, buffelgrass typically grows at a higher density than native grass species, which may reduce the amount of native plants and promotes higher intensity fires (Martin et al. 1999). Buffelgrass is fire tolerant and sprouts immediately post-fire, which further out-competes native grasses (Hamelton and Scifres 1982) leading to expanded pure buffelgrass populations. Drought and climatic

change amplify this cycle. The ultimate loss of plant biodiversity leads towards a loss of vertebrate diversity, soil stability and ecosystem functioning. These factors further degrade ecosystems processes which promote additional unwelcome invasions. Methods that promote early detection of invasive plants would allow for the implementation of proactive management practices to decrease the prevalence of nonnative species through eradication or control efforts, and maintain ecosystem integrity.



Chapter 5

Sampling Design Simulation

Introduction

Knowledge on the occurrence of invasive plants is often gained through incidental or opportunistic observations. These observations may be from park staff or visitors that are familiar with the plant. While these observations provide valuable insight and information for the early detection of invasive plants, there is a need to develop early detection techniques to improve the likelihood of finding invasive species before they become a problem in an area. For example, search effort focused on areas that are most likely to be invaded improves the efficiency of finding invasive species and provides greater information for early detection protocols. Our predictive invasive plant habitat models and risk assessments (Chapters 3 and 4) provide a means to concentrate or allocate survey efforts, especially across large landscapes. The need and ability to identify current and potential distributions of invasive species is often confounded by logistical and financial constraints. These constraints may be minimized by identifying field sampling designs that provide efficient assessment of invasive plant distributions. Although field sampling is not a requirement for finding and eradicating invasive population clusters, sampling procedures can improve search efficiency, allow for assessment of infestation severity, provide a framework to make estimates across unsampled areas, and improve predictive models (Geissler *in review*).

For example, conducting a complete survey for invasive plants in a large National Park is typically not possible. However, we can gain a better understanding by selecting a subset of park, conducting an appropriate survey, and

then extrapolate our findings from the small subset to the entire park. The key is to survey a representative sample, or subset of the entire park. Sampling designs are used to ensure that a representative sample is selected from the entire population. Sampling designs that employ probability sampling procedures provide statistically unbiased estimators. There are multiple sampling designs available to land managers. A few common examples of sampling designs include simple random sampling, stratified random sampling, cluster sampling, systematic, or grid sampling (Thompson 1992).

Simple random sampling is perhaps the most straightforward design. From a list of the population, n numbers of locations are randomly selected (Dowdy and Wearden 1985, Elzinga et al. 1998). Each unit in the population has equal probability of being sampled. For example, in a large park, Universal Transverse Mercator (UTM) coordinates could be selected randomly as survey points.

Stratified random sampling divides the population into groups or strata (Elzinga et al. 1998). Random samples are then selected within each stratum. In our large park example, habitat type or distance to roads could be used to divide the park into natural groupings (or strata) where sample units within groups are similar. This similarity may help reduce variation and provide more accurate inferences. However, in terms of invasive species sampling, choosing strata may help focus sampling on critical areas (e.g., disturbed areas, or along roads) rather than sample areas with low likelihood of invasive plant occurrences. These designs are often considered probability proportion to size

designs because they allocate effort to units with higher likelihood of plant occurrences (Geissler *in review*). Within each stratum, Universal Transverse Mercator (UTM) coordinates could be selected as survey points.

Cluster sampling is similar to stratified sampling. However, typically with cluster sampling, a number of samples are collected at each sampling unit (Elzinga et al. 1998, Thompson 1992, Phillippi 2005). Data collected at points along transects or within subplots are examples of cluster samples. Cluster sampling is efficient when there is a large cost associated with locating sampling units, or when plant distributions are spatially clumped, as in the case of new plant infestations. In our large park example, we might select random samples of the park, either as a simple random sample or a stratified random sample, to survey for invasive plants. If an invasive plant is detected on the sampling unit, additional plant surveys may be conducted in the neighborhood of the site to detect additional plants (Smith et al. 2004). These neighborhoods can take a variety of shapes, including a cross-shaped configuration (search effort conducted in each cardinal direction), or searching in all directions around the original detection (bookshelf configuration) (Thompson 1992). These adaptive cluster sampling approaches lend themselves nicely to adaptive management needs. Statistical procedures are available to reduce the inherent bias in this sampling strategy (Smith et al. 2004).

Systematic sampling is a design that selects every sample within a predefined interval across the population (Elzinga et al. 1998). Other than perhaps a random starting point for the interval, this design typically does not contain a random component. It is however, easy to implement in the field because the time needed to find sample units is reduced.

In our large park example, we might select systematic samples by placing a grid over the entire park. We might select the desired number of samples for the park, and calculate an appropriate interval to survey that allows for the desired number of samples collected systematically across the park or strata. A random starting point or a constrained random starting point (e.g., the grid number to start sampling) is often employed.

Grid sampling is also a common sampling design, especially studies that utilize GIS technology to select samples. Grid sampling would employ placing a grid over the area of interest (Elzinga et al. 1998). The size of the grid may reflect a desired area to survey (quadrat or plot) or primary sample unit to establish secondary units. Grids to receive sampling effort would be selected randomly (Smith and Nichols 2006).

Field sampling designs can also be classified as single-stage, two-stage, and stratified designs with non-adaptive, adaptive, or GIS-based adaptive modifications. Single stage sampling focuses on single samples collected similarly throughout some a priori defined distribution. A simple random sample of points selected throughout the park is an example. Two-stage sampling (synonymous with multi-stage or double sampling) uses data collected at secondary sampling units, or subunits. For example, primary sampling units within a large park or natural area may be randomly selected. Subunits within these grids would be randomly selected and surveyed for the presence of invasive species. These subunits may be a smaller grid, subplots, points, or transects. In some applications, primary units must be sampled prior to secondary unit sampling (Francis 1984, Jolly and Hampton 1990, Salehi and Seber 1997, Christman 2003). However, Salehi and Smith (2005) provide a method to immediately sample secondary units from

primary sampling units. As previously mentioned, stratified sampling divides the study area into homogenous groups or strata. Single stage, two-stage, and stratified designs can be sampled using simple random, probability proportion to size, systematic sample, or grid based approaches as discussed above (Smith and Nichols 2006).

Sampling designs can be modified using adaptive or GIS-based adaptive frameworks; non-adaptive sampling maintains the sampling design without modification for the actual populations. The adaptive sampling effort starts with an initial random sample and is then modified with field detection. Our discussion of cluster sampling with neighborhood searches is an example of adaptive sampling effort. GIS-based adaptive sampling uses auxiliary information in spatial datasets to modify the adaptive sampling framework (Smith and Nichols 2006). Auxiliary information may include habitat suitability, risk analyses, disturbance datasets, or other factors that influence the occurrence of invasive plants.

Greater information and detailed reviews of sampling designs are provided by Thompson (1992), Phillipi et al. (2001), Smith et al. (2004), and Geissler (*in review*). Elzinga et al. (1998) also provides an easy to understand discussion on sampling designs and methods for measuring and monitoring plant populations. Goals and objectives of the survey program and the efficiency that species are detected will ultimately influence the method chosen for field sampling.

The objective of this chapter was to simulate various sampling designs to identify appropriate field sampling procedures that may provide National Park Service personnel efficient methods to assess areas for invasive plant species occurrence. We simulated sampling designs for giant reed (*Arundo*

donax), Bermuda grass (*Cynodon dactylon*), Lehmann's lovegrass (*Eragrostis lehmanniana*), horehound (*Marrubium vulgare*), buffelgrass (*Pennisetum ciliare*), Russian thistle (*Salsola kali*), Johnson grass (*Sorghum halepense*), and saltcedar (*Tamarix* spp.) (see Chapter 2) in Big Bend National Park (BIBE). We also simulated sampling designs for all eight species together to investigate efficient sampling methods that could detect multiple species. Our focus was placed on adaptive sampling comparisons within single or two-stage sampling efforts.

Methods

We used SAMPLE software version 1.4.2 Beta (Smith and Nichols 2006, <http://www.lsc.usgs.gov/aeb/davids/acs/>) to run sampling simulations that identified the most efficient methods for sampling each species and evaluated if adaptive sampling techniques would benefit sampling effort. SAMPLE is public domain software that uses population and suitable habitat data to run simulations of varying sampling designs. SAMPLE allows for the evaluation of single-stage, two-stage, and stratified sampling designs in the context of adaptive and non-adaptive strategies using simulations generated from GIS datasets. The software also allows for comparisons of systematic, proportional, simple random, and systematic sample allocation for single and two-stage sampling designs (Smith and Nichols 2006).

Datasets Used for Simulations

Two sources of information were required for our sampling simulations; 1) defined study area size, and 2) spatial locations of known invasive plant species. We used the entire BIBE as our study area. We further modified the area of interest in the study area depending on our simulations. For example, our single-stage simulations were spatially limited to areas that currently or could in the future

Table 5-1. Matrix of sampling designs evaluated using SAMPLE software (Smith and Nichols 2006) for each invasive plant species in Big Bend National Park.

Stage	Adaptive	Sample Allocation	Acronym
Single	Non-adaptive	Simple Random Sample	Single-stage SRS-NA
Single	Non-adaptive - Grid	Random Start, Systematic	Single-stage Grid-NA
Single	Adaptive	Simple Random Sample	Single-stage SRS-A
Single	Adaptive - GIS	Simple Random Sample	Single-stage SRS-GIS-A
Two Stage	Non-adaptive	Simple Random Sample	Two-stage SRS-NA
Two Stage	Adaptive	Simple Random Sample	Two-stage SRS-A

support populations of each invasive plant species. We used each species predicted habitat model derived in Chapter 3 to define these areas. We converted our predicted habitat models to a binary (0, 1) dataset using a threshold that provided an equal tradeoff between sensitivity and specificity (see Chapter 3 for greater detail). Areas with suitability values above this threshold were considered areas that had high likelihood of hosting suitable habitat conditions for the plant. These areas were classified as a “1.” Areas below the threshold were considered non-suitable habitats and were classified as a “0.” We then rescaled our predictive habitat models from 30 m (98.4 ft) pixel size to 90 m (295 ft) pixel size to facilitate simulation runs.

We also used our risk assessment datasets (created in Chapter 4) in the adaptive GIS simple random sample simulations. These risk grids had values ranging from 0 to 1, with larger values indicating areas at higher risk of invasive plant incursion. Using the predictive habitat models and the risk assessment models in the single-stage, adaptive GIS simulations added additional risk insight to probability of plant incursion in the suitable habitat areas. If the relationship between our risk assessment and species presence is strong, then this sampling design will be effective (D. Smith, personal comm.). However, risk assessment models could also be used as sampling strata by identifying various thresholds (e.g., low, moderate, and high risk areas: see Chapter 4 for greater details).

We used known invasive plant localities (UTM coordinates) gained from our field work and historical park data (see Chapter 3) to provide population estimates, in a presence-only context. SAMPLE can use plant population numbers in analyses; however, plant population data at each location was unavailable. As such, our simulations are not adjusted for population size or extent.

All datasets were projected in UTM zone 13, GRS 1980 Spheroid and NAD 83 Datum. Habitat and risk grids were converted to generic ASCII raster format for analyses in SAMPLE. Likewise, plant localities were saved to a text file (.csv). All data conversions were conducted in ArcGIS 9.2.

Sampling Designs Evaluated

We concentrated our sampling simulations to single-stage and two-stage designs to focus on sampling efficiency between adaptive and non-adaptive techniques within single-stage or two-stage approaches (Table 5-1).

Single-Stage Designs

The simplest of the single-stage sampling designs evaluated was the non-adaptive, simple random sample (Single-stage SRS-NA) design. This approach derived a simple random sample across all predicted suitable habitats in BIBE. The non-adaptive grid based sample (Single-stage GRID-NA) design placed a 270 x 270 m (886 x 886 ft) grid across suitable habitats. A random starting pixel was selected, and subsequent pixels were

systematically sampled across the grid (Smith and Nichols 2006).

The adaptive single random sample (Single-stage SRS-A) design is an adaptive cluster sampling technique where the initial sample is selected randomly in suitable habitats. In cases where the random location yielded a positive plant occurrence, subsequent samples were collected in each of the four cardinal directions (adaptive cluster approach). This process would be repeated until no additional positive plant occurrences were sampled. The adaptive GIS simple random sample (Single-stage SRS-GIS-A) design is similar, except it used a risk dataset (created in Chapter 4) as the basis for adaptive sampling.

Two-Stage Designs

Our simulations for the two-stage sampling designs were conducted across the entire study areas, regardless of habitat suitability. The non-adaptive two-stage simple random sample (Two-stage SRS-NA) design focused sampling efforts in secondary units (subunits) within primary randomly selected sample units. Primary units were 450 x 450 m (1,476 x 1,476 ft), which allowed for 25 secondary units in each primary unit. Likewise, the adaptive two-stage simple random sample (Two-stage SRS-A) approach employed subsequent samples in each of the four cardinal directions (adaptive cluster approach) when there was a positive plant occurrence in

the subunits.

Simulation Settings

We used the default program settings in SAMPLE for our simulations. We ran 100 simulations (replications) for each sample design and plant. Single-stage designs sampled 5% of the study area. We used a cross-shaped adaptive neighborhood with a 1 pixel shift criterion for the adaptive analyses.

Two-stage sampling defaults were similar, with 100 simulations completed for each sample design and plant. Likewise, 5% of the study area was sampled by primary sampling units. Secondary sampling units sampled 5% of the primary units. Adaptive criteria were identical to single-stage analyses.

Evaluation Metrics

We used three metrics provided in SAMPLE outputs to evaluate sampling designs; efficiency, occupancy, and hits (Table 5-2) (Smith and Nichols 2006). The efficiency metric identifies designs that potentially yield lower variability estimates than a simple random sample given the same sampling effort (Smith and Nichols 2006). Efficiency is calculated as the ratio of the population variation and sample size from a non-adaptive simple random sample and the variation of a chosen sampling design. Because the simple random sample variation is calculated using population variations, it is independent of the

Table 5-2. SAMPLE output metrics used to evaluate sampling designs for each of the target invasive plant species in BIBE (Smith and Nichols 2006).

Parameters	Description
Efficiency	Identifies designs that yield lower variability estimates than a simple random sample given the same sampling effort. It is calculated as the ratio of the population variation from a non-adaptive simple random sample and the variation of a chosen sampling design.
Occupancy	An estimate of the proportion of sampling units in the population that are occupied by the target species. This metric provides a measure of the degree of rarity.
Hits	Identifies the proportion of the sample that is occupied by the invasive plant. Value for hit should be greater than occupancy for proficient sampling. Calculation varies from occupancy with modified inclusion probabilities.

simulation results (D. Smith, personal comm.). As such, efficiency values > 1 represent a design that is more efficient (in terms of sampling variance) than simple random sample. Further, efficiency values can be used to evaluate if plant populations are clustered. For example, when efficiency values yielded from an adaptive approach are higher than a non-adaptive approach, it suggests the plant populations are clustered.

Occupancy is an estimate of the proportion of sampling units in the population that are occupied by the target species. As such, occupancy provides a measure to the degree of rarity of a population (Smith and Nichols 2006). Conversely, the metric “hits” is the proportion of the sample that is occupied by the target invasive plant species. Thus, “hits” is analogous to sensitivity (true positive occurrences) discussed in Chapter 3.

In a simple random sample, occupancy and hits yield the same value. However, in more complex designs (e.g., stratified, probabilistic, or adaptive sampling) the calculation of occupancy becomes complex, and depends on the modified inclusion probabilities (D. Smith, personal comm.). Generally, proficient sampling procedures yield higher hits values than occupancy values. As such, we use the proportion of hits to occupancy to guide our selection of adaptive procedures. However,

because of the difference in the area searched between single-stage (suitable habitat only) and two-stage (suitable and non-suitable habitats) designs, we limit our comparisons to designs within single stage and designs within two-stage methodologies.

Results and Discussion

Incorporating knowledge about the risk of invasion into a sampling design should provide greater efficiency in allocating field samples. Unfortunately, we were only able to estimate the Single-stage SRS-GIS-A design for giant reed, horehound, and saltcedar. Likewise, we were unable to estimate the Single-stage GRID-NA design for all invasive plants combined. Our inability to estimate these designs for our plant species was likely a function of our study area size, complexity of the risk grids, and perhaps hardware limitations. For all estimable designs, we present efficiency, occupancy, and hits metrics in Appendix H. Graphs of this data are presented for each species.

Giant Reed

We observed higher efficiency values with the single-stage, adaptive simple random sample GIS method (Single-stage SRS-GIS-A) than all other single-stage approaches (Fig 5-1). Further, there was little difference in estimates of occupancy and hits for all designs.

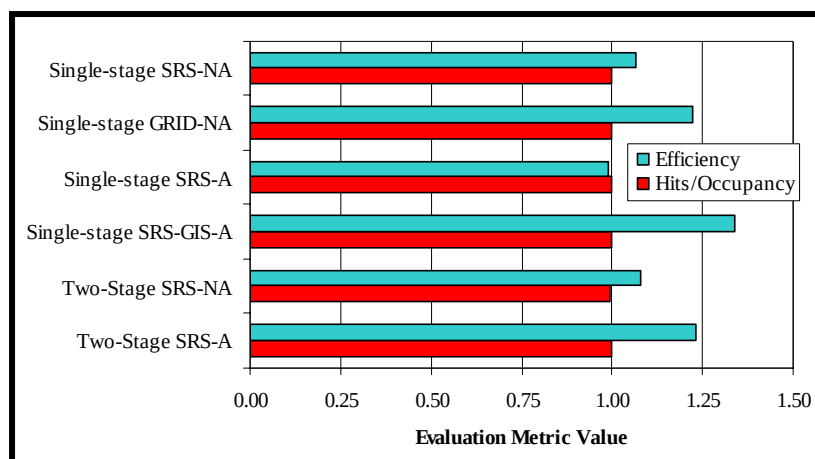


Figure 5-1 (Left). Efficiency and the hits to occupancy ratio for giant reed field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006).

Likewise, the two-stage adaptive (Two-stage SRS-A) design yielded higher efficiency values than the non-adaptive (Two-stage SRS-NA) design, although the hits and occupancy values were the same.

Although comparing single- and two-stage metrics is not valid, their efficiency metrics can provide insight into the degree of population clustering in BIBE and in suitable habitats. For example, our two-stage sampling designs evaluated both suitable and non-suitable habitat. Since efficiency metrics were higher for adaptive techniques than non-adaptive techniques, we can conclude that giant reed may exhibit clustered populations within BIBE. These results are supported by the life history of this plant. Giant reed is a hydrophyte restricted to the perennial waterways of the Rio Grande. Further evaluation of single-stage efficiency metrics (difference between Single-stage SRS-NA and Single-stage SRS-A) indicates that giant reed may not be clustered in suitable habitats, (e.g., higher efficiency values for SRS-NA) (Fig. 5-1). These higher SRS-NA efficiency values indicate that reliable population estimates may be derived from a simple random sample of suitable habitats along the Rio Grande corridor (e.g., waterways and adjacent riparian systems). However, our results may be influenced by the low number of sample

locations recorded ($n=31$), despite large established populations that exist along the Rio Grande. The degree that our opportunistic sample influenced simulations is unknown. Additional samples collected along the Rio Grande, in a random or systematic manner, may influence our results and thus the interpretation of plant clustering.

Although giant reed populations may not be clustered, our results indicate that some populations may be clustered around high risk areas, indicated by greater efficiency values of the SRS-GIS-A simulation. This may be an important consideration when designing a sampling protocol for this plant. Since giant reed is already well established in suitable habitats, incorporating areas at risk into samplings procedures may help in early detection of expanding populations.

Bermuda Grass

Analysis of single-stage designs for Bermuda grass identified the Single-stage GRID-NA design as having higher efficiency than a Single-stage SRS-NA or Single-stage SRS-A design (Fig. 5-2). However, evaluations using hits and occupancy metrics indicated that the SRS adaptive approach (Single-stage SRS-A) was a more effective design (indicated by greater hit metrics than occupancy metrics). The adaptive two-stage design (Two-stage

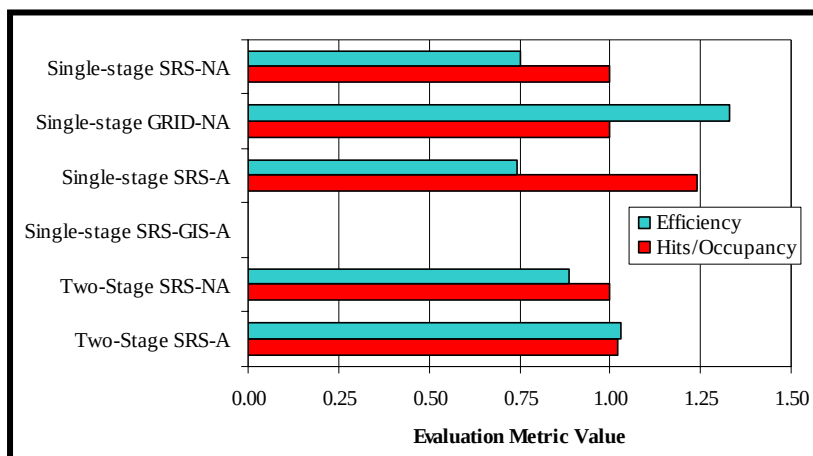


Figure 5-2 (Left). Efficiency and the hits to occupancy ratio for Bermuda grass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset.

SRS-A) had a higher efficiency value and a slightly higher hits to occupancy ratio than the non-adaptive (Two-stage SRS-NA) design (Fig 5-2). Sample size (n=117) likely did not limit our analysis.

Comparison of adaptive and non-adaptive efficiency values also indicates that Bermuda grass is spatially clustered across BIBE, but is likely not clustered in suitable habitats. Bermuda grass is a specialist requiring mesic conditions found along rivers, roadways and stock tanks. There are large populations occurring around developed areas and campgrounds where it is used as a turf or lawn grass. In these areas, sampling efficiency would be enhanced with the systematic sampling of the Grid-NA approach. Otherwise, parameter estimates obtained by sampling in suitable habitat area may not be substantially improved with adaptive techniques.

Lehmann's Lovegrass

Evaluation of single-stage designs for Lehmann's lovegrass indicated similar efficiency values for the grid based non-adaptive sampling (Single-stage GRID-NA) and the adaptive simple random sample (Single-stage SRS-A) design (Fig. 5-3). Despite the similar efficiency estimates, the Single-stage SRS-A yielded a substantially

higher hit to occupancy ratio, indicating that this sampling design may be more proficient than the grid based approach.

Likewise, the two-stage adaptive sampling (Two-stage SRS-A) approach yielded a slightly higher efficiency value and a substantially higher hit to occupancy ratio than the non-adaptive (Two-stage SRS-NA) approach.

Our Lehmann's lovegrass analysis likely benefited from the availability of 253 points for our sampling simulation. However, points were predominately clustered along roads in the park (see Fig. 3-6, Chapter 3). However, predicted areas of suitable habitat were not constrained to roadsides. Lehmann's lovegrass is a generalist species that can spread from both seed and stolons. This plant is an aggressive colonizer of disturbed areas and can easily colonize undisturbed areas with low canopy cover (Sumrall et al. 1991, Roundy et al. 1992, Ruyle et al. 1998). As such, populations may start fairly clustered, and then spread across landscapes. Our analysis indicated that a sampling design that employs adaptive sampling within suitable habitats will likely be the most efficient method of quantifying Lehmann's lovegrass populations. However, because of this plant's life history, sampling designs that incorporate

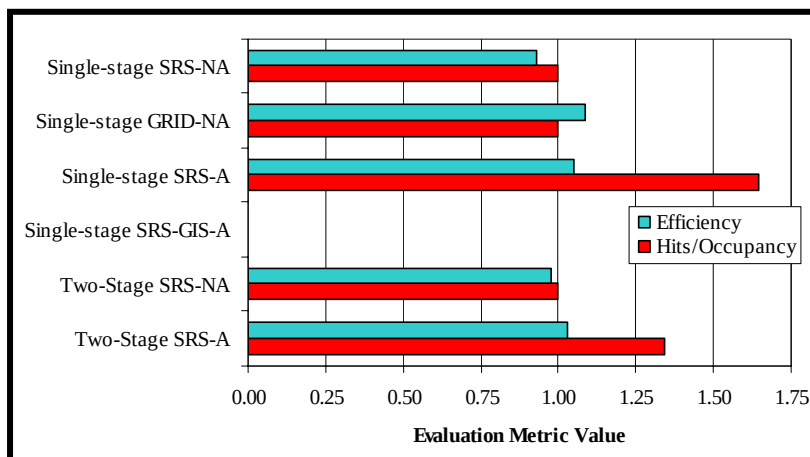


Figure 5-3 (Left). Efficiency and the hits to occupancy ratio for Lehmann's lovegrass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset.

