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Early Detection of Invasive Species on Holloman Air Force Base and the Tularosa Basin

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Early Detection of Invasive Plant Species on Holloman Air Force Base, New Mexico

Conducted by:
New Mexico Cooperative Fish and Wildlife Research Unit,
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Department of Animal and Range Sciences,
New Mexico State University

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Final Report

Early Detection of Invasive Plant Species on Holloman Air Force Base, New Mexico

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Executive Summary

Increases in trans-oceanic commerce, terrestrial transportation systems, and varying land use practices have ensured that many plants have been introduced into the places that are beyond their native range. Many of these introduced plants are considered "invasive" because they are likely to cause economic or environmental harm, or harm to human health. Invasive species have the ability to displace native plant and animal species, disrupt nutrient and fire cycles, and alter the character of the community by enhancing additional invasions. Once invasive plants become established and actively spread, management strategies become limited to expensive control efforts. As such, resource managers seek early detection methods to locate and eradicate new invasions before they threaten native biodiversity and ecological processes.

Predictive theories and models can provide a useful framework for early detection tools, especially when combined with ecological risk assessment procedures. When areas of high risk of invasive species incursion are identified, ground reconnaissance can be more effectively used for verification and control. Areas predicted as potential habitat that are not currently occupied, may be areas for frequent monitoring as undetected plant propagules may exist in the area.

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 on Holloman Air Force Base (HAFB). Holloman Air Force Base is located in the northern Chihuahuan Desert, near Alamogordo, New Mexico. Our proof-of-concept approach was to first create inductive models of potential invasive species habitat based on known plant occurrences on HAFB. We also created spatial models of distributional pathways on HAFB, and conducted a risk assessment that allows for prioritizing areas for conservation efforts. We selected eight invasive plant species for our analyses: Russian knapweed (*Acroptilon repens*), Malta starthistle (*Centaurea melitensis*), African rue (*Peganum harmala*), Russian olive (*Elaeagnus angustifolia*), saltcedar (*Tamarix* spp.), Siberian elm (*Ulmus pumila*), five-horn smotherweed (*Bassia hyssopifolia*) and Russian thistle (*Salsola kali*). These species were a concern because they colonize disturbed areas quickly, out-compete native species, and alter natural processes that further promote invasive species invasions and establishment.

From May through August 2007, we systematically surveyed HAFB for our target invasive plants, navigating to known plant locations to verify their occurrence, and surveying roads, parking lots, other disturbed sites, arroyos, springs, and other water sources throughout HAFB. This systematic sampling approach optimized our chance of detecting additional invasive plant populations on HAFB. We also surveyed 295 randomly selected locations. Ground cover estimates were collected at a random sample (n=159) of the invasive plant locations, and at the 295 random locations using a 25 m (82 ft) point intercept transect. Percent cover of native vegetation, invasive plant, and non-vegetation categories at sites occupied by our target invasive plants and at sites randomly selected across HAFB were summarized.

We used known locations of our target invasive plants in constructing predictive habitat models. We used spectral information and an available land cover map to model potential habitats for our target invasive plants. Spectral reflectance (bands 1-4) and vegetation indices (Normalized

Difference Vegetation Index and Tasseled Cap Transformation) were derived from 1.0 m (3.28 ft) resolution Quickbird imagery. The digital land cover map was developed in 1997, and consisted of 22 land cover classes.

We used a deductive (descriptive) approach to construct predictive habitat models when plant occurrence samples were $\leq$ 25. For plants with sample locations > 25, we used an inductive modeling approach through the program Maxent. Maxent uses the principle of maximum entropy to estimate the target probability distribution that has the broadest distribution compatible with the information available. We selected the most parsimonious model possible by using analyses provided in Maxent and correlation analyses to reduce the number of variables included in the models. We evaluated model performance by randomly withholding 12-20% of the occurrence points. Evaluation metrics included receiver operating characteristic (ROC) curves (threshold-independent metric) and threshold-dependent analyses. The area under the ROC curve (AUC) provided a summary measure of the model's discrimination ability. The threshold-dependent metric evaluated model performance by testing if a model performed significantly better than random by employing a binomial test (Wilcoxon signed-rank test) based on omission and predicted area.

Our approach to build early detection models was based on a risk analysis approach. We incorporated a system of numerical ranks and weighting of spatial factors that influence the distribution of our target invasive plant species. We used five spatial variables for our risk analyses: 1) invasive occurrence locations, 2) predicted species habitat suitability, 3) vectors and pathways, 4) disturbance, and 5) soil moisture. The numerical ranks and weights allowed us to combine multiple stressors, habitats, and species occurrences into a spatial model that represented areas at risk of invasion by nonnative species.

We located 879 invasive plant locations on HAFB. We also found our target invasive plants at 22 (8%) of the randomly sampled sites. We did not detect our invasive plant species in dune, wetland or playa, or creosotebush communities. Our target invasive plants were found in areas that yielded a low percent native vegetation cover (< 10%) and a high percent non-vegetative cover (> 60%), which characterizes areas that have received a moderate to high intensity disturbance. Landscape communities randomly sampled yielded over twice the amount of native vegetation cover than areas sampled with invasive plant occurrences. The high non-vegetative percent cover and low percent native vegetation cover at invasive plant areas sampled likely indicates that invasive plants are colonizing non-vegetative areas and displacing native vegetation on HAFB.

We used between 20-360 occurrences locations to generate individual species habitat models. Small sample sizes (< 15 occurrences) occurred with Russian knapweed, Russian olive, and Siberian elm. As such, the performance of the deductive models for these species could not be adequately determined. Visual interpretations of ROC and omission curves for inductive predictive habitat models indicate that all of the plant models performed better than a random prediction. Training AUC values ranged from 0.91 to 0.99, and test AUC values ranged from 0.93 to 0.98, indicating all models yielded a high discriminate ability to detect potential suitable habitat for each target species. All models contained one band of spectral data, a vegetation index, and the land cover dataset. However, each plant model was represented by a different set of variables and different percent contributions, which underline the need to evaluate multiple environmental variables when creating multiple species predictive models. Likewise, analyses provided in Maxent when combined with correlation analyses can provide effective parsimonious models.

Our results indicate that Malta starthistle and saltcedar may have the most limited potential distribution on HAFB, with ≤ 1,323 ha (≤ 3,200 ac; ≤ 6%) of the base surface modeled as potential habitat. Likewise, we estimated approximately 2,128 ha (5,257 ac; 10%) of suitable habitats for Russian knapweed, five-horn smotherweed, Russian olive, and Siberian elm on HAFB. Russian thistle and African rue may have the widest potential distribution across HAFB. We estimated approximately 3,120 ha (7,710 ac; 15%) of suitable area for Russian thistle and African rue on HAFB. Given the moderate errors of omission with the Russian thistle and African rue predictive models (37% and 27%, respectively), our estimates of the amount of suitable habitat for these species may be conservative. High potential suitable habitat for our target species occurred throughout the base along roadsides, parking lots, disturbed areas, and within fourwing saltbush and grassland communities.

Areas at risk of invasive plant incursion also occurred throughout HAFB. Areas of primary concern included the residential and commercial area in the southern part of the base, areas around installation and test track infrastructures, and areas around paved and gravel roadsides and parking lots. Although Russian knapweed has only been recorded in Hay Draw on HAFB, approximately 1,310 ha (3,237 ac) were considered to be at high risk for this plants incursion or spread, with an additional 2,345 ha (5,794 ac) at moderate risk. Modeled areas at high risk of Russian knapweed incursion were primarily in the southern part of the base around commercial and residential areas, in Hay Draw, and areas next to the test track. *Eradication of Russian knapweed on HAFB may be possible with immediate control efforts exerted on known populations followed by an active monitoring program in high risk areas.*

Approximately 4,739 ha (11,712 ac; 22%) of HAFB was considered at high and moderate risk to five-horn smotherweed incursion, which represents most of the area around residential and commercial areas, along roadsides, and wetlands and arroyos. Similarly, our analyses indicated that approximately 21% of HAFB was at high and moderate risk for Malta starthistle incursion, which primarily represented disturbed areas around residential and commercial development in the southern part of the base, and roadsides and disturbed areas in the middle to north part of the base. This plant appears to be rapidly spreading and may pose to be the next invasive plant issue on HAFB. *At this time, eradication of five-horn smotherweed and Malta starthisle on HAFB may be possible with immediate control efforts exerted on known populations followed by an active monitoring program in high risk areas.*

Approximately 22% (4,562 ha; 11,274 ac) of HAFB were modeled at high and moderate risk of African rue incursion. Large populations of African rue were found along all road right-of-ways, the High Speed Test Track, Space Command testing area, 46 Test Group office areas, and many portions of the flightline infield on HAFB. Further, African rue is actively spreading into the native grasslands and shrublands. Tilling, mowing, and blading along roadsides have likely increased its spread. Eradication of this species is unlikely. Thus, management action should focus on containment.

Russian thistle can be found in a wide variety of habitat types on HAFB, including disturbed areas, fields, roadsides, grasslands and desert communities. Our analyses indicated that approximately 23% (4,915 ha; 12,145 ac) of the base is at high and moderate risk of incursion by Russian thistle. Moderate at risk patches were modeled throughout native vegetation communities, although large patches of areas at risk were modeled around open or disturbed areas/lots, residential and commercial areas, roadsides, and air support facilities. Control efforts for this plant may be difficult given the habitat generalist nature and the specialized seed dispersal mechanism of this plant.

Our risk assessment also indicated that approximately 23% (4,946 ha; 12,223 ac) of HAFB were at moderate and high risk of saltcedar incursion. Saltcedar has moved aggressively into drainage bottoms and lowland depressions at the southern end of the base, and along drainage bottoms and isolated depressions across most of HAFB. While control efforts may have limited the immediate spread, propagules may still exist in suitable habitat areas on the base. Continued monitoring and management of controlled populations and suitable areas is warranted.

Russian olive and Siberian elm were primarily introduced for horticultural purposes, and have similar distributions and predicted risk distributions on HAFB. Our risk assessment indicated that approximately 18% (~3,800 ha; 9,390 ac) were at moderate and high risk of Russian olive or Siberian elm incursion. However, these species are relatively contained to the southern part of the base and they do not appear to be actively spreading. Nevertheless, to avoid future incursion into areas at risk, consideration should be given to eradication efforts of known plants.

Our risk analyses indicate that xeric riparian habitats may have approximately 5-times greater risk to invasive plant incursion than upland habitat types on HAFB. Approximately 48% of riparian habitats and 20% of wetland habitats are at moderate to high risk of invasive plant incursion or spread. These habitats have already been heavily impacted by saltcedar, and host favorable conditions for other species of concern. Approximately 9% of the grassland and fourwing saltbush communities on HAFB are at moderate to high risk of invasive plant incursion or spread.

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 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.

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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 species are sparsely populated. 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 across the Chihuahuan Desert.

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Roadmap Through Document and Deliverables

This report is organized in sections to facilitate dissemination of survey results. We first present a review of invasive species issues and early detection needs followed by brief target invasive plant species profiles. We outline the study objectives and provide detailed methods on our image processing, predictive model development and associated risk assessments. Research results and discussion are provided together, with the report concluding with a conservation implication section.

Our project generated volumes of spatial data, which could not be adequately included in appendices. As such, a variety of digital information accompanies this report, including:

1) Final report that includes background information, study area description, methods, results and discussion, conservation implications and literature cited.
    a. Figures include study area maps, and relevant spatial depiction of research results (species predictive habitat models, and associated risk assessments that lead to early detection tools).
    b. Tables provide summaries of research results.

2) ArcGIS shape files in NAD83, UTM Zone 13 projection:
    a. Locations of invasive plants located during surveys on HAFB.
    b. Locations of invasive plants located during surveys on the Boles Wells Water System Annex.

3) Erdas Imagine image files (.img) of :
    a. Species predictive habitat models.
    b. Early detection risk assessment models, classes of low, moderate, and high risk for each species, and all eight species combined.
    c. Habitat risk models of riparian, grassland, dune, wetland, creosote bush, and fourwing saltbush communities.

4) Quickbird Imagery
    a. Moasic, 16 bit, 1 m (3.28 ft) spatial resolution, multispectral
    b. Moasic, 16 bit, 60 cm (23.6 in) spatial resolution, multispectral

Detailed methods on the creation of these spatial coverages are provided in this document.

INTRODUCTION

Invasive Species Issues

Increases in trans-oceanic commerce, terrestrial transportation systems, and varying land use practices have allowed many plants to have either accidentally or deliberately been introduced into places that are well beyond their native range (Moody and Mack 1988). Many of these introduced plants are considered “invasive.” An invasive species is 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 1,000 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).

Invasive species are deterrents to natural resources management. Invasive plant management is expensive, labor-intensive, and can be a long-term endeavor. The economic impact caused by invasive species is difficult to estimate because their cumulative impacts are poorly understood. However, Pimentel et al. (2000) estimated 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).

Once plants become established and begin to actively spread, management strategies become limited to expensive control efforts. 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 (Smith et al. 1999, Rejmanek 2000). As such, resource managers need to locate and eradicate new invasions before they threaten native biodiversity and ecological processes (Stohlgren et al. 1999). Developing methods to detect invasive species soon after introduction could result in substantial economic savings and circumvent negative ecological impacts.

Early Detection Needs

The potential number of invasive species has led ecologists to explore predictive theories about attributes that make habitats susceptible to invasion (Rejmanek and Richardson 1996). Because traits that consistently predict invasiveness vary by geographic location, species, and habitat, global predictive theories have proven elusive (Alpert et al. 2000). Nevertheless, predictive theories and models can provide a useful framework for prioritizing conservation efforts to prevent new introductions or control existing infestations, especially at site specific areas. 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, 3) rapid reproduction and growth (above or below ground), 4) multiple reproductive strategies (vegetative and seed production), 5) high leaf area index, 6) small seed mass (facilitates seed dispersal), 7) long flowering period or continuous seed production for as long as growing conditions permit, 8) unique or special strategies to disperse propagules, and 9) the ability to regenerate from severed rootstocks (Baker 1965, Goodwin et al. 1999, 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).

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, 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

Remotely sensed and GIS data have been widely used to detect and model invasive plant distributions (Table 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 (Chudamani et al. 2004, Lass et al. 2005, Young and Schrader 2008). 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.

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 times than the surrounding vegetation. Multiple image dates may allow for detecting these

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

phenological differences. For example, remote sensing data that coincided specifically to flowering events aided in the detection of yellow starthistle (*Centaurea solstitalis*) (Lass et al. 1996), and leafy spurge (*Euphorbia esula*) (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.

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).

Early Detection Based on Risk Assessment

Invasive species spread and establishment is partially influenced by population growth, dispersal mode, landscape structure or suitability, and the number and frequency of invasive species 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). Principal 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 (Andersen et al. 2004a). Additional effects analysis involves estimating the probability and severity of economic, health, or environmental consequences of an exposure to invasive species.

Risk characterization integrates information from the problem statement, exposure and effects analyses into an overall conclusion about risk that is informative for conservation efforts (Andersen et al. 2004a). Risk characterizations may range from qualitative judgments to quantitative probabilities (EPA 1998). Although quantitative risk assessments are desirable, quantifying 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. 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.

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 data to model potential habitats and pathways that allow for landscape scale risk assessments. GIS datasets for invasive species pathways are readily available from a variety of GIS data clearinghouses, federal agencies, and private companies. Spatial datasets on potential vectors are not readily available, and should be created specifically for the target species and the area of interest.

Plant Species Characteristics

Life history knowledge of an invasive plant species is helpful in determining the general range and environmental characteristics required for the establishment and spread of the species. Life history information can also aid in determining appropriate spatial data 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 eight invasive plant species selected for this project (Table 2). 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.

All of the selected species were considered a priority species by land managers in our study area (described below). Russian knapweed (*Acroptilon repens*), Malta starthistle (*Centaurea melitensis*), and African rue (*Peganum harmala*) are classified as Class B noxious plants by the New Mexico Department of Agriculture (Office of the Secretary 2009). Class B includes noxious plants that have populations limited to portions of the state. High priority is given to preventing the spread of these species into new areas. Russian olive (*Elaeagnus angustifolia*), saltcedar (*Tamarix* spp.), and Siberian elm (*Ulmus pumila*) are classified as Class C noxious plants in New Mexico. These plants are considered widespread in the state. The remaining two species, five-horn smotherweed (*Bassia hyssopifolia*) and Russian thistle (*Salsola kali*) were not listed on New Mexico's noxious weed list (Office of the Secretary 2009). All of the selected species were a concern because they colonize disturbed areas quickly, out-compete native species, and alter natural processes that further promote invasive species invasions and establishment.

Table 2. Invasive species selected for remote sensing and GIS spatial analyses at Holloman Air Force Base, New Mexico.