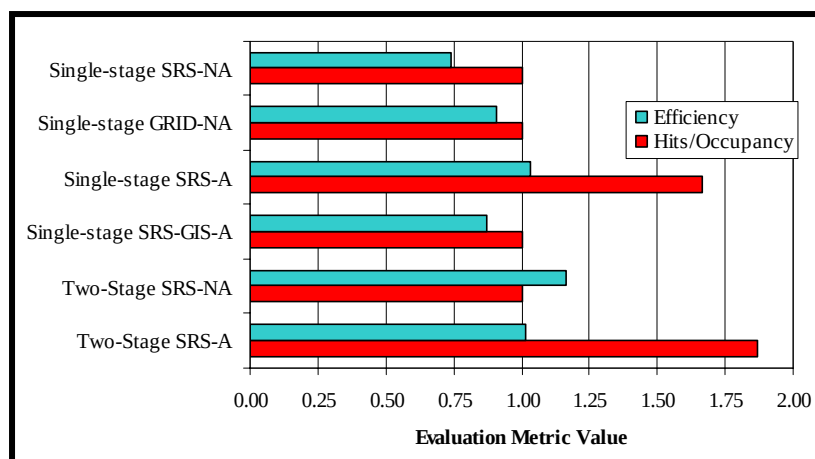


Figure 5-4 (Left). Efficiency and the hits to occupancy ratio for horehound field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006).

stratum (e.g., disturbance regimes) or risk assessment layers may be more efficient, and should be investigated.

Horehound

We had very few occurrence locations (n=26) of horehound to simulate a sampling designs. Horehound occurs in small clusters that are currently rare in BIBE. Further, our predicted suitable habitat occurred in isolated locations in BIBE, with the majority of suitable habitat occurring around developed areas near the Chisos Mountains (see Chapter 3, Fig. 3-7). However, because horehound is not yet widespread in BIBE, this plant may provide a good “test case” for developing sampling strategies for early detection protocols.

Single-stage sampling analyses indicated that the Single-stage SRS-A may be a more efficient sampling design than the other single-stage approaches evaluated (Fig. 5-4). The use of our risk assessment models in the Single-stage SRS-GIS-A simulation did not increase the sampling efficiency. Similar results were observed with the hit to occupancy ratio. Only the Single-stage SRS-A approach yielded a high hit to occupancy metric.

When we incorporated unsuitable habitats into our simulations (two-stage), our results were more ambiguous. The Two-stage SRS-NA approach yielded the highest efficiency metric, but the hit to occupancy metric was substantially higher for the adaptive (Two-stage SRS-A) approach. The limited distribution of our small sample size likely affected this analysis, but the extent is unknown.

Although horehound has wide environmental tolerances, it generally occurs in disturbed areas along roadsides, stockyards, pastures, and dwelling (Sievers 1930, Lippai et al. 1996, BIBE 1998). As such, adaptive sampling strategies that are focused in suitable habitats would likely provide the most robust population estimates and provide an adequate sampling procedure for early detection of this species.

Buffelgrass

Our single-stage design analyses indicated that the SRS adaptive method (Single-stage SRS-A) was more efficient than the other evaluated single-stage designs (Fig. 5-5). The SRS adaptive method also yielded a greater hit than occupancy metric, indicating it is a better approach.

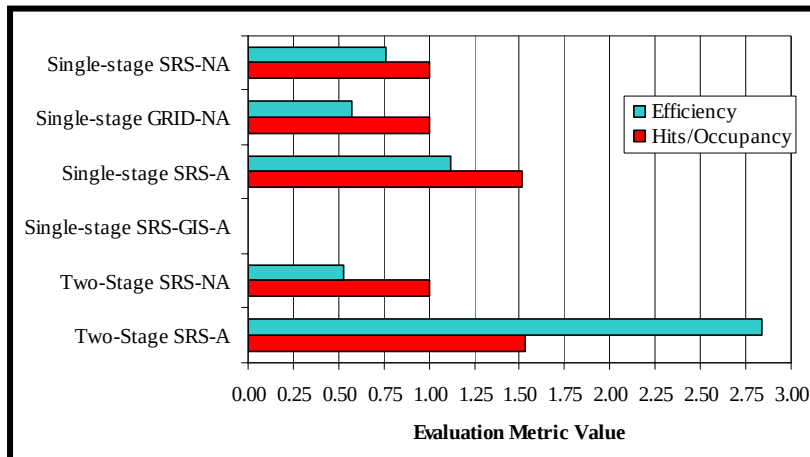


Figure 5-5 (Left). Efficiency and the hits to occupancy ratio for buffelgrass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset.

We observed similar results with the two-stage design analysis. The Two-stage SRS-A design yielded substantially higher efficiency and hit to occupancy ratio metrics than the non-adaptive approach.

Comparison of the non-adaptive versus adaptive metrics for both single-stage and two-stage samples indicate that buffelgrass is likely clustered in BIBE and within suitable habitats. We found buffelgrass throughout the Park, however, samples (n=627) were predominately associated with arroyos, roadways, or other areas where soil moisture availability is enhanced by disturbances. Predicted suitable habitats were not restricted to areas where current plants were observed

(see Chapter 3, Fig. 3-8). Additional predicted habitat occurred in semi-desert grasslands and shrublands. This distribution supports our results indicating clustered populations, and thus warrants an adaptive approach to sampling.

Russian Thistle

Evaluation of single-stage designs for Russian thistle indicated the Single-stage SRS-A was slightly more efficient than the two non-adaptive approaches evaluated (Fig 5-6). Likewise, the hit to occupancy ratio yielded slightly higher estimates for the Single-stage SRS-A approach than the non-adaptive approaches.

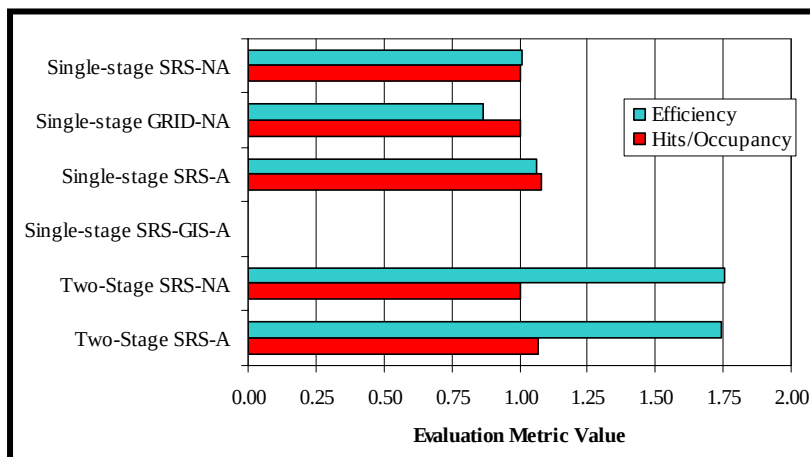


Figure 5-6 (Left). Efficiency and the hits to occupancy ratio for Russian thistle field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset.

There was virtually no difference in efficiency between the two-stage sampling designs evaluated. The Two-stage SRS-A approach yielded a slightly higher hit to occupancy ratio, indicating that an adaptive approach may be slightly more proficient than a non-adaptive approach across the entire study area.

Our results may be greatly influenced by our low sample size ($n=38$) for the simulations and the potential poor quality of our predictive habitat model (see Chapter 3).

Russian thistle is a habitat generalist, and can be found in a number of disturbed areas, grasslands, and desert communities. Our simulations suggest that Russian thistle is best sampled using adaptive procedures in suitable habitats. This conclusion is supported by this plant's reproductive approach. Russian thistle is a prolific seed producer that typically distributes seeds within 4.0 km (2.5 mi) of parent plant (Stallings et al. 1995). Thus, sampling strategies that focus near current populations or in suitable habitats would likely result in more robust estimates.

Johnson Grass

Single-stage analyses of Johnson grass occurrence locations indicated that the Single-stage SRS-NA approach was likely more efficient than the other single-stage designs

evaluated (Fig. 5-7). Conversely, the hit to occupancy ratio indicated the Single-stage SRS-A approach would be a proficient sampling design because a greater number of individuals would be detected using the Single-stage SRS-A approach. Our two-stage analyses indicated a similar trend, with the Two-stage SRS-NA design resulting in greater efficiency, and the Two-stage SRS-A design resulting in greater hit to occupancy ratio.

These results suggest that Johnson grass is not clustered in the Park or in suitable habitats. While our samples used for simulations ($n=147$) were fairly geographically grouped within areas of the Park, predicted suitable habitat occurred throughout the Park (see Chapter 3, Fig. 3-10). Johnson grass occurs in riparian areas or areas where soil moisture availability has been enhanced by disturbances or other factors. This plant can reproduce through seed production and rhizomes. Given this plant's life history and our simulation results, either a non-adaptive or adaptive sampling procedure would likely provide robust results. However, concentrating sampling efforts in suitable habitats would likely increase the number of individuals detected.

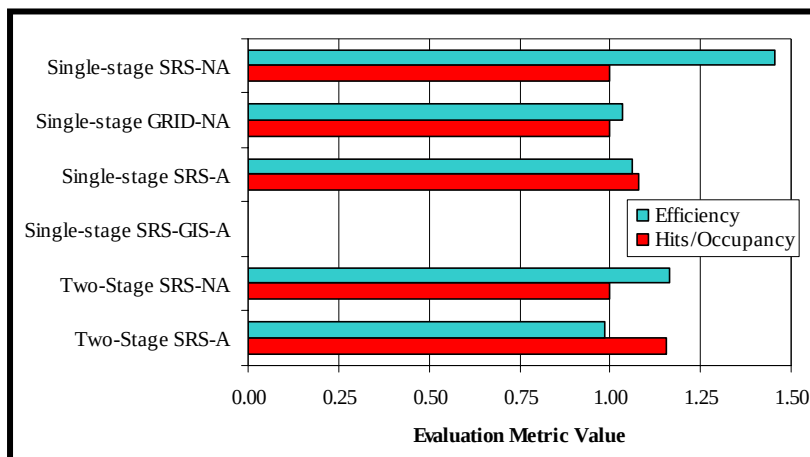


Figure 5-7 (Left). Efficiency and the hits to occupancy ratio for Johnson grass field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage SRS-GIS-A design could not be estimated for this dataset.

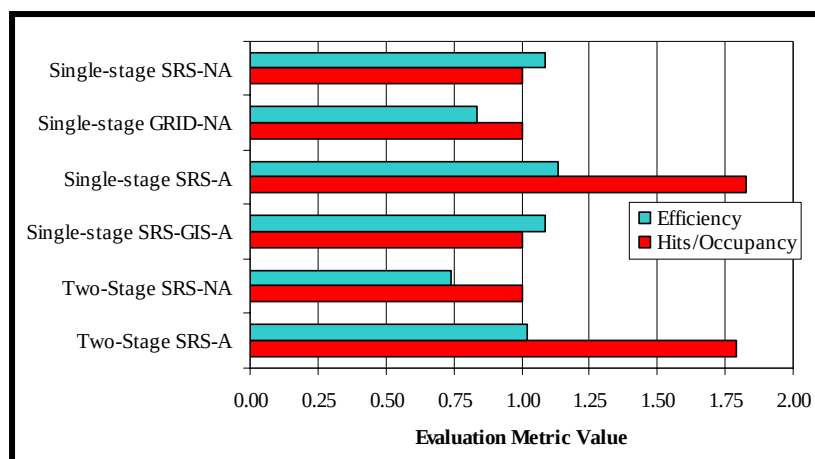


Figure 5-8 (Left). Efficiency and the hits to occupancy ratio for saltcedar field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006).

Saltcedar

Evaluation of single-stage designs for saltcedar identified the Single-stage SRS-A design with a higher efficiency value and a higher hit to occupancy ratio than the other single-stage designs (Fig. 5-8). The Single-stage SRS-GIS-A and the Single-stage GRID-NA yielded similar efficiency and hit to occupancy ratio estimates.

Likewise, the two-stage analyses indicated that the Two-stage SRS-A approach was more efficient than the non-adaptive approach. The Two-stage SRS-A design also yielded a substantially higher hit to occupancy ratio.

Both the single-stage and two-stage analyses suggest that saltcedar is clustered in the Park, and in suitable habitats. We found saltcedar in large and small clusters (depending on available moisture and infestation level) in waterways, riparian areas, and moist drainages. Although we had a fairly large sample ($n=225$) to run simulations, some of our samples were clustered (see Chapter 3, Fig. 3-11).

All Eight Species

We simulated sampling designs for all eight species together to investigate efficient sampling methods that could detect multiple species. Our results were similar for both

single- and two-stage designs; simple random sample adaptive approaches (Single-stage SRS-A and Two-stage SRS-A) yielded higher efficiency values and higher hit to occupancy ratios than non-adaptive approaches (Fig. 5-9). Due to limitations in the software in regards to our datasets, we were not able to simulate more complex designs that may have incorporated strata or probabilistic sampling procedures. However, our predictive modeling results (Chapter 3, Fig. 3-12) and our simulation results indicate that invasive species occurrences in BIBE are likely clustered in the Park and in suitable habitats. As such, adaptive sampling designs that focus near current populations or in suitable habitats would likely provide more efficient and robust estimators.

Summary Results and Discussion

We evaluated six sampling designs using simulations derived from known plant occurrences and potential distributions in BIBE. Each approach was identified as potentially being efficient for at least one of our invasive plant species (Table 5-3). Our results indicate that for single-stage designs, an adaptive simple random sampling (Single-stage SRS-A) procedure could provide reliable estimates for six (75%) of the species evaluated, and it would likely provide reliable estimates for all invasive plant species

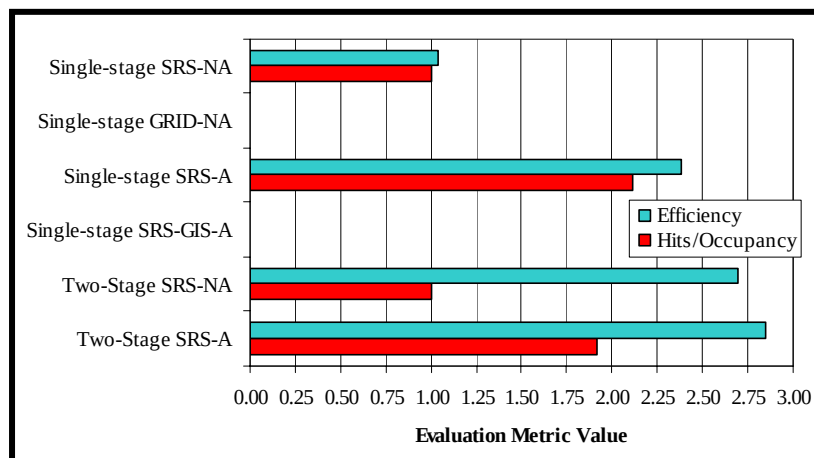


Figure 5-9 (Left). Efficiency and the hits to occupancy ratio for all eight target invasive plants combined field sampling designs evaluated in Big Bend National Park using program SAMPLE (Smith and Nichols 2006). The single-stage GRID-NA and SRS-GIS-A design could not be estimated for this dataset.

combined. The Single-stage SRS-GIS-A approach may provide efficient and reliable population estimates, especially when populations are clustered near high risk areas. Unfortunately, we were unable to evaluate every plant species using this approach. Sampling procedures should consider areas at risk of invasive plant incursion. Including areas at risk as a stratum or using the risk assessment to alter plant probabilities may increase the likelihood of detecting rare species and increase the utility of field sampling procedures for early detection protocols. The grid based, non-adaptive,

systematic sample (Single-stage-GRID-NA) approach would likely be an adequate design for Bermuda grass, especially because this species forms extensive stands of turf or lawn grass. Typically a non-adaptive, simple random sample would not be a practical sampling design for rare or spatially segregated populations (Smith et al. 2004, Edwards et al 2005, Philippi 2005, Smith and Nichols 2006). However, our results indicate that the Single-stage SRS-NA approach was the most efficient method for Russian thistle, and potentially Johnson grass, sampling.

Table 5-3. Summary of sampling designs that could potentially yield robust estimators for invasive plant populations in Big Bend National Park. Decisions were based on sampling design efficiency and the hit to occupancy ratio from simulations.

	Single-stage				Two-stage	
	SRS-NA	SRS-A	SRS-GIS-A	SRS-GRID-NA	SRS-NA	SRS-A
Giant Reed			*			*
Bermuda grass				*		*
Lehmann's lovegrass		*				*
Horehound		*			*	*
Buffelgrass		*				*
Russian thistle		*				*
Johnson grass	*	*			*	*
Saltcedar		*				*
All Species		*				*

Our analyses cannot evaluate the efficacy between single-stage and two-stage approaches. However, recent literature suggests advantages to two-stage sampling approaches in detecting invasive plant species (Salehi and Smith 2005, Smith and Nichols 2006). If considering a two-stage sampling design, our results indicate that an adaptive approach was more efficient for all species than a non-adaptive approach; although a non-adaptive approach would likely suffice for horehound and Johnson grass.

Limiting Factors

Sampling efficiency metrics are influenced by several factors, including: population distribution, sample size, sample unit size, per unit sample cost, and spatial distribution (Smith et al. 2004). Large samples in proportion to the population are associated with high efficiencies. However, these are not often achievable in real applications (Christman 1997, Smith et al. 2004). Efficiency of adaptive sampling is dependent on cluster size, favoring those species that form large clusters (Acharya et al. 2000). Smith et al. (2004) and Philippi (2005) found that adaptive sampling was efficient with small clusters and species that are rare (e.g. invasive species). Further, they found that efficiency declined with small, numerous clusters, and efficiency was dependent on sample size with large clusters of rare species.

We were unable to simulate complex probabilistic sampling designs. However, a stratified design with proportional, optimal or systematic allocation of samples may increase sampling efficiency. These designs can further be modified in an adaptive framework, using cluster sampling, grid approaches, or GIS datasets to further inform the sampling effort. For example, risk assessment information (e.g., models created in Chapter 4) could provide predetermined sampling categories for sampling strata, or help allocate

sampling effort to areas that are at higher risk of invasive plant incursion, or areas that may host sensitive biological or cultural entities.

Our simulation analyses were based on plant location data collected opportunistically, and thus are likely not a representative sample of BIBE. The degree this confounded our results are unknown. However, our results provide insight to sampling strategies that can be used to collect representative samples across the Park. Once new samples are collected, new simulations could be conducted to evaluate the bias associated with our results. Further, representative samples of the Park would better inform the predicted habitat models.

Our results are also limited by the spatial resolution of our simulations (90 m; 295 ft pixels). This large pixel size may provide overestimates of all evaluation metrics. Though some species (saltcedar) will readily be identified in an area the size of a 90 m (295 ft) pixel, many such as horehound may not be identified across large pixels. Additionally separate clusters within these pixels would have been grouped into effectively one large cluster. Future improvements to simulation software may allow for appropriate spatial resolution analyses.

Conclusions and Implications

Plant life histories and phenology affect species occupancy and detection rates, and thus the appropriate sampling strategy. Knowledge of species life histories (see Chapter 2) should be researched to guide sampling methodologies and insure in proper interpretation of sampling results (i.e., sampling is scale dependent). Species with limited habitat (e.g., giant reed, salt cedar) are confined to fewer areas than generalist species and given a larger study area will appear clustered. For example, our results indicated that giant reed was clustered with the BIBE

study area (two-stage analysis results), but populations may not be clustered within suitable habitats (single-stage analysis results).

From a practical standpoint, we created habitat models based on species occurrence locations and life history, and created models of areas at risk to invasion. While the predictive habitat models allow sampling effort to be concentrated in suitable habitats, the risk models allow for the distribution of samples across all of BIBE. When used in combination, they may provide valuable tools in allocating sampling effort. When non-inferential samples are desirable, these tools may correspond to and take the place of sampling designs in that they are meant to identify areas that the species either currently occupies or could occupy in the future. Analyses of these risk surfaces within the context of sampling are still necessary.

Sampling designs should be reflective of the needs of the agency. If the objectives of the sampling are to eradicate invasive species, a sampling design will likely not be the focus of effort, particularly a sampling design identified to provide statistical rigor. However, designs that have a greater probability of hits compared to occupancy have the best chance of detecting new species occurrences. The Heartland Inventory and Monitoring Network adopted a modified two-stage adaptive sampling design to monitor species (Smith and Nichols 2006). Recently this has been the preferred method for detection (Villella and Smith 2005, Salehi and Smith 2005) and was corroborated by our analyses for most species over the entire study area.

The two paradigms of basic research and management must be considered when designing a sampling scheme. Basic research focuses on infestation severity and sampling efficiency. Application and management

focuses on finding and eradicating these species from areas. These paradigms are related with sampling as the efficiency of detecting these species should increase and provide managers the logistical help in eradicating identified populations. This is confounded by the limited knowledge of these invasive species particularly within BIBE. All of our target invasive species rapidly colonize disturbed areas, outcompete native species, and alter natural processes that further promote invasive species invasions and establishment. Many of these plant species are well established. Thus, early detection for these species applies to newly colonized areas.

Tukey (1986) and Smith et al. (2004) stressed that sampling efforts need to be placed in context to practical efficiency. Sampling designs that may yield precise estimates but are too complex or costly to implement are not practical. When evaluating sampling efficiency, it is important to consider sample costs, including travel to and among quadrats, time required to take measurements (function of number and size of sample unit), and the variance associated with the measurements (Elzinga et al. 1998, Philippi 2005). Tukey (1986) equates practical efficiency as the sampling design's statistical efficiency multiplied by the probability that the method would be put into practice.

We initiated this research to evaluate the efficacy of using remotely sensed and GIS data as tools to support early detection of invasive plants in BIBE. Our proof-of-concept approach was to create spatially explicit models of potential invasive species habitats based on known plant occurrences in the park and to conduct a risk assessment that allows for prioritizing areas for field sampling, plant control, eradication or other conservation efforts. The common field sampling designs that we simulated should help provide insight to techniques that would

strengthen the knowledge base. Park and natural resource managers may have information on the presence of invasive plants in their management unit or near their management unit. This information is often gained through incidental or opportunistic observations. Few land managers have quantified data from planned surveys regarding species occurrences or where an invasive species does not occur. While opportunistic observations provide valuable insight and information for the early detection of invasive plants, there is a need to develop

early detection techniques to improve the likelihood of finding invasive species before they become a problem in an area. As such, we feel our approach is appropriate for many large parks or natural resource areas, and that it incorporates the data and tools that are available for most large parks or networks. We anticipate the information provided in this document equates to a high practical efficiency described by Tukey (1986) and will provide the tools necessary for useful predictions for invasive species management.



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Appendices

Appendix A. Invasive plants that have been recognized by the National Biological Information Infrastructure as invading or having the potential to invade the Central Southwest and Gulf Coast Region (<http://cswgcin.nbii.gov/>). Big Bend National Park occurs in this region.

Scientific Name	Common Name
<i>Acrotilon repens</i>	Russian knapweed
<i>Aegilops cylindrica</i>	Jointed goatgrass
<i>Agrostemma githago</i>	Cockle
<i>Agrostis semiverticillata</i>	Bentgrass
<i>Agrostis stolonifera</i>	Creeping bentgrass
<i>Ailanthus altissima</i>	Tree-of-heaven
<i>Albizia julibrissin</i>	Mimosa
<i>Alhagi maurorum</i>	Camelthorn
<i>Alternanthera philoxeroides</i>	Pig weed
<i>Alysicarpus vaginalis</i>	White moneywort
<i>Anagallis arvensis</i>	Scarlet pimpernel
<i>Arundo donax</i>	Giant reed
<i>Asparagus officinalis</i>	Asparagus
<i>Avena fatua</i>	Wild oat
<i>Bothriochloa ischaemum</i>	King ranch bluestem
<i>Briza minor</i>	Little quakinggrass
<i>Bromus catharticus</i>	Rescue brome
<i>Bromus racemosus</i>	Smooth brome
<i>Bromus rubens</i>	Foxtail brome
<i>Bromus secalinus</i>	Bromegrass
<i>Bromus tectorum</i>	Cheat grass
<i>Caesalpinia gilliesii</i>	Bird of paradise
<i>Capsella bursa-pastoris</i>	Shepardspurse
<i>Cardamine hirsuta</i>	Hairy bittercress
<i>Cardiospermum halicacabum</i>	Balloonvine
<i>Carduus nutans</i>	Musk thistle
<i>Carex planostachys</i>	Cedar sedge
<i>Cayratia japonica</i>	bushkiller
<i>Centaurea melitensis</i>	Malta starthistle
<i>Centaurea solstitialis</i>	Yellow star thistle
<i>Cerastium glomeratum</i>	Sticky chickweed
<i>Chenopodium album</i>	Lambsquarters
<i>Chenopodium ambrosioides</i>	Mexican tea
<i>Chenopodium murale</i>	Nettle-leaf goosefoot
<i>Ciclospermum leptophyllum</i>	Fir-leaved celery
<i>Cnicus benedictus</i>	Blessed thistle
<i>Codium fragile</i>	Dead man's fingers
<i>Colocasia esculenta</i>	Wild taro
<i>Convolvulus arvensis</i>	Morningglory
<i>Corispermum nitidum</i>	Neat bug seed
<i>Cryptocoryne beckettii</i>	Sri lankan water trumpet
<i>Cucumis melo</i>	Cantalope

Appendix A. Cont.