Common Name	Scientific Name	Ecological Consequences	Noxious Plant Classification[1]
Russian knapweed	*Acroptilon repens*	Displace native vegetation by forming dense, single species stands through a combination of competition and allelopathy resulting in a loss of biodiversity. Also toxic to livestock.	B
Five-horn smotherweed	*Bassia hyssopifolia*	Ecological consequences of this plant presence are poorly understood. It can become a monoculture in established areas. It is toxic to sheep, and may be a threat to other livestock.	N/A
Malta starthistle	*Centaurea melitensis*	Rapidly displace native vegetation resulting in a loss of biodiversity. This plant is toxic to horses, and contributes to soil moisture depletion and increased soil erosion.	B
Russian olive	*Elaeagnus angustifolia*	Displace native riparian vegetation resulting in a loss of biodiversity and decreases stream flow and conveyance.	C
African rue	*Peganum harmala*	Ecological consequences of this plant presence are poorly understood. Displace native vegetation by forming dense, single species stands through a combination of competition and possible allelopathy resulting in a loss of biodiversity. This plant is toxic to cattle, sheep and horses.	B
Russian thistle	*Salsola kali*	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.	N/A
Saltcedar	*Tamarix* spp.	Aggressive competitor for water resources where it lowers stream flows and water tables, increases soil salinity, and displaces native species. This plant can increase the frequency, intensity, and effect of fires and floods.	C
Siberian elm	*Ulmus pumila*	Competitor for soil moisture and nutrient availability. Displaces native species which effectively reduces biodiversity.	C

1 The New Mexico Department of Agriculture classifies noxious plants into three classes: **Class A** noxious plants that are not in New Mexico or have limited distribution, but have the potential to cause serious problems; **Class B** noxious plants that are limited to portions of the state and high priority is given to preventing the spread of the species into new areas; and **Class C** noxious plants that are widespread in the state (Office of the Secretary 2009).

Russian Knapweed (*Acroptilon repens*)

General Description

Russian knapweed is a native to Mongolia, western Turkestan, Iran, Turkish Armenia and Asia Minor (Moore and Frankton 1974, Zimmerman 1996). It was initially introduced into the U.S. in the early 1900's as a contaminant in alfalfa seed (Watson 1980). Russian knapweed has since spread throughout the U.S. (27 states) and Canada (five states) (Natural Resources Conservation Service 2009), with severe infestations occurring in the western and central states (Maddox et al. 1985).

Russian knapweed is a deep-rooted, long lived perennial with erect stems that grow to heights between 0.3-1.0 m (1.0-3.0 ft) (Zouhar 2001). Basal leaves are typically gray-green in color, 0.8-1.6 cm (2-4 in) long and deeply lobed. Upper leaves are smaller with smooth margins, but can be slightly lobed. Stems and leaves are covered with dense gray hairs. Russian knapweed emerges in early spring (Watson 1980, Zimmerman 1996) and flowers from June to September and produces pink to lavender flowers that grow in solitary heads at the tips of the leafy branches. Seeds are whitish in color with a feather-like plume.

Russian knapweed is a highly aggressive perennial weed that can rapidly spread. It is considered to be the most persistent of the knapweeds (Lacey 1989). Russian knapweed can displace native vegetation by forming dense, single species stands through a combination of competition and allelopathy (Maddox et al. 1985, Stevens 1986, Whitson 1999). Kurz et al. (1995) noted a loss of forage, habitat, and overall rangeland biodiversity in Russian knapweed infested habitats. In addition, Russian knapweed is toxic to livestock (Panter 1991, Robles et al. 1997, Knight and Walter 2001).

Vectors and Pathways

Russian knapweed reproduces from both seeds and its root system. The primary mode of spread is via an extensive horizontal root system. Russian knapweed roots can extend up to 7.0 m (23.0 ft) deep and survive for more than 75 years, reproducing vegetatively through the spreading of rhizomes (Watson 1980, Stubbendieck et al. 1995). As a result of its extensive root system, research suggests this plant can survive severe fires (Bottoms and Whitson 1998). Although Russian knapweed produces approximately 1,200 seeds per plant per year, it lacks effective mechanisms for long distance seed dispersal (Watson 1980, Whitson 1999, Andersen 1993, Zouhar 2001). The amount of time seeds remain viable is unclear, but may be as long as nine years in lab environments (Selleck 1964). Knapweed seed is dispersed by animals, wind, and water. The biggest contributor to the movement of knapweed seeds are humans; through vehicles and road or other heavy equipment, movement on clothes, and as hay contaminants.

Habitat Associations

Russian knapweed occurs on most soil types, including saline and alkaline soils, but does especially well in areas with clay soils (Maddox et al. 1985, Roche et al. 1986). It can be found along roadsides, railways, riverbanks, and irrigation ditches; and in disturbed areas, clear cuts, pastures, orchards, vineyards, and other croplands (Roche et al. 1986, Dall'Armellina and Zimdahl 1988). Russian knapweed is shade intolerant, as flower production declines with decreasing light levels (Dall'Armellina and Zimdahl 1988). Russian Knapweed is common in semi-arid areas (Selleck 1964, Whitson 1999). Plant infestations have been noted to multiply in dry temperatures and decrease in moist locations due to competition with neighboring perennial grasses (Maddox et al. 1985).

Five-horn Smotherweed (*Bassia hyssopifolia*)

General Description

Five-horn smotherweed is native to Europe and Asia, particularly near the Caspian Sea (Collins and Blackwell 1979, Hoshovsky 2003). This plant was first sighted in the U.S. around 1915 near Fallon, Nevada (Collins and Blackwell 1979), and may have been introduced as a seed contaminant (Alex 1982). The current five-horn smotherweed distribution is widespread, and extends from the western U.S. and Canada states, through Montana, Wyoming, Colorado, and Texas (Hoshovsky 2003, Welsh et al. 2003, Natural Resources Conservation Service 2009). There are also reports of smotherweed as far east as Maine.

Five-horn smotherweed is a deep rooted, grayish annual in the Chenopodiaceae family that can reach heights of 1.0 m (3.0 ft) (Huang and Wu 1991, Hoshovsky and Kyser 2000). It has small, flat, linear to lanceolate leaves that are alternate. Five-horn smotherweed produces inconspicuous flowers on a woolly-looking spike between late summer-fall (July-October) (Hoshovsky and Kyser 2000). The fruit has five distinctive hooked structures on each seed that promotes seed dissemination. Young smotherweed stems have long, soft hairs (Hoshovsky and Kyser 2000). Five-horn smotherweed resembles lambs' quarters (*Chenopodium album*), but has smaller, more elongated leaves. It is also often confused with Russian thistle (*Salsola tragus*) and Kochia (*Kochia scoparia*), but is less branched than Russian thistle and hairier than Kochia (Hoshovsky and Kyser 2000).

Ecological consequences of this plant presence are poorly understood. Current evidence that this plant alters ecosystem processes (e.g., fire cycles, hydrological cycles, soil properties) is unavailable, although it can become a monoculture in established areas (Hoshovsky 2003). Five-horn smotherweed is considered toxic to sheep, and may be a threat to other livestock (James et al. 1976).

Vectors and Pathways

Five-horn smotherweed is a prolific seed producer that spreads by seed dispersal (Hoshovsky and Kyser 2000). Seeds host distinctive hooked structures that promote attaching to fur or feathers of animals (Collins and Blackwell 1979). Human activities also serve as common vectors. Seeds are spread by vehicles and road and canal maintenance equipment. Although seeds do not survive well in fresh water for long periods, several plant attributes facilitate germination and growth strategies, including short seed dormancy, warm germination requirements, tolerance of saline/alkaline conditions, and initial rapid growth, especially underground (Hoshovsky and Kyser 2000).

Habitat Associations

Five-horn smotherweed is associated with alkaline habitats, coastal dunes, salt marshes, disturbed areas, abandoned agricultural fields, roadsides, and fields (Hoshovsky and Kyser 2000, Hoshovsky 2003). Smotherweed has spread along the edges of wetland and riparian areas throughout temperate deserts in the U.S. (West and Young 2000). Van Devender et al. (1997) considered five-horn smotherweed as one of the most serious exotic plant invaders of riparian habitats in the Sonoran Desert region. Likewise, Hoshovsky (2003) considers this plant as an important exotic weed in California because of the difficulty in assessing the recovery potential of infected areas.

Malta Starthistle (*Centaurea melitensis*)

General Description

Malta starthistle is native to southwestern Europe in dry landscapes, disturbed areas, and in communities of other winter annuals (Roche and Roche 1991, Callaway et al. 2003). It was likely first introduced into the U.S. in California in the 1700s. The first record of its presence was a seed found in adobe bricks in a 1797 building in San Fernando, California (Hendry 1931). Malta starthistle has spread to most western and southwestern states, with scattered populations also occurring in the Midwest and Eastern U.S. (Natural Resources Conservation Service 2009).

Malta starthistle occurs on noxious weed lists in California, Nevada, and New Mexico (California Interagency Noxious Weed Coordinating Committee 2003, Nevada Administrative Code 2008, Office of the Secretary 2009). Although it is not listed on the state noxious weed lists in Oregon and Washington, Roche and Roche (1991) noted large infestations in these states.

Malta starthistle in the Asteraceae family, and greatly resembles yellow starthistle (Whitson et al. 2000), but is typically less prevalent than yellow starthistle. Malta starthistle is a winter annual < 1 m (< 3.3 ft) tall, with alternate, narrow, and unlobed stem leaves (DiTomaso and Gerlach 2000, Porras and Munoz 2000). Leaves are approximately 1-3 cm (0.4-1.2 in) long, with smooth, toothed, or wavy margins. Up to three flowers form at the tips of branches from April-June (Allred and Lee 1999, DiTomaso and Gerlach 2000). Flowers are yellow, small, (8-12 mm; 0.3-0.5 in diameter) with spiny bracts (Roche and Roche 1991). Malta starthistle exhibits several reproductive adaptations that provide competitive advantages. It has the ability to germinate in the fall and winter, and it can produce flowers in the rosette of basal leaves, before the flowering stalk matures. These early flowers can mature and seed before the other flowers appear (DiTomaso and Gerlach 2000, West 2001). Malta starthistle also has the ability to self-pollinate (Porras and Alvarez 1999) which may provide an advantage in habitats where resource availability is limited (Porras and Munoz 2000).

Malta starthistle can rapidly displace native vegetation and create a monoculture (DiTomaso and Gerlach 2000, Callaway et al. 2003). In addition, this plant contributes to soil moisture depletion, increased soil erosion, and wildfire risk (Roche and Roche 1991, Allred and Lee 1999). Malta starthistle is toxic to horses, causing a nervous disorder called “chewing disease” (Kingsbury 1964).

Vectors and Pathways

Malta starthistle is not known to spread vegetatively but can have compensatory growth when clipped (Callaway et al. 2001). However, it is a proficient seed producer. Inflorescences can produce up to 6,000 seeds (DiTomaso and Gerlach 2000). Although the rate of spread is poorly understood, this plant spreads by seed, and can be a contaminant in crop seed and hay (Roche and Roche 1991, DiTomaso and Gerlach 2000). The spine-like phyllaries and bristles facilitate seed dispersal by vehicles and road maintenance equipment along roadsides, off-road vehicles, non-motorized recreation, or by animals or wind (DiTomaso and Gerlach 2000). Roche (1992) noted that 40 km/hr (25 mph) winds moved yellow starthistle seeds up to 5 m (16 ft). Given that seed head morphology of Malta starthistle is similar to yellow starthistle, wind dispersal may not be a major contributor to long distance dispersal.

Habitat Associations

Malta starthistle can occupy rangelands, grasslands, pastures, agriculture and fallow fields, irrigation ditches, roadsides, and disturbed areas (Parker 1972, Felger 1990, Allred and Lee 1999). In California, it is widely distributed but is infrequent in annual grasslands and forms dense populations only on disturbed or xeric sites (DiTomaso and Gerlach 2000).

Russian Olive (*Elaeagnus angustifolia*)

General Description

Russian olive is a native to southern Europe and central and eastern Asia (Little 1961, Olson and Knopf 1986) where it occurs primarily in relatively moist habitats (Katz and Shafroth 2003). It was intentionally introduced into the U.S. as a horticultural plant, although the year of introduction is unclear (Katz and Shafroth 2003, Zouhar 2005). Russian olive was promoted for cultivation in several states for windbreaks, wildlife, shade trees, hedgerows, and other horticultural purposes as early as the 1900s and as late as the 1990s (Katz and Shafroth 2003). Since its introduction, Russian olive has spread throughout much of the U.S. and Canada, with the exception of the southeast U.S. (Zouhar 2005). In a survey of riparian woody vegetation at stream gauging stations in the 17 western states, Friedman et al. (2005) concluded that Russian olive was the fourth most frequent riparian plant (based on frequency of occurrence) in the western U.S.

Russian olive is in the Elaeagnaceae family and is a fast growing deciduous perennial tree with high stem densities and canopy cover (Katz and Shafroth 2003, Lamers et al. 2006). It has a dark (reddish), smooth, and sometimes shredding bark with branches that often possess thorns. Russian olive grows to heights of 5-12 m (16-40 ft) with large trunks 10-50 cm (4-20 in) diameter at breast height (Zouhar 2005). Leaves are alternate, lanceolate in shape, and have a silvery-gray color. Yellow flowers occur in the spring. Fruits are drupe- or berry-like, oval-shaped, and contain a single, relatively large seed (Young and Young 1992).

This plant has some horticultural and wildlife value; however, it has a host of ecological considerations (Shafroth et al. 1995, Zouhar 2005). Russian olive has several characteristics that give it a competitive advantage over native riparian vegetation, including shade tolerance, ability to form self-replacing stands, high seed production, viability and longevity, seed dispersal mechanisms, drought and salt tolerance, and the ability to establish in the absence of disturbance (Katz and Shafroth 2003, Zouhar 2005). As such, it can replace native riparian species, effectively reducing native plant and animal diversity, and decreasing stream flow and conveyance (Katz and Shafroth 2003, Friedman et al. 2005). Howe and Knoff (1991) considered Russian olive as a partial cause and a symptom of native species declines.

Vectors and Pathways

Although the current rate of spread is poorly understood (Pearce and Smith 2001), Russian olive is thought to have a relatively slow rate of spatial spread (Katz and Shafroth 2003). Russian olive exhibits vegetative reproduction following injury (Zouhar 2005) and has an extensive root system which allows for spread by underground rootstalks (Bovey 1965). However, the primary propagule dispersal mechanism is the spread of seed via wildlife (especially birds) or fluvial events (VanDersal 1939, Olson and Knopf 1986, Brock 1998, Kindschy 1998, Pearce and Smith 2001, Katz and Shafroth 2003). Streams, rivers, flood events, and birds can transport seed long distances (Brock 2003). Likewise, humans assist in the long-distance spread of Russian olive, as this tree is a popular horticultural tree.

Habitat Associations

Russian olive is primarily associated with riparian habitats, but it can occupy a variety of moisture conditions (Knopf and Olson 1984, Lesica and Miles 2001, Katz and Shafroth 2003). Campbell and Dick-Peddie (1964) noted Russian olive in xeric, mesic, and hydric sites along the Rio Grande in New Mexico. It has also been found in moist pastures, rangelands, and meadows (Carman and Brotherson 1982, Currier 1982). Russian olive can endure a wide-range of temperatures, and soil alkalinity and salinity (Katz and Shafroth 2003), although the frequency of occurrence of Russian olive decreases with increasing minimum temperatures (Friedman et al. 2005). Unlike other invasive plants that favor disturbed sites, Russian olive is relatively tolerant of the competitive effects of established vegetation (Lesica and Miles 1999). It can invade beneath the canopies of native woody vegetation or within herbaceous vegetation (Katz and Shafroth 2003).

African Rue (*Peganum harmala*)

General Description

African rue is native to semi-arid environments in northern Africa, the Middle East, and central Asia, and has been introduced to the U.S. and Australia (Whitson et al. 2000, Mahmoudian et al. 2002). In the U.S., African rue was first introduced near Deming, New Mexico, in 1928, (Whitson et al. 2000) for the production of red dye (Parker and Reiser 1997). It has since spread throughout the west, with populations occurring in California, Oregon, Washington, Idaho, Nevada, Arizona, New Mexico, Colorado, and Texas (Parker and Reiser 1997, Natural Resources Conservation Service 2009). Skinner et al. (2000) noted that African rue was one of the 45 frequently listed noxious weeds in the continental U.S. and southern Canada.

African rue (Zygophyllaceae family) is a bright green perennial desert shrub with a bushy growth habit that reaches heights around 0.3 m (1 ft) (Parker and Reiser 1997, Roche 1991). The plant has an extensive root system; with branching roots extending 4-6 m (13-20 ft) below the surface in sandy soils (Parker and Reiser 1997, Abbott et al. 2008). In southern New Mexico, African rue can exhibit a bimodal growth stage, initiating new growth in mid to late March through late June, followed by plant senescence as soils dry in early summer, and then reinitiating growth in late July and early August in response to the monsoonal rainfall (McDaniel and Duncan 2006, Abbott et al. 2007). African rue leaves are alternate, smooth, and divided deeply into narrow segments (Whitson et al. 2000). Flower production begins in mid April with fruit maturation occurring in June and July in New Mexico (Abbott et al. 2007). Flowers are white with five white petals and produce a cylindrical 2-4 celled leathery capsules that contain 45-60 dark brown seeds (Parker and Reiser 1997, Roche 1991, Whitson et al. 2000).