Scientific Name	Common Name
<i>Cuphea carthagenensis</i>	Colombian waxweed
<i>Cuscuta japonica</i>	Giant asian dodder
<i>Cyclospermum leptophyllum</i>	Marsh parsley
<i>Cynodon dactylon</i>	Bermudagrass
<i>Cyperus entrerianus</i>	Woodrush flatsedge
<i>Cyperus esculentus</i>	Yellow nutgrass
<i>Cyperus rotundus</i>	Chaguan humatag
<i>Dactyloctenium aegyptium</i>	Crowfoot grass
<i>Daucus carota</i>	Bird's nest
<i>Dichanthium annulatum</i>	Kleberg bluestem
<i>Dichanthium aristatum</i>	Angleton bluestem
<i>Digitaria ciliaris</i>	Southern crabgrass
<i>Digitaria ischaemum</i>	Small crabgrass
<i>Digitaria sanguinalis</i>	Large crabgrass
<i>Duchesnea indica</i>	India mockstrawberry
<i>Echinochloa colona</i>	Jungle rice
<i>Echinochloa crusgalli</i>	Barnyard-grass
<i>Eichhornia azurea</i>	Anchored water hyacinth
<i>Eichhornia crassipes</i>	Common water hyacinth
<i>Elaeagnus angustifolia</i>	Russian olive
<i>Eleusine indica</i>	Crowsfoot grass
<i>Elymus repens</i>	Quackgrass
<i>Eragrostis barrelieri</i>	Mediterranean lovegrass
<i>Eragrostis cilianensis</i>	Candy grass
<i>Eragrostis lehmanniana</i>	Lehmann's lovegrass
<i>Eremochloa ophiuroides</i>	Centipede grass
<i>Eruca sativa</i>	Arrugula
<i>Eruca vesicaria</i>	Rocketsalad
<i>Ficus carica</i>	Common fig
<i>Glinus lotoides</i>	Glinus
<i>Gymnodinium mikimotoi</i>	Red tide
<i>Heliotropium indicum</i>	India heliotrope
<i>Hemarthria altissima</i>	African jointgrass
<i>Hydrilla verticillata</i>	Water thyme
<i>Hypochaeris microcephala</i>	Smallhead catsear
<i>Imperata cylindrica</i>	Cogon grass
<i>Ipomoea aquatica</i>	Chinese waterspinach
<i>Ipomoea hederacea</i>	Entireleaf morningglory
<i>Ipomoea purpurea</i>	Common morning-glory
<i>Juncus bufonius</i>	toad rush
<i>Kummerowia striata</i>	Common lespedeza
<i>Lamium amplexicaule</i>	Common henbit
<i>Lantana camara</i>	Lantana
<i>Lespedeza striata</i>	Japanese lespedeza
<i>Leucaena leucocephala</i>	White leadtree
<i>Ligustrum japonicum</i>	Japanese privet

Appendix A. Cont.

Scientific Name	Common Name
Ligustrum sinense	Chinese privet
Lindernia crustacea	Malaysian false pimpernel
Lolium perenne	Italian ryegrass
Lolium temulentum	Darnel ryegrass
Lonicera japonica	Chinese honeysuckle
Lotus corniculatus	Birdfoot deervetch
Lycianthes asarifolia	Gingerleaf
Lygodium japonicum	Japanese climbing fern
Lygodium microphyllum	Old world climbing fern
Lythrum salicaria	Purple loosestrife
Marrubium vulgare	Horehound
Mazus pumilus	Japanese mazus
Medicago lupulina	Black medic
Medicago polymorpha	Bur clover
Medicago sativa	Alfalfa
Melaleuca quinquenervia	Bottle brush tree
Melia azedarach	Chinaberry
Melilotus albus	White sweetclover
Melilotus indicus	Annual yellow sweetclover
Melilotus officinalis	Yellow sweet-clover
Mentha aquatica	Water mint
Mentha spicata	Spearmint
Microstegium vimineum	Japanese stiltgrass
Mirabilis jalapa	Common four o'clock
Morus alba	Mulberry
Murdannia nudiflora	Nakedstem dewflower
Myriophyllum spicatum	Eurasian watermilfoil
Nerium oleander	Oleander
Nicotiana glauca	Tree tobacco
Oplismenus hirtellus	Bristle basketgrass
Orobanche ramosa	Branched broomrape
Oryza sativa	Rice
Oxalis debilis	Pink woodsorrel
Panicum antidotale	Blue panic grass
Panicum lanuginosum	Panic grass
Panicum repens	Creeping panic
Parkinsonia aculeata	Jerusalem thorn
Paspalum dilatatum	Dallas grass
Paspalum notatum	Bahiagrass
Paspalum urvillei	Vaseygrass
Peganum harmala	African rue
Pennisetum ciliare	Buffelgrass
Perilla frutescens	Beefsteak
Petunia parviflora	Wild petunia
Phragmites australis	Common reed
Phyllanthus urinaria	Chamber bitter

Appendix A. Cont.

Scientific Name	Common Name
<i>Phyllostachys aurea</i>	Golden bamboo
<i>Physalis philadelphica</i>	Mexican ground-cherry
<i>Pistia stratiotes</i>	Water lettuce
<i>Plantago lanceolata</i>	Buckhorn plantain
<i>Poa annua</i>	Annual bluegrass
<i>Polygonum caespitosum</i>	Oriental ladythumb
<i>Polygonum convolvulus</i>	Black bindweed
<i>Polygonum cuspidatum</i>	Japanese knotweed
<i>Polygonum lapathifolium</i>	Curlytop knotweed
<i>Polygonum persicaria</i>	Spotted knotweed
<i>Polypogon monspeliensis</i>	Annual rabbitsfoot grass
<i>Polypogon viridis</i>	Beardless rabbitsfoot grass
<i>Portulaca oleracea</i>	Garden purslane
<i>Pueraria montana</i>	Japanese arrowroot
<i>Ranunculus muricatus</i>	Spinyfruit buttercup
<i>Ranunculus parviflorus</i>	Smallflower buttercup
<i>Ranunculus sardous</i>	Hairy buttercup
<i>Rapistrum rugosum</i>	Annual bastardcabbage
<i>Richardia brasiliensis</i>	Tropical mexican clover
<i>Ricinus communis</i>	Mgaliya
<i>Rosa bracteata</i>	Macartney rose
<i>Rosa multiflora</i>	Multiflora rose
<i>Rottboellia cochinchinensis</i>	Itchgrass
<i>Rumex crispus</i>	Curley dock
<i>Salsola kali</i>	Russian thistle
<i>Salvinia minima</i>	Common salvinia
<i>Salvinia molesta</i>	African payal
<i>Sanvitalia procumbens</i>	Mexican creeping zinnia
<i>Schedonorus phoenix</i>	Tall fescue
<i>Schinus terebinthifolius</i>	Brazil peppertree
<i>Senna obtusifolia</i>	Java-bean
<i>Senna occidentalis</i>	Coffee senna
<i>Setaria adhaerens</i>	Burr bristlegrass
<i>Setaria faberi</i>	Chinese foxtail
<i>Setaria viridis</i>	Wild millet
<i>Sherardia arvensis</i>	Blue field-madder
<i>Silybum marianum</i>	Milk thistle
<i>Sisymbrium irio</i>	London rocket
<i>Solanum pseudocapsicum</i>	Jerusalem cherry
<i>Solanum sisymbriifolium</i>	Sticky nightshade
<i>Solanum viarum</i>	Tropical soda apple
<i>Soliva sessilis</i>	Field burrweed
<i>Sonchus asper</i>	Perennial sowthistle
<i>Sonchus oleraceus</i>	Annual sowthistle
<i>Sorghum bicolor</i>	Sorghum
<i>Sorghum halepense</i>	Aleppo milletgrass

Appendix A. Cont.

Scientific Name	Common Name
<i>Sphenoclea zeylanica</i>	Chickenspike
<i>Spirodela punctata</i>	Dotted duckmeat,
<i>Sporobolus indicus</i>	Rattail smutgrass
<i>Stellaria media</i>	Chickweed
<i>Stellaria parva</i>	Pygmy starwort
<i>Tamarix aphylla</i>	Athel tamarisk
<i>Tamarix chinensis</i>	China tamarisk,
<i>Tamarix ramosissima</i>	Saltcedar
<i>Taraxacum officinale</i>	Common dandelion
<i>Tragus berteronianus</i>	Spiked burr grass
<i>Triadica sebifera</i>	Chinese tallow
<i>Tribulus terrestris</i>	Puncture vine
<i>Trifolium dubium</i>	Hop clover
<i>Trifolium repens</i>	Dutch clover
<i>Trifolium resupinatum</i>	Persian clover
<i>Ulmus parvifolia</i>	Chinese elm
<i>Ulmus pumila</i>	Siberian elm
<i>Urochloa maxima</i>	Guineagrass
<i>Verbena brasiliensis</i>	Brazilian vervain
<i>Veronica arvensis</i>	Common speedwell
<i>Wisteria sinensis</i>	Chinese wisteria

Appendix B. General soil units in Big Bend National Park (Cochran and Rives 1985).

General Soil Unit	Slope	Location	General Land Cover	Soils	Percent of BIBE
Chamberino-Chilicotal-Upton	1-15%	Uplands	lechuguilla, creosotebush, whitethorn acacia, ocotillo, and yucca	Very gravelly and gravelly	35.0%
Lajitas-Rock Outcrop	3-60%	Igneous hills and mountains	Rocky outcrops, lechuguilla, pricklypear, ocotillo, threeawns, chino grama, and sideoats grama	Very cobbly soils and rock outcrops	26.0%
Lozier-Rock Outcrop-Marical	10-90%	Limestone hills and mountains	Rocky outcrops, creosotebush, lechuguilla, cacti, threeawns, chino grama, and sideoats grama	Very cobbly and very channery	22.0%
Solis-Rock Outcrop Vieja	1-16%	Uplands and valleys	Sandstone exposures and barren areas, scattered cacti, shrubs, and annuals	Loamy and very gravelly clayey soils	8.0%
Tornillo-Pajarito	0-3%	Uplands	Sparse creosotebush, lechuguilla, and mariola	Loamy	4.0%
Brewster-Rock Outcrops-Hurds	3-90%	Igneous hills and mountains (Chisos Mountains)	Pinyon pine, alligator juniper, drooping juniper, gray oak, Emory oak, Grave's oak and Chisos oak.	Very cobbly and very gravelly	3.0%
Glendale-Harkey	0-2%	Flood plains	Mesquite, saltcedar, and giant reed	Loamy	1.5%
Puerta-Madrone	8-60%	Igneous mountains (Chisos Mountains)	Pinyon pine and oaks	Very gravelly	0.5%

Appendix C. Coordinates (UTM, NAD 83, Zone 13) of invasive plant occurrences in Big Bend National Park used to model their potential distribution within the Park.

Giant Reed (*Arundo donax*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
693719	3226042	694916	3229543	656800	3212934	642217	3224830
692370	3223860	643789	3223034	658755	3211669	666118	3209855
687917	3219932	643690	3222621	666035	3209892	664820	3211292
697200	3230351	643689	3224073	667098	3209018	664668	3211015
701058	3231395	677030	3207484	664675	3210973	664578	3210764
702421	3231880	684458	3214381	662573	3212344	662727	3212458
702828	3232316	652502	3215071	636500	3225746	667094	3209013
694761	3229307	654566	3213321	634296	3226972		

Bermuda grass (*Cynodon dactylon*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
693719	3226042	636500	3225746	664216	3239914	653350	3229776
688562	3222636	634296	3226972	664672	3239540	653233	3231751
697200	3230351	642217	3224830	666035	3209892	657666	3233079
694244	3229226	647075	3225591	667098	3209018	658102	3232942
694916	3229543	647296	3225679	662573	3212344	658473	3237823
692810	3237831	644158	3231762	662491	3214058	658430	3239592
690842	3240497	668912	3275463	636500	3225746	656863	3244080
691435	3244017	668798	3275186	634296	3226972	635930	3227055
685479	3254672	670701	3275768	642217	3224830	635960	3226681
685684	3254887	672374	3274544	647075	3225591	636113	3226663
679184	3228840	668415	3273209	647296	3225679	636117	3226511
698202	3229814	664519	3271428	644158	3231762	636108	3226441
681097	3239612	677475	3280634	668912	3275463	636169	3226410
665090	3239100	680757	3277082	668798	3275186	636657	3225979
664855	3239499	680058	3276007	670701	3275768	636997	3225678
664548	3240018	679182	3275198	672374	3274544	638030	3225857
662241	3240362	676697	3277818	668415	3273209	638265	3225914
663107	3239713	674205	3275632	664519	3271428	639054	3226611
664216	3239914	669803	3273897	677475	3280634	665228	3239203
658438	3239587	666118	3209855	680757	3277082	664904	3238913
649164	3226355	664820	3211292	680058	3276007	664715	3239601
644526	3223702	664668	3211015	679182	3275198	698259	3230836
642936	3235438	664578	3210764	676697	3277818	656000	3244218
641931	3232807	662727	3212458	674205	3275632	678219	3234907
635648	3230555	658438	3239587	669803	3273897	643811	3224086
643789	3223034	649164	3226355	666118	3209855	674197	3275623
643690	3222621	644526	3223702	664820	3211292	697986	3229919
643661	3222346	642936	3235438	664668	3211015	697898	3230056
643689	3224073	641931	3232807	664578	3210764	697216	3230267
669669	3247393	635648	3230555	662727	3212458	697170	3230369
669383	3247032	643789	3223034	667094	3209013	673957	3245793
664188	3254031	643690	3222621	679183	3228766	674049	3245748
677030	3207484	643661	3222346	702211	3232074	669159	3246161

Appendix C. Cont.

Bermuda grass (*Cynodon dactylon*) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
652502	3215071	669669	3247393	680549	3255632	673675	3244582
656800	3212934	669383	3247032	668183	3273203	673603	3244808
684458	3214381	643689	3224073	679986	3253125	673908	3245286
658755	3211669	664188	3254031	677214	3280789	673730	3245363
660936	3214774	677030	3207484	645650	3225500	662491	3214058
664672	3239540	684458	3214381	668529	3247099	660936	3214774
666035	3209892	652502	3215071	668361	3247272	674264	3275647
667098	3209018	656800	3212934	662198	3246450		
662573	3212344	658755	3211669	661806	3246334		

Lehmann's Lovegrass (*Eragrostis lehmanniana*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
664548	3240018	669562	3246102	657815	3241820	645159	3242063
664216	3239914	676910	3281565	658380	3240629	644865	3242312
666618	3248847	679852	3277951	658438	3240619	646016	3241460
663028	3246510	681808	3264836	658484	3240182	646145	3241517
659497	3246146	681139	3263079	674260	3245711	646139	3241466
658589	3245978	682023	3265730	674338	3245669	646367	3241553
662027	3240388	679798	3277271	674565	3245519	646811	3241928
660282	3240538	651418	3227581	675044	3245136	647534	3242368
659679	3240140	668952	3246734	675265	3244885	648100	3243391
658530	3233068	665848	3248940	675337	3244795	652142	3245028
655178	3235063	665755	3248864	675559	3244478	655398	3244260
644526	3223702	662602	3246445	676169	3243089	674169	3245790
654990	3244534	662367	3246463	667043	3240691	669276	3246350
647826	3243030	662318	3246467	666938	3240502	669225	3246331
680702	3259541	662242	3246454	666513	3239578	645446	3241430
673006	3253716	661144	3246107	666507	3239628	676950	3241353
672682	3252897	657342	3245107	666318	3239726	677067	3241244
671925	3249385	657515	3245222	666183	3239787	677496	3240837
671510	3248693	675425	3275904	666071	3239743	677628	3240715
669669	3247393	652690	3229115	665211	3239122	677734	3240615
664521	3254874	652737	3229089	665059	3239011	677849	3250396
664417	3254473	652995	3229206	665140	3239217	678160	3280118
664188	3254031	653340	3229722	664921	3239969	674099	3245688
666025	3251328	653360	3229950	664746	3239527	674012	3245771
670607	3245401	653327	3230299	680692	3262665	669140	3246124
670179	3244957	653229	3230457	680665	3262579	668916	3243970
669581	3244884	653387	3231228	680685	3261799	669037	3244355
670222	3245276	653293	3231687	680596	3260362	668738	3243474
670272	3245518	653300	3231865	680526	3257390	668348	3242073
670646	3245952	653343	3231972	680423	3256739	668305	3241977
655897	3244849	653806	3232985	680370	3256089	667761	3241625
656363	3243360	654018	3233240	680317	3255950	668420	3242131
661484	3246207	656003	3233606	680310	3255764	668459	3242627

Appendix C. Cont.

Lehmann's Lovegrass (*Eragrostis lehmanniana*) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
660116	3245915	656590	3233569	680357	3255780	668512	3242742
663599	3247146	656854	3233555	680357	3255478	668550	3242848
665212	3248391	656925	3233505	680360	3254403	668671	3243084
667217	3248157	657170	3233374	680038	3253717	668719	3243238
674663	3251500	657793	3233082	680018	3253601	668825	3243503
680571	3258222	657663	3234652	679992	3253129	668887	3243590
668038	3266517	657787	3234994	679955	3252897	669051	3244148
666111	3266863	657824	3234980	679988	3252837	669118	3244384
680644	3261430	657840	3235011	679603	3251961	669147	3244456
676512	3276185	657864	3235078	679600	3251882	669252	3246208
680347	3255636	657827	3235162	678771	3250853	669814	3247475
673573	3246020	657988	3235950	678021	3250468	674225	3245585
673134	3246143	658028	3236090	677460	3250180	673831	3244744
672989	3246164	658125	3236432	675536	3248856	673706	3244641
672927	3246171	658206	3236647	675443	3248793	673670	3244589
672257	3246240	658166	3232928	675247	3248600	673778	3244606
672098	3246243	658110	3237069	675187	3248361	673791	3244639
671949	3246233	658427	3237763	675187	3248222	673814	3244733
671279	3246160	658470	3237783	674736	3246381	673836	3244266
670913	3246109	658404	3238080	674208	3245727	673608	3244749
670744	3246088	658434	3238087	674238	3245770	673637	3244808
670415	3246064	658483	3239549	684300	3237588	673580	3244806
669908	3246071	658444	3239473	681995	3238820	673796	3244893
670101	3246005	657006	3244676	690153	3233414	673701	3244914
670664	3246043	657023	3244590	689458	3233814	673999	3245375
670903	3246064	656987	3244465	688273	3234650	673711	3245287
673428	3246008	657042	3244468	687496	3235191	673711	3245219
674136	3245725	657528	3243255	686316	3236025	673747	3245162
669424	3246230	657662	3242619	645417	3241613	674113	3245039
669307	3246326	657659	3242550	645378	3241886	674082	3245081
						673965	3245060

Horehound (*Marrubium vulgare*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
664855	3239499	673787	3244623	674027	3245062	665044	3239114
664548	3240018	673694	3244595	664688	3239670	665065	3239142
672147	3249787	673843	3244304	664939	3239769	665037	3239133
669383	3247032	673538	3244681	664986	3239763	665070	3239051
664672	3239540	673599	3244895	664920	3239860	665135	3239077
663756	3271593	673690	3244954	664883	3239860		
664688	3239510	674137	3245016	664848	3239913		

Appendix C. Cont.

Buffelgrass (*Pennisetum ciliare*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
696940	3231888	656737	3233585	673768	3244666	690423	3233119
692901	3237006	657093	3233367	673771	3244667	690424	3233118
692810	3237831	657083	3233324	673770	3244666	690426	3233119
679184	3228840	656948	3233190	673770	3244666	690425	3233118
692141	3225509	657344	3233072	673771	3244666	690425	3233115
698313	3231548	657549	3233082	673772	3244667	690445	3233124
695905	3232036	657669	3234685	673771	3244668	690447	3233121
659497	3246146	658028	3236315	673770	3244667	690450	3233121
649164	3226355	658427	3237813	673771	3244670	690448	3233119
644526	3223702	658427	3237899	673770	3244669	690451	3233086
644480	3240027	658467	3237743	673770	3244669	690474	3233066
642840	3238053	658460	3240035	673771	3244671	690472	3233067
642936	3235438	656886	3244077	673770	3244667	690491	3233033
642871	3233815	656899	3243953	673768	3244669	690493	3233034
643081	3233103	656860	3243959	673764	3244672	690491	3233031
641436	3232265	657258	3243428	673767	3244673	690490	3233032
641931	3232807	657545	3243314	673768	3244674	690510	3233035
635708	3230803	657678	3242978	673764	3244671	690518	3233031
635648	3230555	657662	3242684	673764	3244669	690512	3233029
677487	3226669	657642	3242374	673763	3244674	690526	3233016
679770	3218514	657642	3242270	673774	3244658	690529	3233011
681750	3217867	658477	3240097	673775	3244660	690526	3233010
684139	3217158	635938	3227661	673773	3244662	690565	3232989
644803	3225007	635847	3227530	673736	3244656	690567	3232985
647390	3223929	635851	3227474	673749	3244643	690564	3232985
647256	3220171	635912	3227278	673744	3244639	690544	3233005
660156	3213096	635956	3227282	673746	3244639	690541	3233006
660936	3214774	635943	3227129	673747	3244643	690538	3233007
664672	3239540	636087	3226637	679578	3237928	690538	3233008
686112	3220784	636108	3226550	679578	3237928	677282	3226455
684423	3216999	636161	3226254	679578	3237927	677200	3226340
673106	3215062	636261	3226227	679571	3237930	674533	3224197
670725	3213643	636422	3226049	679569	3237927	674536	3224185
667876	3211500	637076	3225700	679570	3237927	674557	3224081
668967	3212369	637651	3225905	679571	3237928	674555	3224080
669177	3213212	638108	3225892	679572	3237928	674554	3224081
666481	3213208	638540	3226027	679572	3237929	674554	3224078
666088	3215215	639272	3226968	679570	3237927	674552	3224081
663911	3214728	639921	3226899	679572	3237928	674552	3224083
636500	3225746	640034	3226794	679573	3237927	674550	3224080
634296	3226972	640239	3226737	679579	3237932	674563	3224077
635888	3227643	640431	3226742	679573	3237929	674563	3224080
638118	3225923	640792	3226406	679573	3237929	674563	3224079
643326	3225139	640849	3226302	679572	3237928	674565	3224078
641723	3227427	640954	3226275	664546	3239641	674567	3224075

Appendix C. Cont.

Buffelgrass (*Pennisetum ciliare*) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
642217	3224830	641128	3225997	664550	3239638	674566	3224080
638552	3226200	641045	3226145	664547	3239638	674567	3224079
647075	3225591	641564	3226005	664549	3239633	674568	3224080
647296	3225679	641629	3225835	664550	3239632	674443	3223379
642832	3233335	641764	3225413	664553	3239631	675510	3224930
642887	3234776	641947	3225291	664552	3239630	677259	3226416
642659	3237292	642013	3225217	664551	3239630	677798	3230645
697246	3231242	642509	3224912	664549	3239629	677503	3227393
693924	3231447	642544	3224912	664552	3239628	677215	3226400
691567	3231455	642614	3224864	664556	3239623	677240	3226412
691384	3232396	642653	3224920	664557	3239626	677250	3226404
688339	3235127	642792	3224898	664557	3239630	677240	3226422
680056	3278063	643019	3224842	664556	3239630	677228	3226425
679600	3277834	643089	3224724	664555	3239630	677239	3226430
675738	3276992	644087	3224010	664556	3239632	677233	3226378
677557	3276601	644217	3223879	664554	3239633	677239	3226379
693202	3231105	680665	3262552	664555	3239633	677216	3226372
692930	3231926	680596	3260409	664554	3239634	689551	3233757
692433	3232489	680692	3258551	664553	3239634	688815	3234304
692262	3233744	680615	3257512	664556	3239641	688815	3234305
691810	3234470	680562	3257260	664555	3239641	688813	3234305
677255	3226430	680360	3256464	664557	3239642	688813	3234304
667742	3211344	680360	3254513	664557	3239643	688811	3234306
670732	3213621	680068	3253979	664557	3239641	688810	3234308
669109	3213202	697538	3231331	664560	3239641	688809	3234309
698309	3231550	697492	3231418	664560	3239642	688808	3234308
698406	3231539	697251	3231800	664559	3239643	688811	3234305
701135	3231735	697141	3231875	664570	3239620	688818	3234304
701703	3231668	696782	3232062	664572	3239619	688819	3234303
701791	3231637	696782	3232095	664576	3239616	688820	3234301
701818	3231280	696487	3232093	664578	3239623	688820	3234303
701944	3231625	694881	3231390	664574	3239639	658512	3245899
702120	3232089	694583	3231364	664581	3239639	673862	3244361
702211	3232074	694534	3231323	664580	3239638	657921	3245180
676167	3250977	692159	3230856	664580	3239642	657882	3245196
680346	3255529	692139	3230972	664584	3239643	657847	3245210
680345	3255716	691841	3231264	664584	3239642	657531	3245254
644595	3224083	691685	3231546	664585	3239640	657454	3245212
644579	3224142	691490	3231787	664585	3239641	657373	3245132
644675	3224676	691474	3231905	664587	3239640	657357	3245106
644919	3224811	691382	3232044	664599	3239613	657343	3245082
645048	3224852	691133	3232459	664601	3239613	657327	3245066
645092	3224815	690692	3233003	664619	3239580	657306	3245020
645314	3224984	689753	3233660	664617	3239577	657296	3245007
645623	3225417	689561	3233796	664607	3239592	657071	3244812

Appendix C. Cont.