Little is known about the ecological consequences of African rue colonization. In New Mexico, African rue forms dense monoculture stands along roadsides and disturbed areas (K. Young, personal obs), effectively reducing native biodiversity. African rue also has several drought tolerant competitive advantages over native plants in New Mexico. African rue produces a deep root system; exhibits a phenology pattern of rapid growth and reproduction; achieves a stress-induced senescence and resumes growth and reproduction following favorable environmental conditions; and seedlings can tolerate, recover, and establish during periods of drought when native plants are suppressed (Abbott et al. 2006; 2007; 2008). In addition, African rue germinates earlier in spring, and may contain allelopathic chemicals that hinder or prevent other vegetation growth (Roche 1991, Parker and Reiser 1997). These competitive advantages have allowed African rue to expand into fourwing saltbush, creosote bush, and grassland communities (Roche 1991, McDaniel and Duncan 2006, Abbott et al. 2007). African rue is also toxic to cattle, sheep, and horses, as it contains lethal alkaloids (especially the seeds and roots) (Mahmoudian et al. 2002, McDaniel and Duncan 2006). However, Parker and Reiser (1997) noted that poisoning is rare because the plant is extremely unpalatable.

Vectors and Pathways

African Rue spreads primarily by seed proximate to the parent plant, dispersing greater distances via hay, seed mixture, equipment, livestock, vehicles, water, or by adhering to feet, fur, or feathers of animals (Parker and Reiser 1997). African rue can also spread vegetatively by lateral roots producing new shoots (Michelmore 1997, McDaniel and Duncan 2006). Tilling, mowing, and blading along roadsides have done little to reduce populations and have probably increased the spread (Roche 1991, Parker and Reiser 1997). Seeds may also be transported and introduced in contaminated gravel and fill materials (Abbott et al. 2008).

Habitat Associations

African rue tolerates a wide range of precipitation and temperature conditions (Abbott et al. 2007). However, it is adapted to arid environments where it can grow in a variety of soils and alkaline conditions (Parker and Reiser 1997). African rue colonizes frequently disturbed and barren areas, including roadsides, parking areas, corrals, stockyards, degraded desert rangelands, abandoned crop fields, and oil pads (Parker and Reiser 1997, McDaniel and Duncan 2006, Abbott et al. 2007).

Russian Thistle (*Salsola kali*)

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). It is currently widespread across the U.S. and Canada (Natural Resources Conservation Service 2009).

Russian thistle is in the Chenopodiaceae family and is more commonly known as 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 Russian thistle may grow as large 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 a 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, grasslands and desert communities (Young and Evans 1972, Young 1991).

Saltcedar (*Tamarix* sp.)

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 (Natural Resources Conservation Service 2009). 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 (Allred 2002, DiTomaso 1998). Common species that occur in the desert southwest are *T. chinensis* and *T. ramosissima*, but other species are known to occur (DiTomaso 1998).

Saltcedar (Tamaricaceae family) 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 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).

Siberian Elm (*Ulmus pumila*)

General Description

Siberian elm is native to northern China, eastern Siberia, Manchuria, and Korea (Dirr 1998). It was introduced into the United States around 1860 for windbreak planting because of its high resistance to Dutch elm disease (Rosendahl 1955, Kapaun and Cheng 1999). Siberian elm has rapidly spread throughout the U.S. and is known to occur in 43 states (Natural Resources Conservation Service 2009).

Siberian elm is in the Ulmaceae family, and is a fast-growing tree that can grow between 15-21 m (50-70 ft) tall (Czarapata 2005). The bark is a light gray-brown and is often streaked with stains caused by bacterial wetwood (Vines 1960). Leaves are oblong in shape, alternate, and are 2.5-7.6 cm (1-3 in) in length with serrate margins (Moore and Davis 2006). Siberian elm produces greenish flowers in March and April (Vines 1960). It produces samara fruit in April and May. The translucent wing surrounding the nutlet assists in seed dispersal.

Although Siberian elm has been used as a shade tree in much of the Midwest, it has limited ornamental value because the brittle wood is subject to breakage (Dirr 1998). In addition, it is susceptible to insect, disease, and herbicide damage, which makes it an undesirable tree (Moore and Davis 2006). Siberian elm can become weedy or invasive in some habitats and the extensive lateral spreading root system that can be detrimental to neighboring plants because they reduce soil moisture and nutrient availability which displaces native vegetation and reduces biodiversity (Parker and Williams 2003, Czarapata 2005, Moore and Davis 2006).

Vectors and Pathways

Siberian elm primarily spreads through seed dispersal; the winged fruit is carried by wind. Seedlings are fast growing, which allows Siberian elm to quickly outcompete native vegetation (Czarapata 2005). Siberian elms can also sprout vegetatively through its extensive lateral root system (Parker and Williams 2003). Humans also assist in the long-distance spread of Siberian elm, as this tree is still used in the horticultural trade.

Habitat Associations

Siberian elm can tolerate a variety of ecological and soil conditions, including poor, dry soils, cold winters, and long periods of summer drought (Moore and Davis 2006). Siberian elm can be found in riparian areas, roadsides, pastures, grasslands and other mesic sites (Moore and Davis 2006). This tree has naturalized and has become the dominant tree in some riparian areas in southern New Mexico (Parker and Williams 2003).

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 on Holloman Air Force Base (HAFB). Early detection protocols are paramount in developing conservation strategies that restrict or eradicate invasive plants on HAFB. Our proof-of-concept approach was to first create inductive models of potential invasive species habitat based on known plant occurrences on HAFB. We also created spatial models of distributional pathways on HAFB, and conducted a risk assessment that allows for prioritizing areas for conservation efforts. Research objectives include:

1) Collect spatially explicit information on target invasive plant species on HAFB;

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; and

3) Conduct an assessment of selected invasive species predicted distributions and vectors and pathways to provide a spatial model for the early detection of invasive plants on HAFB.

STUDY AREA

Holloman Air Force Base is located in the northern Chihuahuan Desert, approximately 13 km (6 mi) west of Alamogordo, New Mexico (Fig. 1). The main part of the base encompasses approximately 20,543 ha (50,763 ac) in the Tularosa Basin (Reiser et al. 2000). Elevation ranges from 1,224 m (4,015 ft) to 1,320 m (4,330 ft). The average elevation on HAFB is 1,240 m (4,086 feet) (Reiser et al. 2000).

The Air Force base has been recognized for its unique ecological features which include shifting dune fields, alkaline flats, and gypsum lake (Reiser et al. 2000). Soils on HAFB have high gypsum and salt content, and were formed primarily from alluvial and eolian processes (Reiser et al. 2000). Primary soil associations and complexes include: Corvus, Yesum, Gypsic, Bomber, Typic, Nasa, and Alamogordo (Derr et al. 1981, Natural Resource Conservation Service 2008). These soils are typically not productive and are susceptible to wind and water erosion (Reiser et al. 2000).

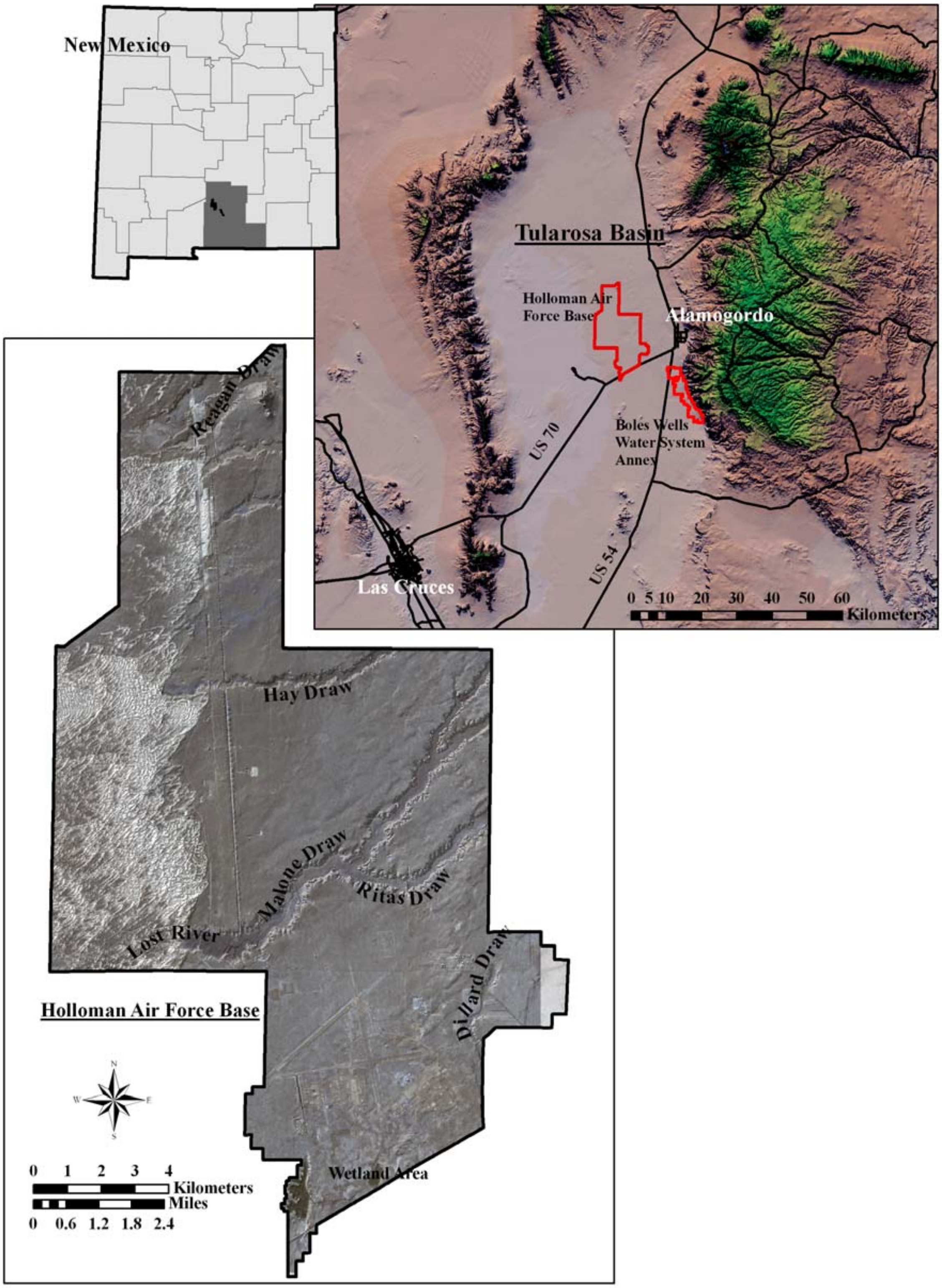

Figure 1. Map of Holloman Air Force Base and the Boles Wells Water System Annex, southern New Mexico.

Vegetation on HAFB consists primarily of fourwing saltbush (*Atriplex canescens*) and alkali sacaton (*Sporobolus airoides*), and subdominant species such as grama (*Bouteloua spp.*), tobosa (*Pleuraphis mutica*), threeawn (*Aristida* spp.), and soaptree yucca (*Yucca elata*) (Allred 1996). Muldavin et al. (1997) mapped 22 land cover classes (organized into 10 grouped classes) across approximately 21,281 ha (52,588 ac) in and immediately adjacent to HAFB (Table 3). Approximately 37% of the mapped area was characterized by fourwing saltbush communities, 23% grassland, 10 % riparian/wetlands, and 12% disturbed areas.

Table 3. Amount of each land cover class mapped by Muldavin et al. (1997) on Holloman Air Force Base.

Grouped Class	Land Cover Class	Hectares	Acres	Percent
Creosote Bush Shrubland				
	Creosote Bush Shrubland	147.3	364.0	0.69
Development / Ground Disturbance				
	Golf Course	17.6	43.5	0.08
	Development / Ground Disturbance	2,221.1	5,488.3	10.44
	Airfield	230.8	570.4	1.08
Dune				
	Rosemarymint Dune Shrubland	2,475.4	6,116.8	11.63
	Barren Duneland	1,000.7	2,472.7	4.70
Fourwing Saltbush Complex				
	Fourwing Saltbush Shrubland w/ Honey Mesquite	936.5	2,314.1	4.40
	Fourwing Saltbush / Gyp Dropseed Shrubland	2,395.3	5,919.0	11.26
	Fourwing Saltbush / Alkali Sacaton Shrubland	3,147.3	7,777.1	14.79
	Sparse Fourwing Saltbush Shrubland	1,493.8	3,691.3	7.02
Grassland				
	Gyp Grama Interdune Grassland	913.5	2,257.4	4.29
	Gyp Dropseed Grassland	2,575.6	6,364.4	12.10
	Alkali Sacaton Grassland	1,313.1	3,244.6	6.17
Rock Outcrop				
	Rock Outcrop	17.7	43.7	0.08
Saltcedar Woodland				
	Saltcedar Woodland	130.9	323.4	0.61
Urban Vegetation				
	Urban Vegetation	44.2	109.3	0.21
Wetland/Playa				
	Wetland	51.1	126.2	0.24
	Surface Water	89.8	221.8	0.42
	Barren Alkaline Playa	160.7	397.2	0.76
Xeric Riparian				
	Pickleweed Shrubland	849.3	2,098.6	3.99
	Semi-riparian Alkali Sacaton Grassland	1,001.5	2,474.8	4.71
	Semi-riparian Honey Mesquite Shrubland	68.3	168.9	0.32

Precipitation on HAFB averages 21 cm (8.6 in) per year (Reiser et al. 2000). Most of the precipitation occurs during July to September from monsoonal rain events. Temperatures on HAFB are the coolest from December through March, averaging from 5-8°C (41-46°F) (Reiser et al. 2000). Summertime high temperatures occur in July, averaging approximately 27°C (81°F), but temperatures can reach 38°C (100°F). Wind velocities and direction on HAFB are variable. May typically has wind velocities > 17 mph, 90% of the time (Reiser et al. 2000). However, between April to July, median wind speeds are 25 mph. Prevailing winds are from the west from February to June, from the south to southeast from July through September, and from the north from October through January (Reiser et al. 2000).

Holloman Air Force Base and the Bureau of Land Management share responsibilities for the Boles Wells Water System Annex (BWWSA), which is located southeast of the main base, and 8 km (5 mi) south of Alamogordo, New Mexico (Fig. 1). The BWWSA provides potable water for the base. The well field annex is adjacent to the western foothills of the Sacramento Mountains, and covers approximately 2,802 ha (6,923 ac) (Reiser et al. 2000). Elevations for the BWWSA range from 1,246 m (4,087 ft) to 1,424 m (4,671 ft). Soils on the BWWSA were formed from alluvial and eolian materials deposited and are deep and gravelly throughout (Reiser et al. 2000). Vegetation communities across the BWWSA are primarily honey mesquite shrubland, creosote bush shrubland, honey mesquite / feather fingergrass shrubland, development or ground disturbance, and urban vegetation communities (Reiser et al. 2000).

A number of invasive plant species of concern exist in the Tularosa Basin, including African rue, kochia, Johnsongrass (*Sorghum halepense*), Lehmann lovegrass (*Eragrostis lehmanniana*), Malta starthistle, Russian knapweed, Russian thistle, and saltcedar. On HAFB, over 1,133 ha (2,800 ac), including approximately 283 ha (700 ac) of disturbed roadsides, have established populations of invasive weeds (Rieser et al. 2000). Saltcedar, African rue, and Russian thistle have been the most problematic invasive plant species on HAFB.

METHODS

Target Invasive Species Presence

Ideally, training data for species models would be obtained from a sampling procedure that involves a random selection and measurement of samples from a defined target population. In terms of sampling landscapes, sampling frames are often a defined spatial extent, which is divided into N smaller units of some set size (e.g., 30 m grid cells or watershed hydrologic units), from which a subset n_i is randomly selected and surveyed for the species of interest (Edwards et al. 2004). These types of sampling designs are considered probabilistic, and allow for inferential statistics. However, when attempting to survey and build species models of rare individuals, randomization procedures may not generate sufficient observations for a predictive habitat model (Edwards et al. 2005). In the case of rare species, or limited budgets, ecologists often actively search for the species of interest using non-probability sampling procedures, or purposive sampling.

In the case of this study, HAFB provided spatial data (UTM coordinates) on the occurrences of the target invasive plants on the base. This dataset was compiled from systematic roadside surveys conducted in 2005 and 2006 on the base. Unfortunately, the spatial accuracy of plant locations in this dataset was unknown. Thus we used a laptop computer with ArcGIS (9.1) interfaced with a Global Positioning System (GPS) to navigate to each plant location to verify plant occurrence and spatial location. We used digital aerial photographs (15 cm; 6 in spatial resolution) provided by the HAFB to plot plant occurrences and derive locations (UTM Zone 13, GRS 1980 Spheroid and NAD 83 Datum) in ArcGIS (9.1). We also collected additional locations of target species populations by systematically surveying roads, parking lots, other disturbed sites, arroyos, springs, and other water sources throughout HAFB. Field work occurred from May through August 2007. This systematic sampling approach optimized our chance of detecting additional invasive plant populations on HAFB.