Buffelgrass (*Pennistenum ciliare*) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
646200	3225664	645411	3241673	664607	3239592	657022	3244779
645836	3225707	689271	3233939	664606	3239593	657043	3244792
646342	3225627	645366	3241919	664603	3239593	656899	3244672
646370	3225454	646355	3241604	664603	3239591	656879	3244657
646777	3225204	647033	3242081	664597	3239593	656832	3244632
646888	3225241	652558	3245286	673764	3244668	656718	3244567
647083	3225478	653122	3245406	673764	3244666	656707	3244561
648806	3226797	653764	3245552	673760	3244669	656680	3244555
648784	3226747	653943	3245510	673761	3244667	656555	3244532
649108	3226401	654156	3245430	673763	3244667	656528	3244526
649164	3226315	656810	3244563	673766	3244669	656504	3244522
649216	3226340	656780	3244629	673765	3244669	659522	3246139
649315	3226191	656651	3244575	673768	3244667	673862	3244361
650084	3226049	654792	3244461	673767	3244668	673829	3244356
650053	3226087	654540	3244710	673763	3244668	659558	3246128
650142	3226210	679121	3228994	673765	3244667	659097	3246088
650673	3226587	635789	3227453	673765	3244667	656142	3244271
651019	3227078	693647	3230559	673766	3244667	655745	3244226
651313	3227383	693468	3230638	673767	3244667	673829	3244356
661451	3246254	693089	3230748	673767	3244667	673792	3244596
659792	3245987	702167	3232060	673766	3244666	673787	3244882
659743	3246063	702392	3231970	673767	3244666	673626	3244857
659627	3246058	702599	3232017	673766	3244664	673692	3244926
659352	3246129	698116	3229986	673766	3244663	673705	3244935
659032	3246045	697606	3230334	673764	3244662	673738	3244929
658569	3245871	697138	3230334	673761	3244665	673749	3244935
658258	3245236	697565	3231326	673760	3244664	673765	3244925
657947	3245138	658444	3245860	673760	3244661	679031	3229109
657618	3245240	691315	3232043	673754	3244661	677669	3227458
657320	3244964	698801	3229576	673751	3244657	677241	3226416
657071	3244751	699291	3229807	673748	3244659	673771	3244923
657004	3244782	673658	3244842	673747	3244657	673773	3244922
676420	3276188	673635	3244988	673746	3244653	673680	3244923
697600	3231322	673601	3245024	673748	3244655	673763	3244918
697770	3231358	673606	3245013	690091	3233465	673785	3244913
698097	3231540	673605	3245015	689845	3233579	673606	3245010
698223	3231522	673649	3244995	689189	3234001	673611	3245019
698932	3231196	673864	3244386	689202	3234091	652567	3228982
699064	3231214	676954	3237590	689226	3233975	652887	3229189
699402	3231217	676962	3237594	689553	3233734	653259	3230373
699869	3231136	676970	3237595	689562	3233730	653816	3232770
699947	3231145	673771	3244651	689572	3233709	654935	3233642
700016	3231214	673753	3244654	689571	3233711	655888	3233602
700292	3231184	673754	3244655	689576	3233706	657015	3233510
700588	3231355	673755	3244655	689592	3233761	657740	3233073

Appendix C. Cont.

Buffelgrass (*Pennisetum ciliare*) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
700974	3231543	673756	3244657	689574	3233756	658429	3238041
700693	3231397	673755	3244656	687579	3235202	658436	3237718
701016	3231564	673762	3244660	689164	3234061	657676	3242215
701750	3231657	673763	3244657	689925	3233554	657677	3242570
701819	3231648	673765	3244659	689927	3233552	657694	3242763
701950	3231780	673764	3244658	689973	3233456	657447	3243351
652230	3228582	673763	3244661	690087	3233479	656897	3244048
652824	3229199	673765	3244661	690089	3233466	688811	3234308
653018	3229216	673766	3244660	690100	3233475	689652	3233703
653347	3229739	673766	3244660	690092	3233478	689755	3233642
653290	3231590	673767	3244659	690092	3233477	690426	3233319
653233	3231785	673767	3244660	690091	3233478	690274	3233493
653283	3231781	673769	3244660	690115	3233478	673645	3244836
653776	3232781	673767	3244660	690355	3233182	673767	3244922
653843	3232972	673770	3244661	690358	3233182	673782	3244888
653967	3233173	673770	3244664	690379	3233170	673753	3244896
654933	3233673	673767	3244663	690380	3233168	673768	3244892
655188	3233673	673766	3244663	690422	3233124	654900	3233628
655637	3233572	673765	3244656	690420	3233123	673674	3244823
655855	3233612	673764	3244664	690421	3233121	654760	3227002
656127	3233622	673766	3244666	690425	3233121	673674	3244830
656603	3233619	673769	3244666	690423	3233118		

Russian Thistle (*Salsola kali*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
644150	3223548	672372	3274928	680073	3276533	676530	3276189
647256	3220171	672374	3274544	675738	3276992	679866	3278215
647733	3220102	668415	3273209	677557	3276601	660161	3246165
652502	3215071	662311	3269577	676512	3276185	664633	3271463
654566	3213321	663765	3271321	676697	3277818	665356	3239440
664672	3239540	663728	3271492	674205	3275632	665167	3239217
642887	3234776	664519	3271428	674307	3274274	655559	3244254
668912	3275463	664588	3271791	670059	3273791	668198	3273235
668934	3275280	678558	3279736	669803	3273897		
669367	3275439	679600	3277834	668130	3273340		

Appendix C. Cont.

Johnson Grass (*Sorghum halepense*)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
678257	3234919	670570	3273395	658072	3236261	680386	3256544
698202	3229814	672768	3274299	658112	3236372	680320	3255926
658589	3245978	672692	3274360	658075	3236463	680313	3255737
635648	3230555	679684	3278674	658146	3233626	680307	3255641
643689	3224073	679698	3278538	656876	3244054	680337	3255177
680702	3259541	679839	3277829	656944	3243689	680675	3255887
677030	3207484	679784	3278320	657773	3241986	680413	3254977
684458	3214381	679602	3279015	658168	3241105	680446	3254871
636500	3225746	676766	3281952	636527	3226018	693975	3231236
634296	3226972	681487	3264278	637921	3225831	653662	3245555
693924	3231447	681376	3263984	638291	3225927	678115	3234816
680571	3258222	681839	3265172	638383	3225914	678193	3234889
671379	3275218	681760	3267929	666567	3239534	664284	3271889
672374	3274544	680456	3272842	665970	3239703	668226	3273301
668415	3273209	679798	3276635	665110	3239200	667767	3273267
667826	3273195	650109	3226120	665282	3239645	681163	3258496
662494	3268680	649164	3226364	665120	3239774	680259	3255398
662311	3269577	650667	3226716	665053	3239763	698216	3231579
663180	3269124	668552	3247077	665053	3239655	680592	3258892
663765	3271321	663656	3247157	664874	3239581	670549	3273435
663800	3271685	663563	3247054	664688	3239723	696871	3230401
663239	3271324	675530	3275852	664833	3239473	696973	3230393
664519	3271428	674736	3275713	664897	3239733	643737	3224137
680056	3278063	674264	3275611	665059	3239922	644142	3223921
679600	3277834	674238	3275615	665329	3239507	644175	3223805
680757	3277082	672728	3274261	665302	3239372	669288	3246377
680058	3276007	672692	3274223	665336	3239338	652899	3245512
680073	3276533	670047	3273770	680605	3260542	680485	3261178
679182	3275198	670410	3273391	680718	3259855	682183	3265932
677557	3276601	670028	3273470	680632	3258647	698648	3229704
674205	3275632	668398	3273101	680559	3258206	674303	3245599
674307	3274274	667794	3273034	680559	3258375	673705	3244937
670059	3273791	697782	3231319	680509	3258571	665021	3239150
692262	3233744	698016	3231456	680529	3257751		
676167	3250977	653830	3233015	680556	3257154		
680345	3255602	657800	3235037	680529	3256985		
680329	3255642	657830	3235208	680493	3257134		
667674	3273170	657934	3235752	680433	3256849		

Appendix C. Cont.

Saltcedar (*Tamarix* spp.)

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
693719	3226042	650105	3226056	657126	3245532	653015	3247429
692370	3223860	650068	3226136	657120	3245600	652969	3247371
687917	3219932	650670	3226642	657086	3245714	652958	3247324
702421	3231880	661655	3246227	657053	3245734	652921	3247252
702828	3232316	657910	3241774	657033	3245763	652887	3247221
694244	3229226	635982	3227086	657032	3245762	652773	3247199
693924	3229376	636082	3226689	656937	3245779	652697	3247255
694761	3229307	636967	3225744	656877	3245775	652603	3247342
685479	3254672	638256	3225970	656733	3245832	652525	3247412
642871	3233815	638331	3225975	656456	3245978	652405	3247433
641436	3232265	638527	3225953	656454	3245928	652354	3247375
641931	3232807	643233	3224633	656397	3245950	652261	3247305
635648	3230555	643280	3224554	656381	3245959	652091	3247344
635829	3228371	679142	3228900	656264	3246000	651907	3247488
643789	3223034	679933	3238920	656228	3246039	651947	3247665
643690	3222621	679028	3276726	656194	3246060	651151	3247716
643661	3222346	698719	3229595	656074	3246262	652034	3247690
644150	3223548	698818	3229577	656050	3246299	652111	3247798
643689	3224073	666640	3230061	656028	3246322	652285	3247904
677921	3250347	667130	3230422	655972	3246320	652410	3248145
679008	3250728	667164	3230438	655883	3246307	652797	3248091
677030	3207484	667703	3230100	655849	3246284	653275	3248788
674145	3218381	667918	3230024	655530	3246415	653294	3248844
684458	3214381	667851	3230133	655474	3246394	653430	3248927
647256	3220171	667739	3230146	655441	3246382	653482	3248954
647733	3220102	660517	3247226	655452	3246380	653550	3249181
652247	3215582	660518	3247288	655393	3246370	653634	3249239
652502	3215071	660527	3247312	655361	3246418	653706	3249391
654566	3213321	660526	3247310	655309	3246465	653757	3249444
656800	3212934	660520	3247300	655271	3246484	653872	3249462
658755	3211669	660536	3247322	655224	3246402	653992	3249306
666035	3209892	660456	3247286	654825	3246445	653997	3249204
667098	3209018	649583	3232116	654807	3246524	654031	3249177
664675	3210973	653401	3227094	654793	3246588	654242	3249178
662573	3212344	664311	3225351	654765	3246641	654305	3249145
662491	3214058	664304	3225373	654680	3246633	654461	3249219
636500	3225746	664329	3225709	654585	3246578	654562	3249294
634296	3226972	664346	3225816	654485	3246514	654710	3249370
642217	3224830	664448	3225991	654410	3246547	654931	3249329
647296	3225679	664426	3225920	654365	3246572	655027	3249249
681637	3236485	664423	3225851	654309	3246605	655074	3249208
692930	3231926	664294	3225488	654200	3246687	655165	3249129
692433	3232489	650670	3226693	654170	3246786	655200	3249097
692262	3233744	649752	3226074	654092	3246827	655811	3246215
691810	3234470	649756	3226064	654038	3246853	655773	3246099

Appendix C. Cont.

Saltcedar (*Tamarix* spp.) cont.

UTM E	UTM N	UTM E	UTM N	UTM E	UTM N	UTM E	UTM N
662562	3214090	649159	3226370	653948	3246919	656167	3245259
666118	3209855	649767	3226031	653979	3246895	655806	3245909
664820	3211292	646121	3225760	653899	3246959	656233	3245076
664668	3211015	646017	3225765	653828	3247081	656579	3244589
664578	3210764	646557	3225387	653771	3247156	656551	3244544
668546	3212298	649162	3226358	653737	3247189	656524	3244527
662842	3212497	649188	3226368	653683	3247215		
662727	3212458	664301	3225571	653670	3247211		
667094	3209013	657434	3245302	653563	3247210		
664567	3215593	657399	3245371	653444	3247296		
680347	3255669	657282	3245409	653387	3247276		
680530	3255674	657256	3245431	653271	3247212		
649253	3226358	657191	3245490	653182	3247293		

Appendix D. Jackknife (500 iterations) results of training gain for each model variable and invasive plant. Higher gain numbers indicate variable that when used in isolation appear to have the most useful information by itself. Blank spaces indicate the variable was not retained in the model.

Model Variable	Invasive Plant								
	ARDO	CYDA	ERLE	MAVU	PECI	SAKA	SOHA	TAMA	ALL
DEM	1.79	0.30	0.40	1.53	0.60	0.24		0.30	0.09
Fall 2000 Band 1			0.06		0.19	0.25			0.02
Fall 2000 Band 2	0.10	0.08	0.07		0.17		0.82	0.08	0.02
Fall 2000 Band 3			0.06		0.15		0.82		
Fall 2000 Band 4		0.06	0.16	0.13	0.12	0.23	0.82	0.08	0.03
Fall 2000 Band 5		0.03	0.16		0.15		0.84	0.07	0.03
Fall 2000 Band 6	0.54	0.07	0.11	0.11	0.13	0.46	0.83		0.01
Fall 2001 Band 1		0.07			0.11	0.21	0.89		0.01
Fall 2001 Band 2					0.15	0.34		0.05	0.01
Fall 2001 Band 3	0.18	0.13	0.10		0.15	0.36	0.93	0.03	0.01
Fall 2001 Band 4	0.33	0.12	0.17		0.10	0.77	0.93	0.16	0.03
Fall 2001 Band 5	0.24	0.07	0.22	0.47			0.84	0.09	0.03
Fall 2001 Band 6	0.41	0.12		0.55	0.17		0.87	0.05	
Fall 1999 Band 1		0.09	0.15	1.38	0.12	0.03	0.88	0.03	0.02
Fall 1999 Band 2		0.09			0.09		0.90	0.01	0.02
Fall 1999 Band 3		0.14	0.10		0.14	0.08	0.90	0.02	
Fall 1999 Band 4	0.01	0.05	0.09	0.26	0.13		0.87	0.01	0.01
Fall 1999 Band 5	0.44	0.09	0.15		0.15		0.86	0.04	0.03
Fall 1999 Band 6	0.52	0.16	0.13		0.15	0.11	0.91	0.06	0.03
Spring 2001 Band 1	0.17	0.08	0.12		0.11		0.81	0.07	0.01
Spring 2001 Band 2					0.19		0.81	0.10	0.01
Spring 2001 Band 3			0.11		0.16		0.82	0.07	0.01
Spring 2001 Band 4	0.00	0.02	0.08	0.00	0.18	0.21	0.85	0.07	0.03
Spring 2001 Band 5		0.10	0.24	0.20	0.23	0.25	0.82	0.15	0.04
Spring 2001 Band 6	0.74	0.21	0.19	0.24	0.16			0.10	0.02
Spring 2002 Band 1		0.06	0.11		0.13		0.85	0.04	
Spring 2002 Band 2			0.11		0.17	0.41			
Spring 2002 Band 3		0.10		0.14		0.46	0.86	0.05	
Spring 2002 Band 4	0.31		0.15		0.11		0.88	0.11	0.00
Spring 2002 Band 5	0.28		0.21		0.24	0.50	0.86	0.13	0.03
Spring 2002 Band 6		0.09	0.13	0.14	0.23	0.52	0.83		0.01
Spring 2003 Band 1		0.08	0.14			0.08	0.85	0.01	0.01
Spring 2003 Band 2		0.08	0.12						
Spring 2003 Band 3	0.12	0.14			0.17	0.23	0.88	0.01	0.01
Spring 2003 Band 4	0.14	0.04	0.12				0.88	0.13	0.02
Spring 2003 Band 5		0.09	0.24		0.22		0.83	0.09	0.05
Spring 2003 Band 6		0.12	0.17	0.36	0.19		0.84	0.05	0.03
Summer 2000 Band 1		0.07			0.15		0.88	0.08	0.01
Summer 2000 Band 2			0.09					0.10	0.01
Summer 2000 Band 3	0.25	0.12		0.34	0.20	0.36	0.89	0.11	0.01
Summer 2000 Band 4	0.24	0.10	0.15	0.03	0.20	0.62	0.98	0.17	0.06
Summer 2000 Band 5		0.07	0.20	0.25	0.21	0.40	0.85	0.12	0.03
Summer 2000 Band 6	0.56	0.16	0.13		0.22			0.10	0.01

Appendix E. Soil attributes of Big Bend National Park (Cockran and Rives 1985). Highlighted soil units were considered as having sufficient soil moisture properties to enhance establishment of invasive plants.

Soil Acronym	Soil unit	Texture	Depth	Water Capacity	Permeability
CHD	Chamberino very gravelly loam, rolling (CHD)	Very cobbly and gravelly loam	Very shallow to shallow	Low	Moderate
CHC	Chamberino very gravelly loam, undulating (CHC)	Very gravelly loam	Deep	Low	Moderate
CHD	Chamberino very gravelly loam, rolling (CHD)	Very cobbly and gravelly loam	Deep	Low	Moderate
CLC	Chilicotal very gravelly fine sandy loam, undulating	Very gravelly fine sandy loam	Deep	Low	Moderate
CMD	Chilicotal-Monterosa association, rolling (CMD)	Very cobbly and gravelly loam	Deep	Low	Moderate
PAC	Paisano very gravelly fine sandy loam, undulating	Very gravelly fine sandy loam	Very shallow and shallow	Very low	Moderate
PNA	Pantera very gravelly sandy loam, frequently flooded	Very gravelly sandy loam	Deep	Very low to low	Moderately rapid
COC	Corazones very gravelly sandy loam, undulating	Very gravelly sandy loam	Deep	Very low	Moderate
COD	Corazones very gravelly sandy loam, strongly rolling	Very gravelly sandy loam	Deep	Very low	Moderate
LAF	Lajitas-Rock outcrop complex, steep (LAF)	Very cobbly and very gravelly	Very shallow and shallow	Low	Moderate
PAA	Pajarito-Agustin association, gently sloping	Fine sandy loam	Deep	Moderate	Moderately rapid
PAA	Pajarito-Agustin association, gently sloping	Fine sandy loam	Deep	Moderate	Moderately rapid
RLA	Riverwash	sandy, gravelly	Deep	Moderate	Moderately rapid
TOA	Tornillo loam, occasionally flooded (TOA)	Loam	Deep	High	Moderate
LRE	Lozier-Rock outcrop complex, dry, hilly	Very cobbly and very gravelly	Very shallow and shallow	Very low	Moderate
MDD	Mariscal-Rock outcrop complex, rolling	Very channery or very flaggy	Very shallow or shallow	Very low	Moderate
MDE	Mariscal-Rock outcrop complex, hilly	Very channery or very flaggy	Very shallow or shallow	Very low	Moderate
MNE	Mariscal-Terlingua complex, hilly	Very gravelly sandy loam	Very shallow and shallow	Very low	Moderate
LZE	Lozier-Rock outcrop complex, warm, hilly	Very cobbly and very gravelly	Very shallow and shallow	Very low	Moderate
TEC	Tencee-Stillwell complex, undulating	Very gravelly sandy loam	Shallow	Very low	Moderate
TED	Tencee-Stillwell complex, strongly rolling	Very gravelly sandy loam	Shallow	Very low	Moderate
VBD	Vieja-Badland complex, rolling (VBD)	Very gravelly clayey	Very shallow and shallow	Low	Slow
BIC	Bissett very gravelly loam, undulating	Very gravelly loam	Deep	Very low	Moderate
LMF	Liv-Mainstay-Rock outcrop complex, steep (LMF)	Very cobbly and very gravelly	Shallow to deep	Low to very low	Moderately slow
HRD	Hurds very gravelly sandy loam, rolling (HRD)	Very gravelly sandy loam	Deep	Low	Moderate
HRF	Hurds very cobbly loam (HRF)	Very cobbly loam	Deep	Low	Moderate
GHA	Glendale-Harkey association, occasionally flooded	Silty clay loam	Deep	High	Moderately slow
RVW	Riverwash (RVW)	sandy, gravelly	Deep	Moderate	Moderately rapid

Appendix E. cont.

Soil Acronym	Soil unit	Texture	Depth	Water Capacity	Permeability
BRG	Brewster-Rock outcrop complex, very steep (BRG)	Rock outcrop to very gravelly loam	Very Shallow	Very Low	Moderate
ERF	Ector-Rock outcrop complex, steep (ERF)	Very gravelly to cobbly	Very shallow to shallow	Low	Moderate
LAE	Lajitas-Rock outcrop complex, hilly (LAE)	Very cobbly and very gravelly	Very shallow and shallow	Low	Moderate
LAF	Lajitas-Rock outcrop complex, steep (LAF)	Very cobbly and very gravelly	Very shallow and shallow	Low	Moderate
LRF	Lozier-Rock outcrop complex, steep (LRF)	Very cobbly and very gravelly	Very shallow and shallow	Very low	Moderate
LRG	Lozier-Rock outcrop complex, very steep (LRG)	Very cobbly and very gravelly	Very shallow and shallow	Very low	Moderate
MRE	Mariscal-Rock outcrop complex, hilly (MRE)	Very channery or very flaggy	Very shallow or shallow	Very low	Moderate
SRD	Solis-Rock outcrop complex, rolling (SRD)	Fine sandy loam	Very shallow and shallow	Very low	Moderate
SRD	Solis-Rock outcrop complex, rolling (SRD)	Fine sandy loam	Very shallow and shallow	Very low	Moderate
TLE	Terlingua-Rock outcrop complex, hilly (TLE)	Very gravelly coarse sandy loam	Very shallow and shallow	Very low	Moderate
BIG	Bissett-Rock outcrop complex, very steep	Very gravelly loam	Shallow	Very low	Moderate
VBD	Vieja-Badland complex, rolling (VBD)	Very gravelly clayey	Very shallow and shallow	Low	Slow
CRB	Crossen gravelly loam, gently undulating	Gravelly loam	Very shallow and shallow	Very low	Moderate slow
DA	Area Not Surveyed	Unknown	Unknown	Unknown	Unknown
TAE	Terlingua-Mariscal association, hilly (TAE)	Very gravelly sandy loam	Very shallow and shallow	Very low	Moderate
W	Water	N/A	N/A	N/A	N/A
UNC	Upton-Nickel association, undulating (UNC)	Gravelly and very gravelly loam	Very shallow and shallow	Very low	Moderate
PRF	Puerta-Madrone complex, steep (PRF)	Very cobbly and very gravelly	Shallow	Very low	Moderately slow

Appendix F. Possible combinations of model input variables that may constitute high risk areas. Only combinations with model values $\geq$ to 450 are provided.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	High	High	High	< 250 m	Fire	900
< 50 m	High	High	High	< 250 m	None	851
< 50 m	High	Moderate	High	< 250 m	Fire	850
< 50 m	High	High	Moderate	< 250 m	Fire	850
< 50 m	High	Moderate	High	< 250 m	None	801
< 50 m	High	High	Moderate	< 250 m	None	801
< 50 m	High	Low	High	< 250 m	Fire	800
< 50 m	High	Moderate	Moderate	< 250 m	Fire	800
< 50 m	High	High	Low	< 250 m	Fire	800
< 50 m	High	Low	High	< 250 m	None	751
< 50 m	High	Moderate	Moderate	< 250 m	None	751
< 50 m	High	High	Low	< 250 m	None	751
< 50 m	High	not-suitable	High	< 250 m	Fire	751
< 50 m	High	High	None	< 250 m	Fire	751
< 50 m	High	Low	Moderate	< 250 m	Fire	750
< 50 m	High	Moderate	Low	< 250 m	Fire	750
< 50 m	High	not-suitable	High	< 250 m	None	702
< 50 m	High	High	None	< 250 m	None	702
< 50 m	High	Low	Moderate	< 250 m	None	701
< 50 m	High	Moderate	Low	< 250 m	None	701
< 50 m	High	not-suitable	Moderate	< 250 m	Fire	701
< 50 m	High	Moderate	None	< 250 m	Fire	701
< 50 m	High	Low	Low	< 250 m	Fire	700
< 50 m	High	not-suitable	Moderate	< 250 m	None	652
< 50 m	High	Moderate	None	< 250 m	None	652
< 50 m	High	Low	Low	< 250 m	None	651
< 50 m	High	not-suitable	Low	< 250 m	Fire	651
< 50 m	High	Low	None	< 250 m	Fire	651
< 50 m	High	not-suitable	Low	< 250 m	None	602
< 50 m	High	Low	None	< 250 m	None	602
< 50 m	High	not-suitable	None	< 250 m	Fire	602
< 50 m	High	not-suitable	None	< 250 m	None	553
< 50 m	High	High	High	> 250 m	Fire	851
< 50 m	High	High	High	> 250 m	None	802
< 50 m	High	Moderate	High	> 250 m	Fire	801
< 50 m	High	High	Moderate	> 250 m	Fire	801
< 50 m	High	Moderate	High	> 250 m	None	752
< 50 m	High	High	Moderate	> 250 m	None	752
< 50 m	High	Low	High	> 250 m	Fire	751
< 50 m	High	Moderate	Moderate	> 250 m	Fire	751
< 50 m	High	High	Low	> 250 m	Fire	751
< 50 m	High	Low	High	> 250 m	None	702
< 50 m	High	Moderate	Moderate	> 250 m	None	702
< 50 m	High	High	Low	> 250 m	None	702
< 50 m	High	not-suitable	High	> 250 m	Fire	702
< 50 m	High	High	None	> 250 m	Fire	702

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	High	Low	Moderate	> 250 m	Fire	701
< 50 m	High	Moderate	Low	> 250 m	Fire	701
< 50 m	High	not-suitable	High	> 250 m	None	653
< 50 m	High	High	None	> 250 m	None	653
< 50 m	High	Low	Moderate	> 250 m	None	652
< 50 m	High	Moderate	Low	> 250 m	None	652
< 50 m	High	not-suitable	Moderate	> 250 m	Fire	652
< 50 m	High	Moderate	None	> 250 m	Fire	652
< 50 m	High	Low	Low	> 250 m	Fire	651
< 50 m	High	not-suitable	Moderate	> 250 m	None	603
< 50 m	High	Moderate	None	> 250 m	None	603
< 50 m	High	Low	Low	> 250 m	None	602
< 50 m	High	not-suitable	Low	> 250 m	Fire	602
< 50 m	High	Low	None	> 250 m	Fire	602
< 50 m	High	not-suitable	Low	> 250 m	None	553
< 50 m	High	Low	None	> 250 m	None	553
< 50 m	High	not-suitable	None	> 250 m	Fire	553
< 50 m	High	not-suitable	None	> 250 m	None	504
< 50 m	Moderate	High	High	< 250 m	Fire	850
< 50 m	Moderate	Moderate	High	< 250 m	Fire	800
< 50 m	Moderate	High	Moderate	< 250 m	Fire	800
< 50 m	Moderate	Low	High	< 250 m	Fire	750
< 50 m	Moderate	Moderate	Moderate	< 250 m	Fire	750
< 50 m	Moderate	High	Low	< 250 m	Fire	750
< 50 m	Moderate	not-suitable	High	< 250 m	Fire	701
< 50 m	Moderate	High	None	< 250 m	Fire	701
< 50 m	Moderate	Low	Moderate	< 250 m	Fire	700
< 50 m	Moderate	Moderate	Low	< 250 m	Fire	700
< 50 m	Moderate	not-suitable	Moderate	< 250 m	Fire	651
< 50 m	Moderate	Moderate	None	< 250 m	Fire	651
< 50 m	Moderate	Low	Low	< 250 m	Fire	650
< 50 m	Moderate	not-suitable	Low	< 250 m	Fire	601
< 50 m	Moderate	Low	None	< 250 m	Fire	601
< 50 m	Moderate	not-suitable	None	< 250 m	Fire	552
< 50 m	Moderate	High	High	< 250 m	None	801
< 50 m	Moderate	Moderate	High	< 250 m	None	751
< 50 m	Moderate	High	Moderate	< 250 m	None	751
< 50 m	Moderate	Low	High	< 250 m	None	701
< 50 m	Moderate	Moderate	Moderate	< 250 m	None	701
< 50 m	Moderate	High	Low	< 250 m	None	701
< 50 m	Moderate	not-suitable	High	< 250 m	None	652
< 50 m	Moderate	High	None	< 250 m	None	652
< 50 m	Moderate	Low	Moderate	< 250 m	None	651
< 50 m	Moderate	Moderate	Low	< 250 m	None	651
< 50 m	Moderate	not-suitable	Moderate	< 250 m	None	602
< 50 m	Moderate	Moderate	None	< 250 m	None	602
< 50 m	Moderate	Low	Low	< 250 m	None	601
< 50 m	Moderate	not-suitable	Low	< 250 m	None	552

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	Moderate	Low	None	< 250 m	None	552
< 50 m	Moderate	not-suitable	None	< 250 m	None	503
< 50 m	Moderate	High	High	> 250 m	Fire	801
< 50 m	Moderate	Moderate	High	> 250 m	Fire	751
< 50 m	Moderate	High	Moderate	> 250 m	Fire	751
< 50 m	Moderate	Low	High	> 250 m	Fire	701
< 50 m	Moderate	Moderate	Moderate	> 250 m	Fire	701
< 50 m	Moderate	High	Low	> 250 m	Fire	701
< 50 m	Moderate	not-suitable	High	> 250 m	Fire	652
< 50 m	Moderate	High	None	> 250 m	Fire	652
< 50 m	Moderate	Low	Moderate	> 250 m	Fire	651
< 50 m	Moderate	Moderate	Low	> 250 m	Fire	651
< 50 m	Moderate	not-suitable	Moderate	> 250 m	Fire	602
< 50 m	Moderate	Moderate	None	> 250 m	Fire	602
< 50 m	Moderate	Low	Low	> 250 m	Fire	601
< 50 m	Moderate	not-suitable	Low	> 250 m	Fire	552
< 50 m	Moderate	Low	None	> 250 m	Fire	552
< 50 m	Moderate	not-suitable	None	> 250 m	Fire	503
< 50 m	Moderate	High	High	> 250 m	None	752
< 50 m	Moderate	Moderate	High	> 250 m	None	702
< 50 m	Moderate	High	Moderate	> 250 m	None	702
< 50 m	Moderate	Low	High	> 250 m	None	652
< 50 m	Moderate	Moderate	Moderate	> 250 m	None	652
< 50 m	Moderate	High	Low	> 250 m	None	652
< 50 m	Moderate	not-suitable	High	> 250 m	None	603
< 50 m	Moderate	High	None	> 250 m	None	603
< 50 m	Moderate	Low	Moderate	> 250 m	None	602
< 50 m	Moderate	Moderate	Low	> 250 m	None	602
< 50 m	Moderate	not-suitable	Moderate	> 250 m	None	553
< 50 m	Moderate	Moderate	None	> 250 m	None	553
< 50 m	Moderate	Low	Low	> 250 m	None	552
< 50 m	Moderate	not-suitable	Low	> 250 m	None	503
< 50 m	Moderate	Low	None	> 250 m	None	503
< 50 m	Moderate	not-suitable	None	> 250 m	None	454
< 50 m	Low	High	High	< 250 m	Fire	800
< 50 m	Low	Moderate	High	< 250 m	Fire	750
< 50 m	Low	High	Moderate	< 250 m	Fire	750
< 50 m	Low	Low	High	< 250 m	Fire	700
< 50 m	Low	Moderate	Moderate	< 250 m	Fire	700
< 50 m	Low	High	Low	< 250 m	Fire	700
< 50 m	Low	not-suitable	High	< 250 m	Fire	651
< 50 m	Low	High	None	< 250 m	Fire	651
< 50 m	Low	Low	Moderate	< 250 m	Fire	650
< 50 m	Low	Moderate	Low	< 250 m	Fire	650
< 50 m	Low	not-suitable	Moderate	< 250 m	Fire	601
< 50 m	Low	Moderate	None	< 250 m	Fire	601
< 50 m	Low	Low	Low	< 250 m	Fire	600
< 50 m	Low	not-suitable	Low	< 250 m	Fire	551

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	Low	Low	None	< 250 m	Fire	551
< 50 m	Low	not-suitable	None	< 250 m	Fire	502
< 50 m	Low	High	High	< 250 m	None	751
< 50 m	Low	Moderate	High	< 250 m	None	701
< 50 m	Low	High	Moderate	< 250 m	None	701
< 50 m	Low	Low	High	< 250 m	None	651
< 50 m	Low	Moderate	Moderate	< 250 m	None	651
< 50 m	Low	High	Low	< 250 m	None	651
< 50 m	Low	not-suitable	High	< 250 m	None	602
< 50 m	Low	High	None	< 250 m	None	602
< 50 m	Low	Low	Moderate	< 250 m	None	601
< 50 m	Low	Moderate	Low	< 250 m	None	601
< 50 m	Low	not-suitable	Moderate	< 250 m	None	552
< 50 m	Low	Moderate	None	< 250 m	None	552
< 50 m	Low	Low	Low	< 250 m	None	551
< 50 m	Low	not-suitable	Low	< 250 m	None	502
< 50 m	Low	Low	None	< 250 m	None	502
< 50 m	Low	not-suitable	None	< 250 m	None	453
< 50 m	Low	High	High	> 250 m	Fire	751
< 50 m	Low	Moderate	High	> 250 m	Fire	701
< 50 m	Low	High	Moderate	> 250 m	Fire	701
< 50 m	Low	Low	High	> 250 m	Fire	651
< 50 m	Low	Moderate	Moderate	> 250 m	Fire	651
< 50 m	Low	High	Low	> 250 m	Fire	651
< 50 m	Low	not-suitable	High	> 250 m	Fire	602
< 50 m	Low	High	None	> 250 m	Fire	602
< 50 m	Low	Low	Moderate	> 250 m	Fire	601
< 50 m	Low	Moderate	Low	> 250 m	Fire	601
< 50 m	Low	not-suitable	Moderate	> 250 m	Fire	552
< 50 m	Low	Moderate	None	> 250 m	Fire	552
< 50 m	Low	Low	Low	> 250 m	Fire	551
< 50 m	Low	not-suitable	Low	> 250 m	Fire	502
< 50 m	Low	Low	None	> 250 m	Fire	502
< 50 m	Low	not-suitable	None	> 250 m	Fire	453
< 50 m	Low	High	High	> 250 m	None	702
< 50 m	Low	Moderate	High	> 250 m	None	652
< 50 m	Low	High	Moderate	> 250 m	None	652
< 50 m	Low	Low	High	> 250 m	None	602
< 50 m	Low	Moderate	Moderate	> 250 m	None	602
< 50 m	Low	High	Low	> 250 m	None	602
< 50 m	Low	not-suitable	High	> 250 m	None	553
< 50 m	Low	High	None	> 250 m	None	553
< 50 m	Low	Low	Moderate	> 250 m	None	552
< 50 m	Low	Moderate	Low	> 250 m	None	552
< 50 m	Low	not-suitable	Moderate	> 250 m	None	503
< 50 m	Low	Moderate	None	> 250 m	None	503
< 50 m	Low	Low	Low	> 250 m	None	502
< 50 m	Low	not-suitable	Low	> 250 m	None	453

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	Low	Low	None	> 250 m	None	453
< 50 m	not-suitable	High	High	< 250 m	Fire	651
< 50 m	not-suitable	Moderate	High	< 250 m	Fire	601
< 50 m	not-suitable	High	Moderate	< 250 m	Fire	601
< 50 m	not-suitable	Low	High	< 250 m	Fire	551
< 50 m	not-suitable	Moderate	Moderate	< 250 m	Fire	551
< 50 m	not-suitable	High	Low	< 250 m	Fire	551
< 50 m	not-suitable	not-suitable	High	< 250 m	Fire	502
< 50 m	not-suitable	High	None	< 250 m	Fire	502
< 50 m	not-suitable	Low	Moderate	< 250 m	Fire	501
< 50 m	not-suitable	Moderate	Low	< 250 m	Fire	501
< 50 m	not-suitable	not-suitable	Moderate	< 250 m	Fire	452
< 50 m	not-suitable	Moderate	None	< 250 m	Fire	452
< 50 m	not-suitable	Low	Low	< 250 m	Fire	451
< 50 m	not-suitable	High	High	< 250 m	None	602
< 50 m	not-suitable	Moderate	High	< 250 m	None	552
< 50 m	not-suitable	High	Moderate	< 250 m	None	552
< 50 m	not-suitable	Low	High	< 250 m	None	502
< 50 m	not-suitable	Moderate	Moderate	< 250 m	None	502
< 50 m	not-suitable	High	Low	< 250 m	None	502
< 50 m	not-suitable	not-suitable	High	< 250 m	None	453
< 50 m	not-suitable	High	None	< 250 m	None	453
< 50 m	not-suitable	Low	Moderate	< 250 m	None	452
< 50 m	not-suitable	Moderate	Low	< 250 m	None	452
< 50 m	not-suitable	High	High	> 250 m	Fire	602
< 50 m	not-suitable	Moderate	High	> 250 m	Fire	552
< 50 m	not-suitable	High	Moderate	> 250 m	Fire	552
< 50 m	not-suitable	Low	High	> 250 m	Fire	502
< 50 m	not-suitable	Moderate	Moderate	> 250 m	Fire	502
< 50 m	not-suitable	High	Low	> 250 m	Fire	502
< 50 m	not-suitable	not-suitable	High	> 250 m	Fire	453
< 50 m	not-suitable	High	None	> 250 m	Fire	453
< 50 m	not-suitable	Low	Moderate	> 250 m	Fire	452
< 50 m	not-suitable	Moderate	Low	> 250 m	Fire	452
< 50 m	not-suitable	High	High	> 250 m	None	553
< 50 m	not-suitable	Moderate	High	> 250 m	None	503
< 50 m	not-suitable	High	Moderate	> 250 m	None	503
< 50 m	not-suitable	Low	High	> 250 m	None	453
< 50 m	not-suitable	Moderate	Moderate	> 250 m	None	453
< 50 m	not-suitable	High	Low	> 250 m	None	453
50 to 100 m	High	High	High	< 250 m	Fire	850
50 to 100 m	High	High	High	< 250 m	None	801
50 to 100 m	High	Moderate	High	< 250 m	Fire	800
50 to 100 m	High	High	Moderate	< 250 m	Fire	800
50 to 100 m	High	Moderate	High	< 250 m	None	751
50 to 100 m	High	High	Moderate	< 250 m	None	751
50 to 100 m	High	Low	High	< 250 m	Fire	750
50 to 100 m	High	Moderate	Moderate	< 250 m	Fire	750

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	High	High	Low	< 250 m	Fire	750
50 to 100 m	High	Low	High	< 250 m	None	701
50 to 100 m	High	Moderate	Moderate	< 250 m	None	701
50 to 100 m	High	High	Low	< 250 m	None	701
50 to 100 m	High	not-suitable	High	< 250 m	Fire	701
50 to 100 m	High	High	None	< 250 m	Fire	701
50 to 100 m	High	Low	Moderate	< 250 m	Fire	700
50 to 100 m	High	Moderate	Low	< 250 m	Fire	700
50 to 100 m	High	not-suitable	High	< 250 m	None	652
50 to 100 m	High	High	None	< 250 m	None	652
50 to 100 m	High	Low	Moderate	< 250 m	None	651
50 to 100 m	High	Moderate	Low	< 250 m	None	651
50 to 100 m	High	not-suitable	Moderate	< 250 m	Fire	651
50 to 100 m	High	Moderate	None	< 250 m	Fire	651
50 to 100 m	High	Low	Low	< 250 m	Fire	650
50 to 100 m	High	not-suitable	Moderate	< 250 m	None	602
50 to 100 m	High	Moderate	None	< 250 m	None	602
50 to 100 m	High	Low	Low	< 250 m	None	601
50 to 100 m	High	not-suitable	Low	< 250 m	Fire	601
50 to 100 m	High	Low	None	< 250 m	Fire	601
50 to 100 m	High	not-suitable	Low	< 250 m	None	552
50 to 100 m	High	Low	None	< 250 m	None	552
50 to 100 m	High	not-suitable	None	< 250 m	Fire	552
50 to 100 m	High	not-suitable	None	< 250 m	None	503
50 to 100 m	High	High	High	> 250 m	Fire	801
50 to 100 m	High	High	High	> 250 m	None	752
50 to 100 m	High	Moderate	High	> 250 m	Fire	751
50 to 100 m	High	High	Moderate	> 250 m	Fire	751
50 to 100 m	High	Moderate	High	> 250 m	None	702
50 to 100 m	High	High	Moderate	> 250 m	None	702
50 to 100 m	High	Low	High	> 250 m	Fire	701
50 to 100 m	High	Moderate	Moderate	> 250 m	Fire	701
50 to 100 m	High	High	Low	> 250 m	Fire	701
50 to 100 m	High	Low	High	> 250 m	None	652
50 to 100 m	High	Moderate	Moderate	> 250 m	None	652
50 to 100 m	High	High	Low	> 250 m	None	652
50 to 100 m	High	not-suitable	High	> 250 m	Fire	652
50 to 100 m	High	High	None	> 250 m	Fire	652
50 to 100 m	High	Low	Moderate	> 250 m	Fire	651
50 to 100 m	High	Moderate	Low	> 250 m	Fire	651
50 to 100 m	High	not-suitable	High	> 250 m	None	603
50 to 100 m	High	High	None	> 250 m	None	603
50 to 100 m	High	Low	Moderate	> 250 m	None	602
50 to 100 m	High	Moderate	Low	> 250 m	None	602
50 to 100 m	High	not-suitable	Moderate	> 250 m	Fire	602
50 to 100 m	High	Moderate	None	> 250 m	Fire	602
50 to 100 m	High	Low	Low	> 250 m	Fire	601
50 to 100 m	High	not-suitable	Moderate	> 250 m	None	553

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	High	Moderate	None	> 250 m	None	553
50 to 100 m	High	Low	Low	> 250 m	None	552
50 to 100 m	High	not-suitable	Low	> 250 m	Fire	552
50 to 100 m	High	Low	None	> 250 m	Fire	552
50 to 100 m	High	not-suitable	Low	> 250 m	None	503
50 to 100 m	High	Low	None	> 250 m	None	503
50 to 100 m	High	not-suitable	None	> 250 m	Fire	503
50 to 100 m	High	not-suitable	None	> 250 m	None	454
50 to 100 m	Moderate	High	High	< 250 m	Fire	800
50 to 100 m	Moderate	Moderate	High	< 250 m	Fire	750
50 to 100 m	Moderate	High	Moderate	< 250 m	Fire	750
50 to 100 m	Moderate	Low	High	< 250 m	Fire	700
50 to 100 m	Moderate	Moderate	Moderate	< 250 m	Fire	700
50 to 100 m	Moderate	High	Low	< 250 m	Fire	700
50 to 100 m	Moderate	not-suitable	High	< 250 m	Fire	651
50 to 100 m	Moderate	High	None	< 250 m	Fire	651
50 to 100 m	Moderate	Low	Moderate	< 250 m	Fire	650
50 to 100 m	Moderate	Moderate	Low	< 250 m	Fire	650
50 to 100 m	Moderate	not-suitable	Moderate	< 250 m	Fire	601
50 to 100 m	Moderate	Moderate	None	< 250 m	Fire	601
50 to 100 m	Moderate	Low	Low	< 250 m	Fire	600
50 to 100 m	Moderate	not-suitable	Low	< 250 m	Fire	551
50 to 100 m	Moderate	Low	None	< 250 m	Fire	551
50 to 100 m	Moderate	not-suitable	None	< 250 m	Fire	502
50 to 100 m	Moderate	High	High	< 250 m	None	751
50 to 100 m	Moderate	Moderate	High	< 250 m	None	701
50 to 100 m	Moderate	High	Moderate	< 250 m	None	701
50 to 100 m	Moderate	Low	High	< 250 m	None	651
50 to 100 m	Moderate	Moderate	Moderate	< 250 m	None	651
50 to 100 m	Moderate	High	Low	< 250 m	None	651
50 to 100 m	Moderate	not-suitable	High	< 250 m	None	602
50 to 100 m	Moderate	High	None	< 250 m	None	602
50 to 100 m	Moderate	Low	Moderate	< 250 m	None	601
50 to 100 m	Moderate	Moderate	Low	< 250 m	None	601
50 to 100 m	Moderate	not-suitable	Moderate	< 250 m	None	552
50 to 100 m	Moderate	Moderate	None	< 250 m	None	552
50 to 100 m	Moderate	Low	Low	< 250 m	None	551
50 to 100 m	Moderate	not-suitable	Low	< 250 m	None	502
50 to 100 m	Moderate	Low	None	< 250 m	None	502
50 to 100 m	Moderate	not-suitable	None	< 250 m	None	453
50 to 100 m	Moderate	High	High	> 250 m	Fire	751
50 to 100 m	Moderate	Moderate	High	> 250 m	Fire	701
50 to 100 m	Moderate	High	Moderate	> 250 m	Fire	701
50 to 100 m	Moderate	Low	High	> 250 m	Fire	651
50 to 100 m	Moderate	Moderate	Moderate	> 250 m	Fire	651
50 to 100 m	Moderate	High	Low	> 250 m	Fire	651
50 to 100 m	Moderate	not-suitable	High	> 250 m	Fire	602
50 to 100 m	Moderate	High	None	> 250 m	Fire	602

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	Moderate	Low	Moderate	> 250 m	Fire	601
50 to 100 m	Moderate	Moderate	Low	> 250 m	Fire	601
50 to 100 m	Moderate	not-suitable	Moderate	> 250 m	Fire	552
50 to 100 m	Moderate	Moderate	None	> 250 m	Fire	552
50 to 100 m	Moderate	Low	Low	> 250 m	Fire	551
50 to 100 m	Moderate	not-suitable	Low	> 250 m	Fire	502
50 to 100 m	Moderate	Low	None	> 250 m	Fire	502
50 to 100 m	Moderate	not-suitable	None	> 250 m	Fire	453
50 to 100 m	Moderate	High	High	> 250 m	None	702
50 to 100 m	Moderate	Moderate	High	> 250 m	None	652
50 to 100 m	Moderate	High	Moderate	> 250 m	None	652
50 to 100 m	Moderate	Low	High	> 250 m	None	602
50 to 100 m	Moderate	Moderate	Moderate	> 250 m	None	602
50 to 100 m	Moderate	High	Low	> 250 m	None	602
50 to 100 m	Moderate	not-suitable	High	> 250 m	None	553
50 to 100 m	Moderate	High	None	> 250 m	None	553
50 to 100 m	Moderate	Low	Moderate	> 250 m	None	552
50 to 100 m	Moderate	Moderate	Low	> 250 m	None	552
50 to 100 m	Moderate	not-suitable	Moderate	> 250 m	None	503
50 to 100 m	Moderate	Moderate	None	> 250 m	None	503
50 to 100 m	Moderate	Low	Low	> 250 m	None	502
50 to 100 m	Moderate	not-suitable	Low	> 250 m	None	453
50 to 100 m	Moderate	Low	None	> 250 m	None	453
50 to 100 m	Low	High	High	< 250 m	Fire	750
50 to 100 m	Low	Moderate	High	< 250 m	Fire	700
50 to 100 m	Low	High	Moderate	< 250 m	Fire	700
50 to 100 m	Low	Low	High	< 250 m	Fire	650
50 to 100 m	Low	Moderate	Moderate	< 250 m	Fire	650
50 to 100 m	Low	High	Low	< 250 m	Fire	650
50 to 100 m	Low	not-suitable	High	< 250 m	Fire	601
50 to 100 m	Low	High	None	< 250 m	Fire	601
50 to 100 m	Low	Low	Moderate	< 250 m	Fire	600
50 to 100 m	Low	Moderate	Low	< 250 m	Fire	600
50 to 100 m	Low	not-suitable	Moderate	< 250 m	Fire	551
50 to 100 m	Low	Moderate	None	< 250 m	Fire	551
50 to 100 m	Low	Low	Low	< 250 m	Fire	550
50 to 100 m	Low	not-suitable	Low	< 250 m	Fire	501
50 to 100 m	Low	Low	None	< 250 m	Fire	501
50 to 100 m	Low	not-suitable	None	< 250 m	Fire	452
50 to 100 m	Low	High	High	< 250 m	None	701
50 to 100 m	Low	Moderate	High	< 250 m	None	651
50 to 100 m	Low	High	Moderate	< 250 m	None	651
50 to 100 m	Low	Low	High	< 250 m	None	601
50 to 100 m	Low	Moderate	Moderate	< 250 m	None	601
50 to 100 m	Low	High	Low	< 250 m	None	601
50 to 100 m	Low	not-suitable	High	< 250 m	None	552
50 to 100 m	Low	High	None	< 250 m	None	552
50 to 100 m	Low	Low	Moderate	< 250 m	None	551