Randomly Selected Sites

Target invasive plant species presence/absence information was also collected at randomly selected sites across HAFB. A stratified random sample (449 locations) was allocated to nine grouped land cover classes based on those mapped by Muldavin et al. (1997). We did not allocate samples to the Urban Vegetation class, as field measurements at these sites would be artificial, and land access would be difficult. Random samples were generated using ArcGIS 9.1 using the Hawth's tool random point generator. Due to time constraints, we sampled 295 of the 449 (66%) random samples (Table 4). A minimum of five samples for each land cover type was generated and sampled. We navigated to each random location with the aid of a GPS unit.

Plant Canopy Cover and Unvegetated Estimates

We estimated plant canopy cover and surface soil objects (unvegetated) to assist in predictive model interpretations. Since the goal of this study was to evaluate the efficacy of using remotely sensed and GIS data as tools to support early detection of invasive plants, we chose a ground sampling strategy that best approximated features detected by remotely sensed data.

Table 4. Number of randomly selected sites allocated and sampled per grouped land cover class on Holloman Air Force Base, New Mexico.

Group Land Cover Class	Number of Random Samples Allocated	Number of Sampled Locations	Percent Sampled
Creosote Bush Shrubland	5	5	100.0
Development / Ground Disturbance	44	38	86.4
Dune	74	46	62.2
Fourwing Saltbush Complex	170	80	47.1
Grassland	102	74	72.5
Rock Outcrop	5	5	100.0
Saltcedar Woodland	5	5	100.0
Wetland/Playa	6	6	100.0
Xeric Riparian	38	36	94.7
Total Number of Samples	449	295	65.7

Remote sensors measure the electromagnetic energy reflected from an object or area on the earth's surface. The spectral signature within a pixel of remotely sensed data consists of an average of the reflectance of all materials within that pixel. For example, spectral values for a vegetated area will consist of a combination of the spectra of all vegetation types and the amount and type of soil reflected. In these mixed pixels, ground-truthing using a spectrometer is needed to adequately estimate the composition (cover, distribution, and species variations) of the vegetation patch. This detailed evaluation of spectral signatures was beyond the scope of this research. Since detecting objects low in multistoried environments with remotely sensed data is challenging, and overstory objects tend to skew the resulting reflectance values, we chose a ground sampling strategy that estimated only the top canopy cover for vegetation species, e.g., an aerial view of plant canopy.

Plant canopy cover refers to the vertical projection of vegetation onto the ground when viewed from above. Plant canopy can be measured along different vertical strata (e.g., first foliar hit or multiple point interceptions) and different horizontal areas (e.g., plots, lines, points) (Bonham 1989, Elzinga et al. 1998). Vegetation cover is often segregated between basal cover (area where the plant intersects the ground) and foliar cover (measure of the leaf area that covers the ground surface) (Elzinga et al. 1998). We used a point-intercept transect method that recorded the first layer of plant canopy (first foliar intercept) at each point to estimate total percent plant cover. Transects were 25 m (82 ft) in length, and point-intercepts were recorded along 20 cm (8 in) intervals.

We randomly sampled 159 (~18%) of the invasive plant locations and all of the 295 randomly selected sites across HAFB. At each random location, one point-intercept transect was established and measured to estimate vegetated cover and the amount of unvegetated areas. We recorded the presence of grass, shrubs, forbs, shrubs, litter, exposed soil, gravel (2-60 mm; 0.08-2.4 in), and rock (> 60 mm; 2.4 in) at each point along transects. Plant species were identified to the lowest possible taxonomic level. Plant canopy hits included only the top (aerial) foliar or basal current year's production (Muldavin et al. 2001). Because remotely sensed imagery has difficulty differentiating basal and foliar hits, we did not distinguish between the two cover types with our ground surveys.

Litter included dead plant parts on the surface of the soil and attached dead material from previous growing seasons. Attached or detached litter appears equivalent to a satellite sensor (Muldavin et al. 2001). Exposed soil hits (bare ground) were recorded at points that were without a plant canopy structure, rock, gravel, or litter. As such, live plant canopy cover, exposed soil, rock, gravel, and litter were mutually exclusive and summed to 100%. Percent cover of vegetated and unvegetated features was estimated based on the number of hits on the target feature out of the total number of points measured.

Point-intercept transects were constrained to the designated land cover class. In linear habitats (e.g., roads or right of ways), a random direction (parallel to the gradient) was selected so that the entire transect was restricted to the designated land cover class to be sampled. When the random location fell in the center of the road, one side of the road was randomly chosen. In patch habitats (non-linear habitats), transects were placed in a random direction.

In this report, we summarize percent native vegetated cover, invasive plant cover, and percent unvegetated (exposed soil, rock, gravel, and litter) at sites occupied by our target invasive plants and at sites randomly selected across HAFB. Means accompanied by standard errors (se).

Invasive Species Predictive Habitat Models

We used all known locations of our target invasive plants in constructing predictive models. This dataset consisted of the corrected locations of invasive plants provided by HAFB and additional plant locations derived from our systematic ground survey efforts.

Spatial Datasets

We used spectral information and the available land cover map to model potential habitats for our target invasive plants (Table 5). Spectral reflectance and vegetation indices were derived from Quickbird imagery. The land cover map was created by Muldavin et al. (1997) which hosted 22 land cover classes.

Satellite Imagery

We obtained ortho-ready Quickbird imagery from Digital Globe (http://www.digitalglobe.com). Quickbird imagery has a spatial resolution of 2.4 m (8 ft) for the multispectral bands and 60 cm (23.6 in) for the panchromatic band. Images were obtained in UTM, Zone 13, WGS 84 Spheroid and Datum projection. We choose image dates that were close to our field data collection timeframe (spring/summer 07) and would encompass some of the invasive plant greeenup period. Three different dates were necessary to obtain a complete coverage of HAFB: November 10, 2006, April 11, 2007, and May 04, 2007, although there was significant spatial overlap between dates (Fig. 2).

Several image processing steps were needed to place the Quickbird images in the necessary format for sequential plant modeling (Fig. 3). All image processing was conducted in Erdas Imagine (9.0), unless otherwise noted. Orthorectification and image mosaics were conducted using the Lieca Photogrammetry Suite in Erdas Imagine (9.0). The raw Quickbird images were orthorectified using ground control points derived from 15 cm (6 in) digital aerial photos.

Table 5. Spatial datasets used to construct predictive models for invasive plants in Holloman Air Force Base.

Spatial Data Type	Number of Variables	Description	Data Type
Quickbird Imagery	4	Remotely sensed data used to detect vegetation phenology differences between invasive plant populations and surrounding landscape.	Continuous, raw spectral reflectance values for the four bands.
Vegetation Indices	4	NDVI, and tasseled cap transformation indices developed from Quickbird Imagery to help differentiate invasive species communities.	Continuous, NDVI values and Tasseled cap transformation for soil brightness, vegetation greenness, and soil/vegetation wetness
Vegetation or Habitat Type Association	1	Land cover map (Muldavin et al. 1997). See Table 3.	Categorical, 22 land cover classes

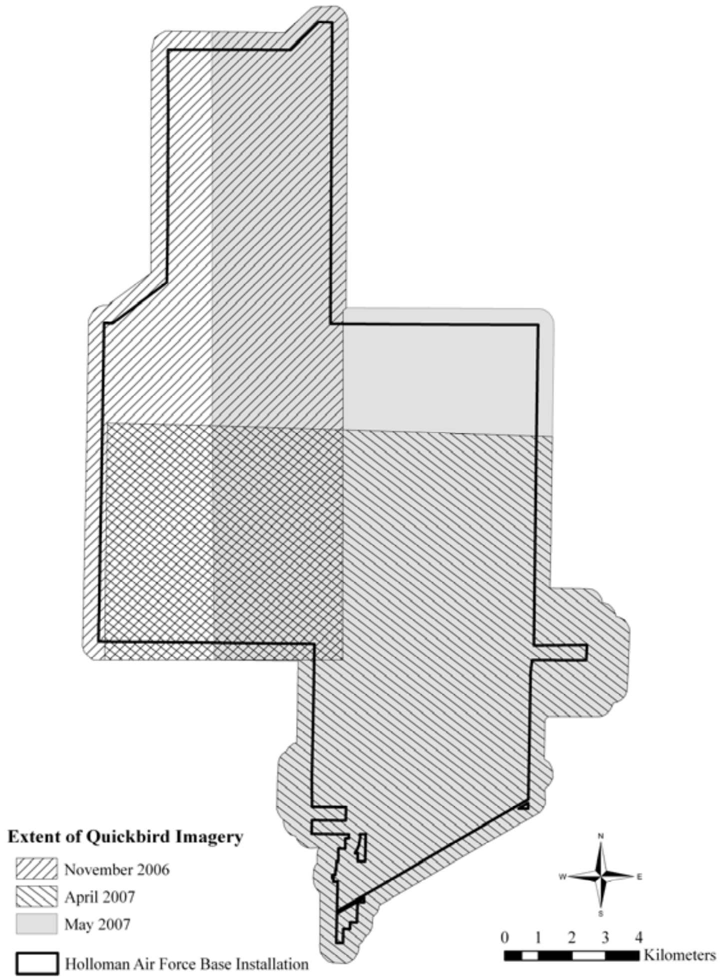

Figure 2. Spatial overlap of the three dates of Quickbird imagery used to obtain a complete coverage of Holloman Air Force Base.