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	Low	Moderate	Low	< 250 m	None	551
50 to 100 m	Low	not-suitable	Moderate	< 250 m	None	502
50 to 100 m	Low	Moderate	None	< 250 m	None	502
50 to 100 m	Low	Low	Low	< 250 m	None	501
50 to 100 m	Low	not-suitable	Low	< 250 m	None	452
50 to 100 m	Low	Low	None	< 250 m	None	452
50 to 100 m	Low	High	High	> 250 m	Fire	701
50 to 100 m	Low	Moderate	High	> 250 m	Fire	651
50 to 100 m	Low	High	Moderate	> 250 m	Fire	651
50 to 100 m	Low	Low	High	> 250 m	Fire	601
50 to 100 m	Low	Moderate	Moderate	> 250 m	Fire	601
50 to 100 m	Low	High	Low	> 250 m	Fire	601
50 to 100 m	Low	not-suitable	High	> 250 m	Fire	552
50 to 100 m	Low	High	None	> 250 m	Fire	552
50 to 100 m	Low	Low	Moderate	> 250 m	Fire	551
50 to 100 m	Low	Moderate	Low	> 250 m	Fire	551
50 to 100 m	Low	not-suitable	Moderate	> 250 m	Fire	502
50 to 100 m	Low	Moderate	None	> 250 m	Fire	502
50 to 100 m	Low	Low	Low	> 250 m	Fire	501
50 to 100 m	Low	not-suitable	Low	> 250 m	Fire	452
50 to 100 m	Low	Low	None	> 250 m	Fire	452
50 to 100 m	Low	High	High	> 250 m	None	652
50 to 100 m	Low	Moderate	High	> 250 m	None	602
50 to 100 m	Low	High	Moderate	> 250 m	None	602
50 to 100 m	Low	Low	High	> 250 m	None	552
50 to 100 m	Low	Moderate	Moderate	> 250 m	None	552
50 to 100 m	Low	High	Low	> 250 m	None	552
50 to 100 m	Low	not-suitable	High	> 250 m	None	503
50 to 100 m	Low	High	None	> 250 m	None	503
50 to 100 m	Low	Low	Moderate	> 250 m	None	502
50 to 100 m	Low	Moderate	Low	> 250 m	None	502
50 to 100 m	Low	not-suitable	Moderate	> 250 m	None	453
50 to 100 m	Low	Moderate	None	> 250 m	None	453
50 to 100 m	Low	Low	Low	> 250 m	None	452
50 to 100 m	not-suitable	High	High	< 250 m	Fire	601
50 to 100 m	not-suitable	High	High	< 250 m	None	552
50 to 100 m	not-suitable	Moderate	High	< 250 m	Fire	551
50 to 100 m	not-suitable	High	Moderate	< 250 m	Fire	551
50 to 100 m	not-suitable	Moderate	High	< 250 m	None	502
50 to 100 m	not-suitable	High	Moderate	< 250 m	None	502
50 to 100 m	not-suitable	Low	High	< 250 m	Fire	501
50 to 100 m	not-suitable	Moderate	Moderate	< 250 m	Fire	501
50 to 100 m	not-suitable	High	Low	< 250 m	Fire	501
50 to 100 m	not-suitable	Low	High	< 250 m	None	452
50 to 100 m	not-suitable	Moderate	Moderate	< 250 m	None	452
50 to 100 m	not-suitable	High	Low	< 250 m	None	452
50 to 100 m	not-suitable	not-suitable	High	< 250 m	Fire	452
50 to 100 m	not-suitable	High	None	< 250 m	Fire	452

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	not-suitable	Low	Moderate	< 250 m	Fire	451
50 to 100 m	not-suitable	Moderate	Low	< 250 m	Fire	451
50 to 100 m	not-suitable	High	High	> 250 m	Fire	552
50 to 100 m	not-suitable	High	High	> 250 m	None	503
50 to 100 m	not-suitable	Moderate	High	> 250 m	Fire	502
50 to 100 m	not-suitable	High	Moderate	> 250 m	Fire	502
50 to 100 m	not-suitable	Moderate	High	> 250 m	None	453
50 to 100 m	not-suitable	High	Moderate	> 250 m	None	453
50 to 100 m	not-suitable	Low	High	> 250 m	Fire	452
50 to 100 m	not-suitable	Moderate	Moderate	> 250 m	Fire	452
50 to 100 m	not-suitable	High	Low	> 250 m	Fire	452
100 to 150 m	High	High	High	< 250 m	Fire	800
100 to 150 m	High	High	High	< 250 m	None	751
100 to 150 m	High	Moderate	High	< 250 m	Fire	750
100 to 150 m	High	High	Moderate	< 250 m	Fire	750
100 to 150 m	High	Moderate	High	< 250 m	None	701
100 to 150 m	High	High	Moderate	< 250 m	None	701
100 to 150 m	High	Low	High	< 250 m	Fire	700
100 to 150 m	High	Moderate	Moderate	< 250 m	Fire	700
100 to 150 m	High	High	Low	< 250 m	Fire	700
100 to 150 m	High	Low	High	< 250 m	None	651
100 to 150 m	High	Moderate	Moderate	< 250 m	None	651
100 to 150 m	High	High	Low	< 250 m	None	651
100 to 150 m	High	not-suitable	High	< 250 m	Fire	651
100 to 150 m	High	High	None	< 250 m	Fire	651
100 to 150 m	High	Low	Moderate	< 250 m	Fire	650
100 to 150 m	High	Moderate	Low	< 250 m	Fire	650
100 to 150 m	High	not-suitable	High	< 250 m	None	602
100 to 150 m	High	High	None	< 250 m	None	602
100 to 150 m	High	Low	Moderate	< 250 m	None	601
100 to 150 m	High	Moderate	Low	< 250 m	None	601
100 to 150 m	High	not-suitable	Moderate	< 250 m	Fire	601
100 to 150 m	High	Moderate	None	< 250 m	Fire	601
100 to 150 m	High	Low	Low	< 250 m	Fire	600
100 to 150 m	High	not-suitable	Moderate	< 250 m	None	552
100 to 150 m	High	Moderate	None	< 250 m	None	552
100 to 150 m	High	Low	Low	< 250 m	None	551
100 to 150 m	High	not-suitable	Low	< 250 m	Fire	551
100 to 150 m	High	Low	None	< 250 m	Fire	551
100 to 150 m	High	not-suitable	Low	< 250 m	None	502
100 to 150 m	High	Low	None	< 250 m	None	502
100 to 150 m	High	not-suitable	None	< 250 m	Fire	502
100 to 150 m	High	not-suitable	None	< 250 m	None	453
100 to 150 m	High	High	High	> 250 m	Fire	751
100 to 150 m	High	High	High	> 250 m	None	702
100 to 150 m	High	Moderate	High	> 250 m	Fire	701
100 to 150 m	High	High	Moderate	> 250 m	Fire	701
100 to 150 m	High	Moderate	High	> 250 m	None	652

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
100 to 150 m	High	High	Moderate	> 250 m	None	652
100 to 150 m	High	Low	High	> 250 m	Fire	651
100 to 150 m	High	Moderate	Moderate	> 250 m	Fire	651
100 to 150 m	High	High	Low	> 250 m	Fire	651
100 to 150 m	High	Low	High	> 250 m	None	602
100 to 150 m	High	Moderate	Moderate	> 250 m	None	602
100 to 150 m	High	High	Low	> 250 m	None	602
100 to 150 m	High	not-suitable	High	> 250 m	Fire	602
100 to 150 m	High	High	None	> 250 m	Fire	602
100 to 150 m	High	Low	Moderate	> 250 m	Fire	601
100 to 150 m	High	Moderate	Low	> 250 m	Fire	601
100 to 150 m	High	not-suitable	High	> 250 m	None	553
100 to 150 m	High	High	None	> 250 m	None	553
100 to 150 m	High	Low	Moderate	> 250 m	None	552
100 to 150 m	High	Moderate	Low	> 250 m	None	552
100 to 150 m	High	not-suitable	Moderate	> 250 m	Fire	552
100 to 150 m	High	Moderate	None	> 250 m	Fire	552
100 to 150 m	High	Low	Low	> 250 m	Fire	551
100 to 150 m	High	not-suitable	Moderate	> 250 m	None	503
100 to 150 m	High	Moderate	None	> 250 m	None	503
100 to 150 m	High	Low	Low	> 250 m	None	502
100 to 150 m	High	not-suitable	Low	> 250 m	Fire	502
100 to 150 m	High	Low	None	> 250 m	Fire	502
100 to 150 m	High	not-suitable	Low	> 250 m	None	453
100 to 150 m	High	Low	None	> 250 m	None	453
100 to 150 m	High	not-suitable	None	> 250 m	Fire	453
100 to 150 m	Moderate	High	High	< 250 m	Fire	750
100 to 150 m	Moderate	Moderate	High	< 250 m	Fire	700
100 to 150 m	Moderate	High	Moderate	< 250 m	Fire	700
100 to 150 m	Moderate	Low	High	< 250 m	Fire	650
100 to 150 m	Moderate	Moderate	Moderate	< 250 m	Fire	650
100 to 150 m	Moderate	High	Low	< 250 m	Fire	650
100 to 150 m	Moderate	not-suitable	High	< 250 m	Fire	601
100 to 150 m	Moderate	High	None	< 250 m	Fire	601
100 to 150 m	Moderate	Low	Moderate	< 250 m	Fire	600
100 to 150 m	Moderate	Moderate	Low	< 250 m	Fire	600
100 to 150 m	Moderate	not-suitable	Moderate	< 250 m	Fire	551
100 to 150 m	Moderate	Moderate	None	< 250 m	Fire	551
100 to 150 m	Moderate	Low	Low	< 250 m	Fire	550
100 to 150 m	Moderate	not-suitable	Low	< 250 m	Fire	501
100 to 150 m	Moderate	Low	None	< 250 m	Fire	501
100 to 150 m	Moderate	not-suitable	None	< 250 m	Fire	452
100 to 150 m	Moderate	High	High	< 250 m	None	701
100 to 150 m	Moderate	Moderate	High	< 250 m	None	651
100 to 150 m	Moderate	High	Moderate	< 250 m	None	651
100 to 150 m	Moderate	Low	High	< 250 m	None	601
100 to 150 m	Moderate	Moderate	Moderate	< 250 m	None	601
100 to 150 m	Moderate	High	Low	< 250 m	None	601

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
100 to 150 m	Moderate	not-suitable	High	< 250 m	None	552
100 to 150 m	Moderate	High	None	< 250 m	None	552
100 to 150 m	Moderate	Low	Moderate	< 250 m	None	551
100 to 150 m	Moderate	Moderate	Low	< 250 m	None	551
100 to 150 m	Moderate	not-suitable	Moderate	< 250 m	None	502
100 to 150 m	Moderate	Moderate	None	< 250 m	None	502
100 to 150 m	Moderate	Low	Low	< 250 m	None	501
100 to 150 m	Moderate	not-suitable	Low	< 250 m	None	452
100 to 150 m	Moderate	Low	None	< 250 m	None	452
100 to 150 m	Moderate	High	High	> 250 m	Fire	701
100 to 150 m	Moderate	Moderate	High	> 250 m	Fire	651
100 to 150 m	Moderate	High	Moderate	> 250 m	Fire	651
100 to 150 m	Moderate	Low	High	> 250 m	Fire	601
100 to 150 m	Moderate	Moderate	Moderate	> 250 m	Fire	601
100 to 150 m	Moderate	High	Low	> 250 m	Fire	601
100 to 150 m	Moderate	not-suitable	High	> 250 m	Fire	552
100 to 150 m	Moderate	High	None	> 250 m	Fire	552
100 to 150 m	Moderate	Low	Moderate	> 250 m	Fire	551
100 to 150 m	Moderate	Moderate	Low	> 250 m	Fire	551
100 to 150 m	Moderate	not-suitable	Moderate	> 250 m	Fire	502
100 to 150 m	Moderate	Moderate	None	> 250 m	Fire	502
100 to 150 m	Moderate	Low	Low	> 250 m	Fire	501
100 to 150 m	Moderate	not-suitable	Low	> 250 m	Fire	452
100 to 150 m	Moderate	Low	None	> 250 m	Fire	452
100 to 150 m	Moderate	High	High	> 250 m	None	652
100 to 150 m	Moderate	Moderate	High	> 250 m	None	602
100 to 150 m	Moderate	High	Moderate	> 250 m	None	602
100 to 150 m	Moderate	Low	High	> 250 m	None	552
100 to 150 m	Moderate	Moderate	Moderate	> 250 m	None	552
100 to 150 m	Moderate	High	Low	> 250 m	None	552
100 to 150 m	Moderate	not-suitable	High	> 250 m	None	503
100 to 150 m	Moderate	High	None	> 250 m	None	503
100 to 150 m	Moderate	Low	Moderate	> 250 m	None	502
100 to 150 m	Moderate	Moderate	Low	> 250 m	None	502
100 to 150 m	Moderate	not-suitable	Moderate	> 250 m	None	453
100 to 150 m	Moderate	Moderate	None	> 250 m	None	453
100 to 150 m	Moderate	Low	Low	> 250 m	None	452
100 to 150 m	Low	High	High	< 250 m	Fire	700
100 to 150 m	Low	Moderate	High	< 250 m	Fire	650
100 to 150 m	Low	High	Moderate	< 250 m	Fire	650
100 to 150 m	Low	Low	High	< 250 m	Fire	600
100 to 150 m	Low	Moderate	Moderate	< 250 m	Fire	600
100 to 150 m	Low	High	Low	< 250 m	Fire	600
100 to 150 m	Low	not-suitable	High	< 250 m	Fire	551
100 to 150 m	Low	High	None	< 250 m	Fire	551
100 to 150 m	Low	Low	Moderate	< 250 m	Fire	550
100 to 150 m	Low	Moderate	Low	< 250 m	Fire	550
100 to 150 m	Low	not-suitable	Moderate	< 250 m	Fire	501

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
100 to 150 m	Low	Moderate	None	< 250 m	Fire	501
100 to 150 m	Low	Low	Low	< 250 m	Fire	500
100 to 150 m	Low	not-suitable	Low	< 250 m	Fire	451
100 to 150 m	Low	Low	None	< 250 m	Fire	451
100 to 150 m	Low	High	High	< 250 m	None	651
100 to 150 m	Low	Moderate	High	< 250 m	None	601
100 to 150 m	Low	High	Moderate	< 250 m	None	601
100 to 150 m	Low	Low	High	< 250 m	None	551
100 to 150 m	Low	Moderate	Moderate	< 250 m	None	551
100 to 150 m	Low	High	Low	< 250 m	None	551
100 to 150 m	Low	not-suitable	High	< 250 m	None	502
100 to 150 m	Low	High	None	< 250 m	None	502
100 to 150 m	Low	Low	Moderate	< 250 m	None	501
100 to 150 m	Low	Moderate	Low	< 250 m	None	501
100 to 150 m	Low	not-suitable	Moderate	< 250 m	None	452
100 to 150 m	Low	Moderate	None	< 250 m	None	452
100 to 150 m	Low	Low	Low	< 250 m	None	451
100 to 150 m	Low	High	High	> 250 m	Fire	651
100 to 150 m	Low	Moderate	High	> 250 m	Fire	601
100 to 150 m	Low	High	Moderate	> 250 m	Fire	601
100 to 150 m	Low	Low	High	> 250 m	Fire	551
100 to 150 m	Low	Moderate	Moderate	> 250 m	Fire	551
100 to 150 m	Low	High	Low	> 250 m	Fire	551
100 to 150 m	Low	not-suitable	High	> 250 m	Fire	502
100 to 150 m	Low	High	None	> 250 m	Fire	502
100 to 150 m	Low	Low	Moderate	> 250 m	Fire	501
100 to 150 m	Low	Moderate	Low	> 250 m	Fire	501
100 to 150 m	Low	not-suitable	Moderate	> 250 m	Fire	452
100 to 150 m	Low	Moderate	None	> 250 m	Fire	452
100 to 150 m	Low	Low	Low	> 250 m	Fire	451
100 to 150 m	Low	High	High	> 250 m	None	602
100 to 150 m	Low	Moderate	High	> 250 m	None	552
100 to 150 m	Low	High	Moderate	> 250 m	None	552
100 to 150 m	Low	Low	High	> 250 m	None	502
100 to 150 m	Low	Moderate	Moderate	> 250 m	None	502
100 to 150 m	Low	High	Low	> 250 m	None	502
100 to 150 m	Low	not-suitable	High	> 250 m	None	453
100 to 150 m	Low	High	None	> 250 m	None	453
100 to 150 m	Low	Low	Moderate	> 250 m	None	452
100 to 150 m	Low	Moderate	Low	> 250 m	None	452
100 to 150 m	not-suitable	High	High	< 250 m	Fire	551
100 to 150 m	not-suitable	High	High	< 250 m	None	502
100 to 150 m	not-suitable	Moderate	High	< 250 m	Fire	501
100 to 150 m	not-suitable	High	Moderate	< 250 m	Fire	501
100 to 150 m	not-suitable	Moderate	High	< 250 m	None	452
100 to 150 m	not-suitable	High	Moderate	< 250 m	None	452
100 to 150 m	not-suitable	Low	High	< 250 m	Fire	451
100 to 150 m	not-suitable	Moderate	Moderate	< 250 m	Fire	451

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
100 to 150 m	not-suitable	High	Low	< 250 m	Fire	451
100 to 150 m	not-suitable	High	High	> 250 m	Fire	502
100 to 150 m	not-suitable	High	High	> 250 m	None	453
100 to 150 m	not-suitable	Moderate	High	> 250 m	Fire	452
100 to 150 m	not-suitable	High	Moderate	> 250 m	Fire	452
150 to 200 m	High	High	High	< 250 m	Fire	750
150 to 200 m	High	High	High	< 250 m	None	701
150 to 200 m	High	Moderate	High	< 250 m	Fire	700
150 to 200 m	High	High	Moderate	< 250 m	Fire	700
150 to 200 m	High	Moderate	High	< 250 m	None	651
150 to 200 m	High	High	Moderate	< 250 m	None	651
150 to 200 m	High	Low	High	< 250 m	Fire	650
150 to 200 m	High	Moderate	Moderate	< 250 m	Fire	650
150 to 200 m	High	High	Low	< 250 m	Fire	650
150 to 200 m	High	Low	High	< 250 m	None	601
150 to 200 m	High	Moderate	Moderate	< 250 m	None	601
150 to 200 m	High	High	Low	< 250 m	None	601
150 to 200 m	High	not-suitable	High	< 250 m	Fire	601
150 to 200 m	High	High	None	< 250 m	Fire	601
150 to 200 m	High	Low	Moderate	< 250 m	Fire	600
150 to 200 m	High	Moderate	Low	< 250 m	Fire	600
150 to 200 m	High	not-suitable	High	< 250 m	None	552
150 to 200 m	High	High	None	< 250 m	None	552
150 to 200 m	High	Low	Moderate	< 250 m	None	551
150 to 200 m	High	Moderate	Low	< 250 m	None	551
150 to 200 m	High	not-suitable	Moderate	< 250 m	Fire	551
150 to 200 m	High	Moderate	None	< 250 m	Fire	551
150 to 200 m	High	Low	Low	< 250 m	Fire	550
150 to 200 m	High	not-suitable	Moderate	< 250 m	None	502
150 to 200 m	High	Moderate	None	< 250 m	None	502
150 to 200 m	High	Low	Low	< 250 m	None	501
150 to 200 m	High	not-suitable	Low	< 250 m	Fire	501
150 to 200 m	High	Low	None	< 250 m	Fire	501
150 to 200 m	High	not-suitable	Low	< 250 m	None	452
150 to 200 m	High	Low	None	< 250 m	None	452
150 to 200 m	High	not-suitable	None	< 250 m	Fire	452
150 to 200 m	High	High	High	> 250 m	Fire	701
150 to 200 m	High	High	High	> 250 m	None	652
150 to 200 m	High	Moderate	High	> 250 m	Fire	651
150 to 200 m	High	High	Moderate	> 250 m	Fire	651
150 to 200 m	High	Moderate	High	> 250 m	None	602
150 to 200 m	High	High	Moderate	> 250 m	None	602
150 to 200 m	High	Low	High	> 250 m	Fire	601
150 to 200 m	High	Moderate	Moderate	> 250 m	Fire	601
150 to 200 m	High	High	Low	> 250 m	Fire	601
150 to 200 m	High	Low	High	> 250 m	None	552
150 to 200 m	High	Moderate	Moderate	> 250 m	None	552
150 to 200 m	High	High	Low	> 250 m	None	552

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
150 to 200 m	High	not-suitable	High	> 250 m	Fire	552
150 to 200 m	High	High	None	> 250 m	Fire	552
150 to 200 m	High	Low	Moderate	> 250 m	Fire	551
150 to 200 m	High	Moderate	Low	> 250 m	Fire	551
150 to 200 m	High	not-suitable	High	> 250 m	None	503
150 to 200 m	High	High	None	> 250 m	None	503
150 to 200 m	High	Low	Moderate	> 250 m	None	502
150 to 200 m	High	Moderate	Low	> 250 m	None	502
150 to 200 m	High	not-suitable	Moderate	> 250 m	Fire	502
150 to 200 m	High	Moderate	None	> 250 m	Fire	502
150 to 200 m	High	Low	Low	> 250 m	Fire	501
150 to 200 m	High	not-suitable	Moderate	> 250 m	None	453
150 to 200 m	High	Moderate	None	> 250 m	None	453
150 to 200 m	High	Low	Low	> 250 m	None	452
150 to 200 m	High	not-suitable	Low	> 250 m	Fire	452
150 to 200 m	High	Low	None	> 250 m	Fire	452
150 to 200 m	Moderate	High	High	< 250 m	Fire	700
150 to 200 m	Moderate	Moderate	High	< 250 m	Fire	650
150 to 200 m	Moderate	High	Moderate	< 250 m	Fire	650
150 to 200 m	Moderate	Low	High	< 250 m	Fire	600
150 to 200 m	Moderate	Moderate	Moderate	< 250 m	Fire	600
150 to 200 m	Moderate	High	Low	< 250 m	Fire	600
150 to 200 m	Moderate	not-suitable	High	< 250 m	Fire	551
150 to 200 m	Moderate	High	None	< 250 m	Fire	551
150 to 200 m	Moderate	Low	Moderate	< 250 m	Fire	550
150 to 200 m	Moderate	Moderate	Low	< 250 m	Fire	550
150 to 200 m	Moderate	not-suitable	Moderate	< 250 m	Fire	501
150 to 200 m	Moderate	Moderate	None	< 250 m	Fire	501
150 to 200 m	Moderate	Low	Low	< 250 m	Fire	500
150 to 200 m	Moderate	not-suitable	Low	< 250 m	Fire	451
150 to 200 m	Moderate	Low	None	< 250 m	Fire	451
150 to 200 m	Moderate	High	High	< 250 m	None	651
150 to 200 m	Moderate	Moderate	High	< 250 m	None	601
150 to 200 m	Moderate	High	Moderate	< 250 m	None	601
150 to 200 m	Moderate	Low	High	< 250 m	None	551
150 to 200 m	Moderate	Moderate	Moderate	< 250 m	None	551
150 to 200 m	Moderate	High	Low	< 250 m	None	551
150 to 200 m	Moderate	not-suitable	High	< 250 m	None	502
150 to 200 m	Moderate	High	None	< 250 m	None	502
150 to 200 m	Moderate	Low	Moderate	< 250 m	None	501
150 to 200 m	Moderate	Moderate	Low	< 250 m	None	501
150 to 200 m	Moderate	not-suitable	Moderate	< 250 m	None	452
150 to 200 m	Moderate	Moderate	None	< 250 m	None	452
150 to 200 m	Moderate	Low	Low	< 250 m	None	451
150 to 200 m	Moderate	High	High	> 250 m	Fire	651
150 to 200 m	Moderate	Moderate	High	> 250 m	Fire	601
150 to 200 m	Moderate	High	Moderate	> 250 m	Fire	601
150 to 200 m	Moderate	Low	High	> 250 m	Fire	551

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
150 to 200 m	Moderate	Moderate	Moderate	> 250 m	Fire	551
150 to 200 m	Moderate	High	Low	> 250 m	Fire	551
150 to 200 m	Moderate	not-suitable	High	> 250 m	Fire	502
150 to 200 m	Moderate	High	None	> 250 m	Fire	502
150 to 200 m	Moderate	Low	Moderate	> 250 m	Fire	501
150 to 200 m	Moderate	Moderate	Low	> 250 m	Fire	501
150 to 200 m	Moderate	not-suitable	Moderate	> 250 m	Fire	452
150 to 200 m	Moderate	Moderate	None	> 250 m	Fire	452
150 to 200 m	Moderate	Low	Low	> 250 m	Fire	451
150 to 200 m	Moderate	High	High	> 250 m	None	602
150 to 200 m	Moderate	Moderate	High	> 250 m	None	552
150 to 200 m	Moderate	High	Moderate	> 250 m	None	552
150 to 200 m	Moderate	Low	High	> 250 m	None	502
150 to 200 m	Moderate	Moderate	Moderate	> 250 m	None	502
150 to 200 m	Moderate	High	Low	> 250 m	None	502
150 to 200 m	Moderate	not-suitable	High	> 250 m	None	453
150 to 200 m	Moderate	High	None	> 250 m	None	453
150 to 200 m	Moderate	Low	Moderate	> 250 m	None	452
150 to 200 m	Moderate	Moderate	Low	> 250 m	None	452
150 to 200 m	Low	High	High	< 250 m	Fire	650
150 to 200 m	Low	Moderate	High	< 250 m	Fire	600
150 to 200 m	Low	High	Moderate	< 250 m	Fire	600
150 to 200 m	Low	Low	High	< 250 m	Fire	550
150 to 200 m	Low	Moderate	Moderate	< 250 m	Fire	550
150 to 200 m	Low	High	Low	< 250 m	Fire	550
150 to 200 m	Low	not-suitable	High	< 250 m	Fire	501
150 to 200 m	Low	High	None	< 250 m	Fire	501
150 to 200 m	Low	Low	Moderate	< 250 m	Fire	500
150 to 200 m	Low	Moderate	Low	< 250 m	Fire	500
150 to 200 m	Low	not-suitable	Moderate	< 250 m	Fire	451
150 to 200 m	Low	Moderate	None	< 250 m	Fire	451
150 to 200 m	Low	Low	Low	< 250 m	Fire	450
150 to 200 m	Low	High	High	< 250 m	None	601
150 to 200 m	Low	Moderate	High	< 250 m	None	551
150 to 200 m	Low	High	Moderate	< 250 m	None	551
150 to 200 m	Low	Low	High	< 250 m	None	501
150 to 200 m	Low	Moderate	Moderate	< 250 m	None	501
150 to 200 m	Low	High	Low	< 250 m	None	501
150 to 200 m	Low	not-suitable	High	< 250 m	None	452
150 to 200 m	Low	High	None	< 250 m	None	452
150 to 200 m	Low	Low	Moderate	< 250 m	None	451
150 to 200 m	Low	Moderate	Low	< 250 m	None	451
150 to 200 m	Low	High	High	> 250 m	Fire	601
150 to 200 m	Low	Moderate	High	> 250 m	Fire	551
150 to 200 m	Low	High	Moderate	> 250 m	Fire	551
150 to 200 m	Low	Low	High	> 250 m	Fire	501
150 to 200 m	Low	Moderate	Moderate	> 250 m	Fire	501
150 to 200 m	Low	High	Low	> 250 m	Fire	501

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
150 to 200 m	Low	not-suitable	High	> 250 m	Fire	452
150 to 200 m	Low	High	None	> 250 m	Fire	452
150 to 200 m	Low	Low	Moderate	> 250 m	Fire	451
150 to 200 m	Low	Moderate	Low	> 250 m	Fire	451
150 to 200 m	Low	High	High	> 250 m	None	552
150 to 200 m	Low	Moderate	High	> 250 m	None	502
150 to 200 m	Low	High	Moderate	> 250 m	None	502
150 to 200 m	Low	Low	High	> 250 m	None	452
150 to 200 m	Low	Moderate	Moderate	> 250 m	None	452
150 to 200 m	Low	High	Low	> 250 m	None	452
150 to 200 m	not-suitable	High	High	< 250 m	Fire	501
150 to 200 m	not-suitable	High	High	< 250 m	None	452
150 to 200 m	not-suitable	Moderate	High	< 250 m	Fire	451
150 to 200 m	not-suitable	High	Moderate	< 250 m	Fire	451
150 to 200 m	not-suitable	High	High	> 250 m	Fire	452
200 to 250 m	High	High	High	< 250 m	Fire	700
200 to 250 m	High	High	High	< 250 m	None	651
200 to 250 m	High	Moderate	High	< 250 m	Fire	650
200 to 250 m	High	High	Moderate	< 250 m	Fire	650
200 to 250 m	High	Moderate	High	< 250 m	None	601
200 to 250 m	High	High	Moderate	< 250 m	None	601
200 to 250 m	High	Low	High	< 250 m	Fire	600
200 to 250 m	High	Moderate	Moderate	< 250 m	Fire	600
200 to 250 m	High	High	Low	< 250 m	Fire	600
200 to 250 m	High	Low	High	< 250 m	None	551
200 to 250 m	High	Moderate	Moderate	< 250 m	None	551
200 to 250 m	High	High	Low	< 250 m	None	551
200 to 250 m	High	not-suitable	High	< 250 m	Fire	551
200 to 250 m	High	High	None	< 250 m	Fire	551
200 to 250 m	High	Low	Moderate	< 250 m	Fire	550
200 to 250 m	High	Moderate	Low	< 250 m	Fire	550
200 to 250 m	High	not-suitable	High	< 250 m	None	502
200 to 250 m	High	High	None	< 250 m	None	502
200 to 250 m	High	Low	Moderate	< 250 m	None	501
200 to 250 m	High	Moderate	Low	< 250 m	None	501
200 to 250 m	High	not-suitable	Moderate	< 250 m	Fire	501
200 to 250 m	High	Moderate	None	< 250 m	Fire	501
200 to 250 m	High	Low	Low	< 250 m	Fire	500
200 to 250 m	High	not-suitable	Moderate	< 250 m	None	452
200 to 250 m	High	Moderate	None	< 250 m	None	452
200 to 250 m	High	Low	Low	< 250 m	None	451
200 to 250 m	High	not-suitable	Low	< 250 m	Fire	451
200 to 250 m	High	Low	None	< 250 m	Fire	451
200 to 250 m	High	High	High	> 250 m	Fire	651
200 to 250 m	High	High	High	> 250 m	None	602
200 to 250 m	High	Moderate	High	> 250 m	Fire	601
200 to 250 m	High	High	Moderate	> 250 m	Fire	601
200 to 250 m	High	Moderate	High	> 250 m	None	552

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
200 to 250 m	High	High	Moderate	> 250 m	None	552
200 to 250 m	High	Low	High	> 250 m	Fire	551
200 to 250 m	High	Moderate	Moderate	> 250 m	Fire	551
200 to 250 m	High	High	Low	> 250 m	Fire	551
200 to 250 m	High	Low	High	> 250 m	None	502
200 to 250 m	High	Moderate	Moderate	> 250 m	None	502
200 to 250 m	High	High	Low	> 250 m	None	502
200 to 250 m	High	not-suitable	High	> 250 m	Fire	502
200 to 250 m	High	High	None	> 250 m	Fire	502
200 to 250 m	High	Low	Moderate	> 250 m	Fire	501
200 to 250 m	High	Moderate	Low	> 250 m	Fire	501
200 to 250 m	High	not-suitable	High	> 250 m	None	453
200 to 250 m	High	High	None	> 250 m	None	453
200 to 250 m	High	Low	Moderate	> 250 m	None	452
200 to 250 m	High	Moderate	Low	> 250 m	None	452
200 to 250 m	High	not-suitable	Moderate	> 250 m	Fire	452
200 to 250 m	High	Moderate	None	> 250 m	Fire	452
200 to 250 m	High	Low	Low	> 250 m	Fire	451
200 to 250 m	Moderate	High	High	< 250 m	Fire	650
200 to 250 m	Moderate	Moderate	High	< 250 m	Fire	600
200 to 250 m	Moderate	High	Moderate	< 250 m	Fire	600
200 to 250 m	Moderate	Low	High	< 250 m	Fire	550
200 to 250 m	Moderate	Moderate	Moderate	< 250 m	Fire	550
200 to 250 m	Moderate	High	Low	< 250 m	Fire	550
200 to 250 m	Moderate	not-suitable	High	< 250 m	Fire	501
200 to 250 m	Moderate	High	None	< 250 m	Fire	501
200 to 250 m	Moderate	Low	Moderate	< 250 m	Fire	500
200 to 250 m	Moderate	Moderate	Low	< 250 m	Fire	500
200 to 250 m	Moderate	not-suitable	Moderate	< 250 m	Fire	451
200 to 250 m	Moderate	Moderate	None	< 250 m	Fire	451
200 to 250 m	Moderate	Low	Low	< 250 m	Fire	450
200 to 250 m	Moderate	High	High	< 250 m	None	601
200 to 250 m	Moderate	Moderate	High	< 250 m	None	551
200 to 250 m	Moderate	High	Moderate	< 250 m	None	551
200 to 250 m	Moderate	Low	High	< 250 m	None	501
200 to 250 m	Moderate	Moderate	Moderate	< 250 m	None	501
200 to 250 m	Moderate	High	Low	< 250 m	None	501
200 to 250 m	Moderate	not-suitable	High	< 250 m	None	452
200 to 250 m	Moderate	High	None	< 250 m	None	452
200 to 250 m	Moderate	Low	Moderate	< 250 m	None	451
200 to 250 m	Moderate	Moderate	Low	< 250 m	None	451
200 to 250 m	Moderate	High	High	> 250 m	Fire	601
200 to 250 m	Moderate	Moderate	High	> 250 m	Fire	551
200 to 250 m	Moderate	High	Moderate	> 250 m	Fire	551
200 to 250 m	Moderate	Low	High	> 250 m	Fire	501
200 to 250 m	Moderate	Moderate	Moderate	> 250 m	Fire	501
200 to 250 m	Moderate	High	Low	> 250 m	Fire	501
200 to 250 m	Moderate	not-suitable	High	> 250 m	Fire	452

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
200 to 250 m	Moderate	High	None	> 250 m	Fire	452
200 to 250 m	Moderate	Low	Moderate	> 250 m	Fire	451
200 to 250 m	Moderate	Moderate	Low	> 250 m	Fire	451
200 to 250 m	Moderate	High	High	> 250 m	None	552
200 to 250 m	Moderate	Moderate	High	> 250 m	None	502
200 to 250 m	Moderate	High	Moderate	> 250 m	None	502
200 to 250 m	Moderate	Low	High	> 250 m	None	452
200 to 250 m	Moderate	Moderate	Moderate	> 250 m	None	452
200 to 250 m	Moderate	High	Low	> 250 m	None	452
200 to 250 m	Low	High	High	< 250 m	Fire	600
200 to 250 m	Low	Moderate	High	< 250 m	Fire	550
200 to 250 m	Low	High	Moderate	< 250 m	Fire	550
200 to 250 m	Low	Low	High	< 250 m	Fire	500
200 to 250 m	Low	Moderate	Moderate	< 250 m	Fire	500
200 to 250 m	Low	High	Low	< 250 m	Fire	500
200 to 250 m	Low	not-suitable	High	< 250 m	Fire	451
200 to 250 m	Low	High	None	< 250 m	Fire	451
200 to 250 m	Low	Low	Moderate	< 250 m	Fire	450
200 to 250 m	Low	Moderate	Low	< 250 m	Fire	450
200 to 250 m	Low	High	High	< 250 m	None	551
200 to 250 m	Low	Moderate	High	< 250 m	None	501
200 to 250 m	Low	High	Moderate	< 250 m	None	501
200 to 250 m	Low	Low	High	< 250 m	None	451
200 to 250 m	Low	Moderate	Moderate	< 250 m	None	451
200 to 250 m	Low	High	Low	< 250 m	None	451
200 to 250 m	Low	High	High	> 250 m	Fire	551
200 to 250 m	Low	Moderate	High	> 250 m	Fire	501
200 to 250 m	Low	High	Moderate	> 250 m	Fire	501
200 to 250 m	Low	Low	High	> 250 m	Fire	451
200 to 250 m	Low	Moderate	Moderate	> 250 m	Fire	451
200 to 250 m	Low	High	Low	> 250 m	Fire	451
200 to 250 m	Low	High	High	> 250 m	None	502
200 to 250 m	Low	Moderate	High	> 250 m	None	452
200 to 250 m	Low	High	Moderate	> 250 m	None	452
200 to 250 m	not-suitable	High	High	< 250 m	Fire	451
> 250 m	High	High	High	< 250 m	Fire	651
> 250 m	High	High	High	< 250 m	None	602
> 250 m	High	Moderate	High	< 250 m	Fire	601
> 250 m	High	High	Moderate	< 250 m	Fire	601
> 250 m	High	Moderate	High	< 250 m	None	552
> 250 m	High	High	Moderate	< 250 m	None	552
> 250 m	High	Low	High	< 250 m	Fire	551
> 250 m	High	Moderate	Moderate	< 250 m	Fire	551
> 250 m	High	High	Low	< 250 m	Fire	551
> 250 m	High	Low	High	< 250 m	None	502
> 250 m	High	Moderate	Moderate	< 250 m	None	502
> 250 m	High	High	Low	< 250 m	None	502
> 250 m	High	not-suitable	High	< 250 m	Fire	502

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
> 250 m	High	High	None	< 250 m	Fire	502
> 250 m	High	Low	Moderate	< 250 m	Fire	501
> 250 m	High	Moderate	Low	< 250 m	Fire	501
> 250 m	High	not-suitable	High	< 250 m	None	453
> 250 m	High	High	None	< 250 m	None	453
> 250 m	High	Low	Moderate	< 250 m	None	452
> 250 m	High	Moderate	Low	< 250 m	None	452
> 250 m	High	not-suitable	Moderate	< 250 m	Fire	452
> 250 m	High	Moderate	None	< 250 m	Fire	452
> 250 m	High	Low	Low	< 250 m	Fire	451
> 250 m	High	High	High	> 250 m	Fire	602
> 250 m	High	High	High	> 250 m	None	553
> 250 m	High	Moderate	High	> 250 m	Fire	552
> 250 m	High	High	Moderate	> 250 m	Fire	552
> 250 m	High	Moderate	High	> 250 m	None	503
> 250 m	High	High	Moderate	> 250 m	None	503
> 250 m	High	Low	High	> 250 m	Fire	502
> 250 m	High	Moderate	Moderate	> 250 m	Fire	502
> 250 m	High	High	Low	> 250 m	Fire	502
> 250 m	High	Low	High	> 250 m	None	453
> 250 m	High	Moderate	Moderate	> 250 m	None	453
> 250 m	High	High	Low	> 250 m	None	453
> 250 m	High	not-suitable	High	> 250 m	Fire	453
> 250 m	High	High	None	> 250 m	Fire	453
> 250 m	High	Low	Moderate	> 250 m	Fire	452
> 250 m	High	Moderate	Low	> 250 m	Fire	452
> 250 m	Moderate	High	High	< 250 m	Fire	601
> 250 m	Moderate	Moderate	High	< 250 m	Fire	551
> 250 m	Moderate	High	Moderate	< 250 m	Fire	551
> 250 m	Moderate	Low	High	< 250 m	Fire	501
> 250 m	Moderate	Moderate	Moderate	< 250 m	Fire	501
> 250 m	Moderate	High	Low	< 250 m	Fire	501
> 250 m	Moderate	not-suitable	High	< 250 m	Fire	452
> 250 m	Moderate	High	None	< 250 m	Fire	452
> 250 m	Moderate	Low	Moderate	< 250 m	Fire	451
> 250 m	Moderate	Moderate	Low	< 250 m	Fire	451
> 250 m	Moderate	High	High	< 250 m	None	552
> 250 m	Moderate	Moderate	High	< 250 m	None	502
> 250 m	Moderate	High	Moderate	< 250 m	None	502
> 250 m	Moderate	Low	High	< 250 m	None	452
> 250 m	Moderate	Moderate	Moderate	< 250 m	None	452
> 250 m	Moderate	High	Low	< 250 m	None	452
> 250 m	Moderate	High	High	> 250 m	Fire	552
> 250 m	Moderate	Moderate	High	> 250 m	Fire	502
> 250 m	Moderate	High	Moderate	> 250 m	Fire	502
> 250 m	Moderate	Low	High	> 250 m	Fire	452
> 250 m	Moderate	Moderate	Moderate	> 250 m	Fire	452
> 250 m	Moderate	High	Low	> 250 m	Fire	452

Appendix F. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
> 250 m	Moderate	High	High	> 250 m	None	503
> 250 m	Moderate	Moderate	High	> 250 m	None	453
> 250 m	Moderate	High	Moderate	> 250 m	None	453
> 250 m	Low	High	High	< 250 m	Fire	551
> 250 m	Low	Moderate	High	< 250 m	Fire	501
> 250 m	Low	High	Moderate	< 250 m	Fire	501
> 250 m	Low	Low	High	< 250 m	Fire	451
> 250 m	Low	Moderate	Moderate	< 250 m	Fire	451
> 250 m	Low	High	Low	< 250 m	Fire	451
> 250 m	Low	High	High	< 250 m	None	502
> 250 m	Low	Moderate	High	< 250 m	None	452
> 250 m	Low	High	Moderate	< 250 m	None	452
> 250 m	Low	High	High	> 250 m	Fire	502
> 250 m	Low	Moderate	High	> 250 m	Fire	452
> 250 m	Low	High	Moderate	> 250 m	Fire	452
> 250 m	Low	High	High	> 250 m	None	453

Appendix G. Possible combinations of model input variables that may constitute moderate risk areas. Only combinations with model values from 270-499 are provided.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
< 50 m	Low	not-suitable	None	> 250 m	None	404
< 50 m	not-suitable	not-suitable	High	> 250 m	None	404
< 50 m	not-suitable	High	None	> 250 m	None	404
< 50 m	not-suitable	Moderate	Low	> 250 m	None	403
< 50 m	not-suitable	Low	Moderate	> 250 m	None	403
< 50 m	not-suitable	not-suitable	Moderate	> 250 m	Fire	403
< 50 m	not-suitable	not-suitable	Moderate	< 250 m	None	403
< 50 m	not-suitable	Moderate	None	> 250 m	Fire	403
< 50 m	not-suitable	Moderate	None	< 250 m	None	403
< 50 m	not-suitable	Low	Low	> 250 m	Fire	402
< 50 m	not-suitable	Low	Low	< 250 m	None	402
< 50 m	not-suitable	not-suitable	Low	< 250 m	Fire	402
< 50 m	not-suitable	Low	None	< 250 m	Fire	402
< 50 m	not-suitable	not-suitable	Moderate	> 250 m	None	354
< 50 m	not-suitable	Moderate	None	> 250 m	None	354
< 50 m	not-suitable	Low	Low	> 250 m	None	353
< 50 m	not-suitable	not-suitable	Low	> 250 m	Fire	353
< 50 m	not-suitable	not-suitable	Low	< 250 m	None	353
< 50 m	not-suitable	Low	None	> 250 m	Fire	353
< 50 m	not-suitable	Low	None	< 250 m	None	353
< 50 m	not-suitable	not-suitable	None	< 250 m	Fire	353
< 50 m	not-suitable	not-suitable	Low	> 250 m	None	304
< 50 m	not-suitable	Low	None	> 250 m	None	304
< 50 m	not-suitable	not-suitable	None	> 250 m	Fire	304
< 50 m	not-suitable	not-suitable	None	< 250 m	None	304
50 to 100 m	Low	not-suitable	Low	> 250 m	None	403
50 to 100 m	Low	Low	None	> 250 m	None	403
50 to 100 m	Low	not-suitable	None	> 250 m	Fire	403
50 to 100 m	Low	not-suitable	None	< 250 m	None	403
50 to 100 m	Low	not-suitable	None	> 250 m	None	354
50 to 100 m	Moderate	not-suitable	None	> 250 m	None	404
50 to 100 m	not-suitable	Low	High	> 250 m	None	403
50 to 100 m	not-suitable	not-suitable	High	> 250 m	Fire	403
50 to 100 m	not-suitable	not-suitable	High	< 250 m	None	403
50 to 100 m	not-suitable	High	Low	> 250 m	None	403
50 to 100 m	not-suitable	Moderate	Moderate	> 250 m	None	403
50 to 100 m	not-suitable	High	None	> 250 m	Fire	403
50 to 100 m	not-suitable	High	None	< 250 m	None	403
50 to 100 m	not-suitable	Moderate	Low	> 250 m	Fire	402
50 to 100 m	not-suitable	Moderate	Low	< 250 m	None	402
50 to 100 m	not-suitable	Low	Moderate	> 250 m	Fire	402
50 to 100 m	not-suitable	Low	Moderate	< 250 m	None	402
50 to 100 m	not-suitable	not-suitable	Moderate	< 250 m	Fire	402
50 to 100 m	not-suitable	Moderate	None	< 250 m	Fire	402
50 to 100 m	not-suitable	Low	Low	< 250 m	Fire	401
50 to 100 m	not-suitable	not-suitable	High	> 250 m	None	354

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
50 to 100 m	not-suitable	High	None	> 250 m	None	354
50 to 100 m	not-suitable	Moderate	Low	> 250 m	None	353
50 to 100 m	not-suitable	Low	Moderate	> 250 m	None	353
50 to 100 m	not-suitable	not-suitable	Moderate	> 250 m	Fire	353
50 to 100 m	not-suitable	not-suitable	Moderate	< 250 m	None	353
50 to 100 m	not-suitable	Moderate	None	> 250 m	Fire	353
50 to 100 m	not-suitable	Moderate	None	< 250 m	None	353
50 to 100 m	not-suitable	Low	Low	> 250 m	Fire	352
50 to 100 m	not-suitable	Low	Low	< 250 m	None	352
50 to 100 m	not-suitable	not-suitable	Low	< 250 m	Fire	352
50 to 100 m	not-suitable	Low	None	< 250 m	Fire	352
50 to 100 m	not-suitable	not-suitable	Moderate	> 250 m	None	304
50 to 100 m	not-suitable	Moderate	None	> 250 m	None	304
50 to 100 m	not-suitable	Low	Low	> 250 m	None	303
50 to 100 m	not-suitable	not-suitable	Low	> 250 m	Fire	303
50 to 100 m	not-suitable	not-suitable	Low	< 250 m	None	303
50 to 100 m	not-suitable	Low	None	> 250 m	Fire	303
50 to 100 m	not-suitable	Low	None	< 250 m	None	303
50 to 100 m	not-suitable	not-suitable	None	< 250 m	Fire	303
100 to 150 m	High	not-suitable	None	> 250 m	None	404
100 to 150 m	Low	not-suitable	Moderate	> 250 m	None	403
100 to 150 m	Low	Moderate	None	> 250 m	None	403
100 to 150 m	Low	Low	Low	> 250 m	None	402
100 to 150 m	Low	not-suitable	Low	> 250 m	Fire	402
100 to 150 m	Low	not-suitable	Low	< 250 m	None	402
100 to 150 m	Low	Low	None	> 250 m	Fire	402
100 to 150 m	Low	Low	None	< 250 m	None	402
100 to 150 m	Low	not-suitable	None	< 250 m	Fire	402
100 to 150 m	Low	not-suitable	Low	> 250 m	None	353
100 to 150 m	Low	Low	None	> 250 m	None	353
100 to 150 m	Low	not-suitable	None	> 250 m	Fire	353
100 to 150 m	Low	not-suitable	None	< 250 m	None	353
100 to 150 m	Low	not-suitable	None	> 250 m	None	304
100 to 150 m	Moderate	not-suitable	Low	> 250 m	None	403
100 to 150 m	Moderate	Low	None	> 250 m	None	403
100 to 150 m	Moderate	not-suitable	None	> 250 m	Fire	403
100 to 150 m	Moderate	not-suitable	None	< 250 m	None	403
100 to 150 m	Moderate	not-suitable	None	> 250 m	None	354
100 to 150 m	not-suitable	Moderate	High	> 250 m	None	403
100 to 150 m	not-suitable	High	Moderate	> 250 m	None	403
100 to 150 m	not-suitable	Low	High	> 250 m	Fire	402
100 to 150 m	not-suitable	Low	High	< 250 m	None	402
100 to 150 m	not-suitable	not-suitable	High	< 250 m	Fire	402
100 to 150 m	not-suitable	High	Low	> 250 m	Fire	402
100 to 150 m	not-suitable	High	Low	< 250 m	None	402
100 to 150 m	not-suitable	Moderate	Moderate	> 250 m	Fire	402
100 to 150 m	not-suitable	Moderate	Moderate	< 250 m	None	402
100 to 150 m	not-suitable	High	None	< 250 m	Fire	402

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
100 to 150 m	not-suitable	Moderate	Low	< 250 m	Fire	401
100 to 150 m	not-suitable	Low	Moderate	< 250 m	Fire	401
100 to 150 m	not-suitable	Low	High	> 250 m	None	353
100 to 150 m	not-suitable	not-suitable	High	> 250 m	Fire	353
100 to 150 m	not-suitable	not-suitable	High	< 250 m	None	353
100 to 150 m	not-suitable	High	Low	> 250 m	None	353
100 to 150 m	not-suitable	Moderate	Moderate	> 250 m	None	353
100 to 150 m	not-suitable	High	None	> 250 m	Fire	353
100 to 150 m	not-suitable	High	None	< 250 m	None	353
100 to 150 m	not-suitable	Moderate	Low	> 250 m	Fire	352
100 to 150 m	not-suitable	Moderate	Low	< 250 m	None	352
100 to 150 m	not-suitable	Low	Moderate	> 250 m	Fire	352
100 to 150 m	not-suitable	Low	Moderate	< 250 m	None	352
100 to 150 m	not-suitable	not-suitable	Moderate	< 250 m	Fire	352
100 to 150 m	not-suitable	Moderate	None	< 250 m	Fire	352
100 to 150 m	not-suitable	Low	Low	< 250 m	Fire	351
100 to 150 m	not-suitable	not-suitable	High	> 250 m	None	304
100 to 150 m	not-suitable	High	None	> 250 m	None	304
100 to 150 m	not-suitable	Moderate	Low	> 250 m	None	303
100 to 150 m	not-suitable	Low	Moderate	> 250 m	None	303
100 to 150 m	not-suitable	not-suitable	Moderate	> 250 m	Fire	303
100 to 150 m	not-suitable	not-suitable	Moderate	< 250 m	None	303
100 to 150 m	not-suitable	Moderate	None	> 250 m	Fire	303
100 to 150 m	not-suitable	Moderate	None	< 250 m	None	303
100 to 150 m	not-suitable	Low	Low	> 250 m	Fire	302
100 to 150 m	not-suitable	Low	Low	< 250 m	None	302
100 to 150 m	not-suitable	not-suitable	Low	< 250 m	Fire	302
100 to 150 m	not-suitable	Low	None	< 250 m	Fire	302
150 to 200 m	High	not-suitable	Low	> 250 m	None	403
150 to 200 m	High	Low	None	> 250 m	None	403
150 to 200 m	High	not-suitable	None	> 250 m	Fire	403
150 to 200 m	High	not-suitable	None	< 250 m	None	403
150 to 200 m	High	not-suitable	None	> 250 m	None	354
150 to 200 m	Low	not-suitable	High	> 250 m	None	403
150 to 200 m	Low	High	None	> 250 m	None	403
150 to 200 m	Low	Moderate	Low	> 250 m	None	402
150 to 200 m	Low	Low	Moderate	> 250 m	None	402
150 to 200 m	Low	not-suitable	Moderate	> 250 m	Fire	402
150 to 200 m	Low	not-suitable	Moderate	< 250 m	None	402
150 to 200 m	Low	Moderate	None	> 250 m	Fire	402
150 to 200 m	Low	Moderate	None	< 250 m	None	402
150 to 200 m	Low	Low	Low	> 250 m	Fire	401
150 to 200 m	Low	Low	Low	< 250 m	None	401
150 to 200 m	Low	not-suitable	Low	< 250 m	Fire	401
150 to 200 m	Low	Low	None	< 250 m	Fire	401
150 to 200 m	Low	not-suitable	Moderate	> 250 m	None	353
150 to 200 m	Low	Moderate	None	> 250 m	None	353
150 to 200 m	Low	Low	Low	> 250 m	None	352

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
150 to 200 m	Low	not-suitable	Low	> 250 m	Fire	352
150 to 200 m	Low	not-suitable	Low	< 250 m	None	352
150 to 200 m	Low	Low	None	> 250 m	Fire	352
150 to 200 m	Low	Low	None	< 250 m	None	352
150 to 200 m	Low	not-suitable	None	< 250 m	Fire	352
150 to 200 m	Low	not-suitable	Low	> 250 m	None	303
150 to 200 m	Low	Low	None	> 250 m	None	303
150 to 200 m	Low	not-suitable	None	> 250 m	Fire	303
150 to 200 m	Low	not-suitable	None	< 250 m	None	303
150 to 200 m	Moderate	not-suitable	Moderate	> 250 m	None	403
150 to 200 m	Moderate	Moderate	None	> 250 m	None	403
150 to 200 m	Moderate	Low	Low	> 250 m	None	402
150 to 200 m	Moderate	not-suitable	Low	> 250 m	Fire	402
150 to 200 m	Moderate	not-suitable	Low	< 250 m	None	402
150 to 200 m	Moderate	Low	None	> 250 m	Fire	402
150 to 200 m	Moderate	Low	None	< 250 m	None	402
150 to 200 m	Moderate	not-suitable	None	< 250 m	Fire	402
150 to 200 m	Moderate	not-suitable	Low	> 250 m	None	353
150 to 200 m	Moderate	Low	None	> 250 m	None	353
150 to 200 m	Moderate	not-suitable	None	> 250 m	Fire	353
150 to 200 m	Moderate	not-suitable	None	< 250 m	None	353
150 to 200 m	Moderate	not-suitable	None	> 250 m	None	304
150 to 200 m	not-suitable	High	High	> 250 m	None	403
150 to 200 m	not-suitable	Moderate	High	> 250 m	Fire	402
150 to 200 m	not-suitable	Moderate	High	< 250 m	None	402
150 to 200 m	not-suitable	High	Moderate	> 250 m	Fire	402
150 to 200 m	not-suitable	High	Moderate	< 250 m	None	402
150 to 200 m	not-suitable	Low	High	< 250 m	Fire	401
150 to 200 m	not-suitable	High	Low	< 250 m	Fire	401
150 to 200 m	not-suitable	Moderate	Moderate	< 250 m	Fire	401
150 to 200 m	not-suitable	Moderate	High	> 250 m	None	353
150 to 200 m	not-suitable	High	Moderate	> 250 m	None	353
150 to 200 m	not-suitable	Low	High	> 250 m	Fire	352
150 to 200 m	not-suitable	Low	High	< 250 m	None	352
150 to 200 m	not-suitable	not-suitable	High	< 250 m	Fire	352
150 to 200 m	not-suitable	High	Low	> 250 m	Fire	352
150 to 200 m	not-suitable	High	Low	< 250 m	None	352
150 to 200 m	not-suitable	Moderate	Moderate	> 250 m	Fire	352
150 to 200 m	not-suitable	Moderate	Moderate	< 250 m	None	352
150 to 200 m	not-suitable	High	None	< 250 m	Fire	352
150 to 200 m	not-suitable	Moderate	Low	< 250 m	Fire	351
150 to 200 m	not-suitable	Low	Moderate	< 250 m	Fire	351
150 to 200 m	not-suitable	Low	High	> 250 m	None	303
150 to 200 m	not-suitable	not-suitable	High	> 250 m	Fire	303
150 to 200 m	not-suitable	not-suitable	High	< 250 m	None	303
150 to 200 m	not-suitable	High	Low	> 250 m	None	303
150 to 200 m	not-suitable	Moderate	Moderate	> 250 m	None	303
150 to 200 m	not-suitable	High	None	> 250 m	Fire	303

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
150 to 200 m	not-suitable	High	None	< 250 m	None	303
150 to 200 m	not-suitable	Moderate	Low	> 250 m	Fire	302
150 to 200 m	not-suitable	Moderate	Low	< 250 m	None	302
150 to 200 m	not-suitable	Low	Moderate	> 250 m	Fire	302
150 to 200 m	not-suitable	Low	Moderate	< 250 m	None	302
150 to 200 m	not-suitable	not-suitable	Moderate	< 250 m	Fire	302
150 to 200 m	not-suitable	Moderate	None	< 250 m	Fire	302
150 to 200 m	not-suitable	Low	Low	< 250 m	Fire	301
200 to 250 m	High	not-suitable	Moderate	> 250 m	None	403
200 to 250 m	High	Moderate	None	> 250 m	None	403
200 to 250 m	High	Low	Low	> 250 m	None	402
200 to 250 m	High	not-suitable	Low	> 250 m	Fire	402
200 to 250 m	High	not-suitable	Low	< 250 m	None	402
200 to 250 m	High	Low	None	> 250 m	Fire	402
200 to 250 m	High	Low	None	< 250 m	None	402
200 to 250 m	High	not-suitable	None	< 250 m	Fire	402
200 to 250 m	High	not-suitable	Low	> 250 m	None	353
200 to 250 m	High	Low	None	> 250 m	None	353
200 to 250 m	High	not-suitable	None	> 250 m	Fire	353
200 to 250 m	High	not-suitable	None	< 250 m	None	353
200 to 250 m	High	not-suitable	None	> 250 m	None	304
200 to 250 m	Low	Low	High	> 250 m	None	402
200 to 250 m	Low	not-suitable	High	> 250 m	Fire	402
200 to 250 m	Low	not-suitable	High	< 250 m	None	402
200 to 250 m	Low	High	Low	> 250 m	None	402
200 to 250 m	Low	Moderate	Moderate	> 250 m	None	402
200 to 250 m	Low	High	None	> 250 m	Fire	402
200 to 250 m	Low	High	None	< 250 m	None	402
200 to 250 m	Low	Moderate	Low	> 250 m	Fire	401
200 to 250 m	Low	Moderate	Low	< 250 m	None	401
200 to 250 m	Low	Low	Moderate	> 250 m	Fire	401
200 to 250 m	Low	Low	Moderate	< 250 m	None	401
200 to 250 m	Low	not-suitable	Moderate	< 250 m	Fire	401
200 to 250 m	Low	Moderate	None	< 250 m	Fire	401
200 to 250 m	Low	Low	Low	< 250 m	Fire	400
200 to 250 m	Low	not-suitable	High	> 250 m	None	353
200 to 250 m	Low	High	None	> 250 m	None	353
200 to 250 m	Low	Moderate	Low	> 250 m	None	352
200 to 250 m	Low	Low	Moderate	> 250 m	None	352
200 to 250 m	Low	not-suitable	Moderate	> 250 m	Fire	352
200 to 250 m	Low	not-suitable	Moderate	< 250 m	None	352
200 to 250 m	Low	Moderate	None	> 250 m	Fire	352
200 to 250 m	Low	Moderate	None	< 250 m	None	352
200 to 250 m	Low	Low	Low	> 250 m	Fire	351
200 to 250 m	Low	Low	Low	< 250 m	None	351
200 to 250 m	Low	not-suitable	Low	< 250 m	Fire	351
200 to 250 m	Low	Low	None	< 250 m	Fire	351
200 to 250 m	Low	not-suitable	Moderate	> 250 m	None	303

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
200 to 250 m	Low	Moderate	None	> 250 m	None	303
200 to 250 m	Low	Low	Low	> 250 m	None	302
200 to 250 m	Low	not-suitable	Low	> 250 m	Fire	302
200 to 250 m	Low	not-suitable	Low	< 250 m	None	302
200 to 250 m	Low	Low	None	> 250 m	Fire	302
200 to 250 m	Low	Low	None	< 250 m	None	302
200 to 250 m	Low	not-suitable	None	< 250 m	Fire	302
200 to 250 m	Moderate	not-suitable	High	> 250 m	None	403
200 to 250 m	Moderate	High	None	> 250 m	None	403
200 to 250 m	Moderate	Moderate	Low	> 250 m	None	402
200 to 250 m	Moderate	Low	Moderate	> 250 m	None	402
200 to 250 m	Moderate	not-suitable	Moderate	> 250 m	Fire	402
200 to 250 m	Moderate	not-suitable	Moderate	< 250 m	None	402
200 to 250 m	Moderate	Moderate	None	> 250 m	Fire	402
200 to 250 m	Moderate	Moderate	None	< 250 m	None	402
200 to 250 m	Moderate	Low	Low	> 250 m	Fire	401
200 to 250 m	Moderate	Low	Low	< 250 m	None	401
200 to 250 m	Moderate	not-suitable	Low	< 250 m	Fire	401
200 to 250 m	Moderate	Low	None	< 250 m	Fire	401
200 to 250 m	Moderate	not-suitable	Moderate	> 250 m	None	353
200 to 250 m	Moderate	Moderate	None	> 250 m	None	353
200 to 250 m	Moderate	Low	Low	> 250 m	None	352
200 to 250 m	Moderate	not-suitable	Low	> 250 m	Fire	352
200 to 250 m	Moderate	not-suitable	Low	< 250 m	None	352
200 to 250 m	Moderate	Low	None	> 250 m	Fire	352
200 to 250 m	Moderate	Low	None	< 250 m	None	352
200 to 250 m	Moderate	not-suitable	None	< 250 m	Fire	352
200 to 250 m	Moderate	not-suitable	Low	> 250 m	None	303
200 to 250 m	Moderate	Low	None	> 250 m	None	303
200 to 250 m	Moderate	not-suitable	None	> 250 m	Fire	303
200 to 250 m	Moderate	not-suitable	None	< 250 m	None	303
200 to 250 m	not-suitable	High	High	> 250 m	Fire	402
200 to 250 m	not-suitable	High	High	< 250 m	None	402
200 to 250 m	not-suitable	Moderate	High	< 250 m	Fire	401
200 to 250 m	not-suitable	High	Moderate	< 250 m	Fire	401
200 to 250 m	not-suitable	High	High	> 250 m	None	353
200 to 250 m	not-suitable	Moderate	High	> 250 m	Fire	352
200 to 250 m	not-suitable	Moderate	High	< 250 m	None	352
200 to 250 m	not-suitable	High	Moderate	> 250 m	Fire	352
200 to 250 m	not-suitable	High	Moderate	< 250 m	None	352
200 to 250 m	not-suitable	Low	High	< 250 m	Fire	351
200 to 250 m	not-suitable	High	Low	< 250 m	Fire	351
200 to 250 m	not-suitable	Moderate	Moderate	< 250 m	Fire	351
200 to 250 m	not-suitable	Moderate	High	> 250 m	None	303
200 to 250 m	not-suitable	High	Moderate	> 250 m	None	303
200 to 250 m	not-suitable	Low	High	> 250 m	Fire	302
200 to 250 m	not-suitable	Low	High	< 250 m	None	302
200 to 250 m	not-suitable	not-suitable	High	< 250 m	Fire	302

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
200 to 250 m	not-suitable	High	Low	> 250 m	Fire	302
200 to 250 m	not-suitable	High	Low	< 250 m	None	302
200 to 250 m	not-suitable	Moderate	Moderate	> 250 m	Fire	302
200 to 250 m	not-suitable	Moderate	Moderate	< 250 m	None	302
200 to 250 m	not-suitable	High	None	< 250 m	Fire	302
200 to 250 m	not-suitable	Moderate	Low	< 250 m	Fire	301
200 to 250 m	not-suitable	Low	Moderate	< 250 m	Fire	301
> 250 m	High	not-suitable	High	> 250 m	None	404
> 250 m	High	High	None	> 250 m	None	404
> 250 m	High	Moderate	Low	> 250 m	None	403
> 250 m	High	Low	Moderate	> 250 m	None	403
> 250 m	High	not-suitable	Moderate	> 250 m	Fire	403
> 250 m	High	not-suitable	Moderate	< 250 m	None	403
> 250 m	High	Moderate	None	> 250 m	Fire	403
> 250 m	High	Moderate	None	< 250 m	None	403
> 250 m	High	Low	Low	> 250 m	Fire	402
> 250 m	High	Low	Low	< 250 m	None	402
> 250 m	High	not-suitable	Low	< 250 m	Fire	402
> 250 m	High	Low	None	< 250 m	Fire	402
> 250 m	High	not-suitable	Moderate	> 250 m	None	354
> 250 m	High	Moderate	None	> 250 m	None	354
> 250 m	High	Low	Low	> 250 m	None	353
> 250 m	High	not-suitable	Low	> 250 m	Fire	353
> 250 m	High	not-suitable	Low	< 250 m	None	353
> 250 m	High	Low	None	> 250 m	Fire	353
> 250 m	High	Low	None	< 250 m	None	353
> 250 m	High	not-suitable	None	< 250 m	Fire	353
> 250 m	High	not-suitable	Low	> 250 m	None	304
> 250 m	High	Low	None	> 250 m	None	304
> 250 m	High	not-suitable	None	> 250 m	Fire	304
> 250 m	High	not-suitable	None	< 250 m	None	304
> 250 m	High	not-suitable	None	> 250 m	None	255
> 250 m	Low	Moderate	High	> 250 m	None	403
> 250 m	Low	High	Moderate	> 250 m	None	403
> 250 m	Low	Low	High	> 250 m	Fire	402
> 250 m	Low	Low	High	< 250 m	None	402
> 250 m	Low	not-suitable	High	< 250 m	Fire	402
> 250 m	Low	High	Low	> 250 m	Fire	402
> 250 m	Low	High	Low	< 250 m	None	402
> 250 m	Low	Moderate	Moderate	> 250 m	Fire	402
> 250 m	Low	Moderate	Moderate	< 250 m	None	402
> 250 m	Low	High	None	< 250 m	Fire	402
> 250 m	Low	Moderate	Low	< 250 m	Fire	401
> 250 m	Low	Low	Moderate	< 250 m	Fire	401
> 250 m	Low	Low	High	> 250 m	None	353
> 250 m	Low	not-suitable	High	> 250 m	Fire	353
> 250 m	Low	not-suitable	High	< 250 m	None	353
> 250 m	Low	High	Low	> 250 m	None	353

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
> 250 m	Low	Moderate	Moderate	> 250 m	None	353
> 250 m	Low	High	None	> 250 m	Fire	353
> 250 m	Low	High	None	< 250 m	None	353
> 250 m	Low	Moderate	Low	> 250 m	Fire	352
> 250 m	Low	Moderate	Low	< 250 m	None	352
> 250 m	Low	Low	Moderate	> 250 m	Fire	352
> 250 m	Low	Low	Moderate	< 250 m	None	352
> 250 m	Low	not-suitable	Moderate	< 250 m	Fire	352
> 250 m	Low	Moderate	None	< 250 m	Fire	352
> 250 m	Low	Low	Low	< 250 m	Fire	351
> 250 m	Low	not-suitable	High	> 250 m	None	304
> 250 m	Low	High	None	> 250 m	None	304
> 250 m	Low	Moderate	Low	> 250 m	None	303
> 250 m	Low	Low	Moderate	> 250 m	None	303
> 250 m	Low	not-suitable	Moderate	> 250 m	Fire	303
> 250 m	Low	not-suitable	Moderate	< 250 m	None	303
> 250 m	Low	Moderate	None	> 250 m	Fire	303
> 250 m	Low	Moderate	None	< 250 m	None	303
> 250 m	Low	Low	Low	> 250 m	Fire	302
> 250 m	Low	Low	Low	< 250 m	None	302
> 250 m	Low	not-suitable	Low	< 250 m	Fire	302
> 250 m	Low	Low	None	< 250 m	Fire	302
> 250 m	Moderate	Low	High	> 250 m	None	403
> 250 m	Moderate	not-suitable	High	> 250 m	Fire	403
> 250 m	Moderate	not-suitable	High	< 250 m	None	403
> 250 m	Moderate	High	Low	> 250 m	None	403
> 250 m	Moderate	Moderate	Moderate	> 250 m	None	403
> 250 m	Moderate	High	None	> 250 m	Fire	403
> 250 m	Moderate	High	None	< 250 m	None	403
> 250 m	Moderate	Moderate	Low	> 250 m	Fire	402
> 250 m	Moderate	Moderate	Low	< 250 m	None	402
> 250 m	Moderate	Low	Moderate	> 250 m	Fire	402
> 250 m	Moderate	Low	Moderate	< 250 m	None	402
> 250 m	Moderate	not-suitable	Moderate	< 250 m	Fire	402
> 250 m	Moderate	Moderate	None	< 250 m	Fire	402
> 250 m	Moderate	Low	Low	< 250 m	Fire	401
> 250 m	Moderate	not-suitable	High	> 250 m	None	354
> 250 m	Moderate	High	None	> 250 m	None	354
> 250 m	Moderate	Moderate	Low	> 250 m	None	353
> 250 m	Moderate	Low	Moderate	> 250 m	None	353
> 250 m	Moderate	not-suitable	Moderate	> 250 m	Fire	353
> 250 m	Moderate	not-suitable	Moderate	< 250 m	None	353
> 250 m	Moderate	Moderate	None	> 250 m	Fire	353
> 250 m	Moderate	Moderate	None	< 250 m	None	353
> 250 m	Moderate	Low	Low	> 250 m	Fire	352
> 250 m	Moderate	Low	Low	< 250 m	None	352
> 250 m	Moderate	not-suitable	Low	< 250 m	Fire	352
> 250 m	Moderate	Low	None	< 250 m	Fire	352

Appendix G. cont.

Distance to Known Plat Occurrences	Habitat Suitability	Hydrologic Features	Disturbance	Wind Direction	Fire History	Model Value
> 250 m	Moderate	not-suitable	Moderate	> 250 m	None	304
> 250 m	Moderate	Moderate	None	> 250 m	None	304
> 250 m	Moderate	Low	Low	> 250 m	None	303
> 250 m	Moderate	not-suitable	Low	> 250 m	Fire	303
> 250 m	Moderate	not-suitable	Low	< 250 m	None	303
> 250 m	Moderate	Low	None	> 250 m	Fire	303
> 250 m	Moderate	Low	None	< 250 m	None	303
> 250 m	Moderate	not-suitable	None	< 250 m	Fire	303
> 250 m	not-suitable	High	High	< 250 m	Fire	402
> 250 m	not-suitable	High	High	> 250 m	Fire	353
> 250 m	not-suitable	High	High	< 250 m	None	353
> 250 m	not-suitable	Moderate	High	< 250 m	Fire	352
> 250 m	not-suitable	High	Moderate	< 250 m	Fire	352
> 250 m	not-suitable	High	High	> 250 m	None	304
> 250 m	not-suitable	Moderate	High	> 250 m	Fire	303
> 250 m	not-suitable	Moderate	High	< 250 m	None	303
> 250 m	not-suitable	High	Moderate	> 250 m	Fire	303
> 250 m	not-suitable	High	Moderate	< 250 m	None	303
> 250 m	not-suitable	Low	High	< 250 m	Fire	302
> 250 m	not-suitable	High	Low	< 250 m	Fire	302
> 250 m	not-suitable	Moderate	Moderate	< 250 m	Fire	302

Appendix H. Efficiency, occupancy, and hits metric values derived from SAMPLE (Smith and Nichols 2006) used to evaluate each sampling design for the target invasive plant species in Big Bend National Park. Definitions of each metric are provided in Table 5-2. Sample size (n) represents the number of locations used in the simulations.

Design Type	Efficiency	Occupancy	Hits
Giant Reed (n=31)			
Single-stage SRS-NA	1.064	0.001769	0.001769
Single-stage SRS-A	0.988	0.001912	0.001897
Single-stage SRS-GIS-A	1.341	0.001758	0.001758
Single-stage GRID-NA	1.222	0.001712	0.001712
Two-stage SRS-NA	1.081	0.000039	0.000039
Two-stage SRS-A	1.233	0.000033	0.000033
Bermuda grass (n=117)			
Single-stage SRS-NA	0.750	0.000678	0.000678
Single-stage SRS-A	0.742	0.000683	0.000847
Single-stage GRID-NA	1.331	0.000584	0.000584
Two-stage SRS-NA	0.885	0.000125	0.000125
Two-stage SRS-A	1.031	0.00013	0.000133
Lehmann lovegrass (n=253)			
Single-stage SRS-NA	0.930	0.001897	0.001897
Single-stage SRS-A	1.049	0.001954	0.003213
Single-stage GRID-NA	1.088	0.001869	0.001869
Two-stage SRS-NA	0.975	0.00026	0.00026
Two-stage SRS-A	1.030	0.000259	0.000348
Horehound (n=26)			
Single-stage SRS-NA	0.740	0.002844	0.002844
Single-stage SRS-A	1.030	0.002381	0.003972
Single-stage SRS-GIS-A	0.869	0.002305	0.002305
Single-stage GRID-NA	0.908	0.002753	0.002753
Two-stage SRS-NA	1.165	0.000019	0.000019
Two-stage SRS-A	1.014	0.000031	0.000058
Buffelgrass (n=627)			
Single-stage SRS-NA	0.763	0.002603	0.002603
Single-stage SRS-A	1.117	0.002572	0.003904
Single-stage GRID-NA	0.577	0.002782	0.002782
Two-stage SRS-NA	0.532	0.000418	0.000418
Two-stage SRS-A	2.843	0.000347	0.00053
Russian thistle (n=38)			
Single-stage SRS-NA	1.010	0.000275	0.000275
Single-stage SRS-A	1.062	0.000261	0.000282
Single-stage GRID-NA	0.866	0.000269	0.000269
Two Stage SRS-NA	1.753	0.000028	0.000028
Two Stage SRS-A	1.746	0.000028	0.000030
Johnson grass (n=147)			
Single-stage SRS-NA	1.457	0.000716	0.000716
Single-stage SRS-A	1.062	0.000261	0.000282
Single-stage GRID-NA	1.033	0.000709	0.000709
Two-stage SRS-NA	1.162	0.00016	0.00016
Two-stage SRS-A	0.983	0.000153	0.000177

Appendix H. cont.

Design Type	Efficiency	Occupancy	Hits
Saltcedar (n=225)			
Single-stage SRS-NA	1.088	0.002323	0.002323
Single-stage SRS-A	1.135	0.002321	0.004238
Single-stage SRS-GIS-A	1.085	0.002482	0.002513
Single-stage GRID-NA	0.835	0.001963	0.001963
Two-stage SRS NA	0.741	0.000285	0.000285
Two-stage SRS A	1.022	0.000217	0.000389
All Species (n=1076)			
Single-stage SRS-NA	1.040	0.005737	0.005737
Single-stage SRS-A	2.380	0.005748	0.01216
Two-stage SRS-NA	2.695	0.000999	0.000999
Two-stage SRS-A	2.847	0.001041	0.001998