We spatially matched the Quickbird images and the digital aerial photos using 149-170 locations (Table 6). Elevation data for the orthorectification process were derived from a 10 m (32.8 ft) Digital Elevation Model (DEM) obtained from the U.S. Geological Survey (http://gisdata.usgs.net). A cubic convolution resampling algorithm was used for the orthorectification process. Root mean square error for the orthorectified images were $\leq$ 0.58 m (1.89 ft) for the multispectral images and $\leq$ 0.34 m (1.12 ft) for the panchromatic images (Table 7) indicating an excellent spatial accuracy resulting from the orthorectification process.

Orthorectified images were combined using a mosaic tool to produced one multispectral and one panchromatic image. We created custom cutlines to choose image dates for particular areas across Holloman (Fig. 4). Cutlines were drawn along the dune interface to reduce spectral variance. We minimized the amount of area used by the November image, as this dataset may have hosted environmental attributes (e.g., soil moisture or vegetation greenup) that

Table 6. Number of points used in orthorectification of Quickbird images for Holloman Air Force Base.

Number of Points Used in Orthorectification	Multispectral (2.4 m)				Panchromatic (60 cm)			
	April	May	November	Total	April	May	November	Total
Ground Control Points	25	40	38	103	30	45	41	116
Check Points	10	20	16	46	12	22	20	54
Total	35	60	54	149	42	67	61	170

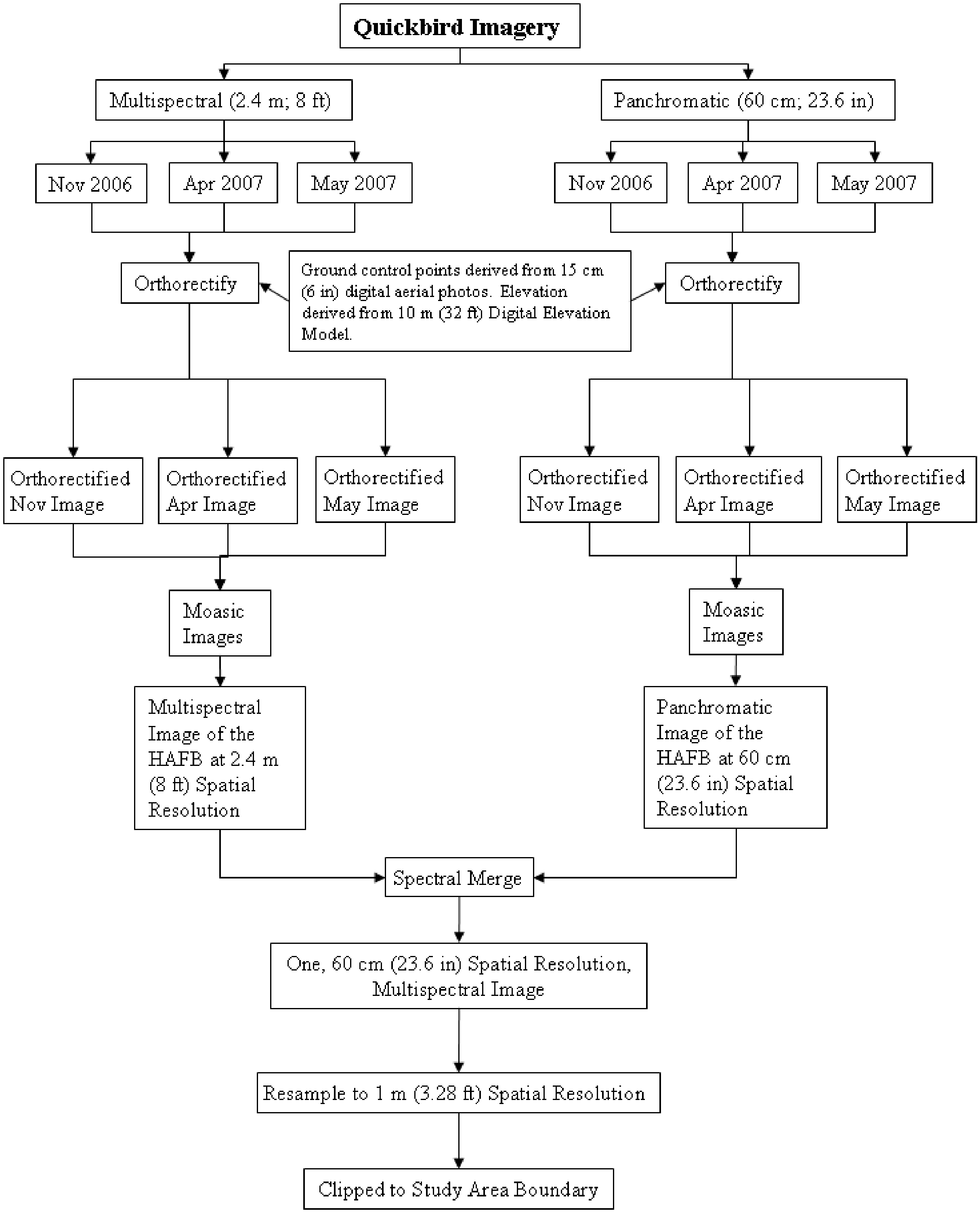

Figure 3. Image processing steps used to place Quickbird images in the necessary format for invasive plant modeling.

Table 7. Root Mean Square Error (RMSE) of Quickbird images resulting from the orthorectification process for Holloman Air Force Base.

Root Mean Square Error for Images	Multispectral (2.4 m)			Panchromatic (60 cm)		
	April	May	Nov	April	May	Nov
RMSE (pixels)	0.23	0.24	0.24	0.57	0.57	0.53
RMSE (meters)	0.54	0.58	0.58	0.34	0.34	0.32
RMSE (feet)	1.79	1.89	1.89	1.12	1.12	1.05

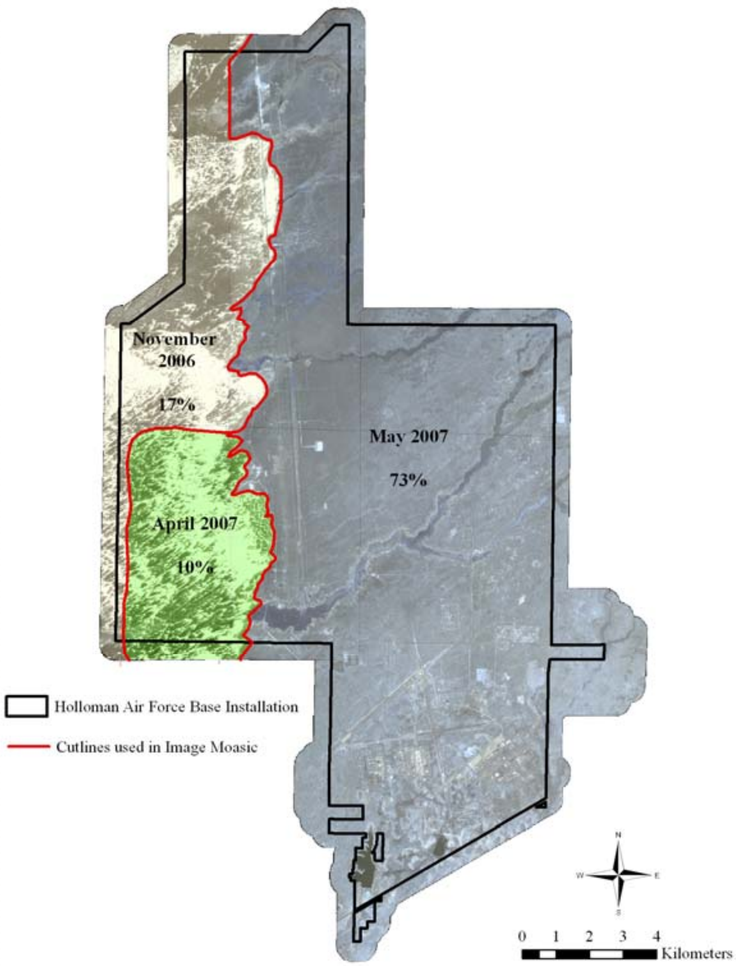

Figure 4. Custom cutlines used to choose image dates for a complete image mosaic across Holloman Air Force Base.

were different than the spring images. As such, the May 2007 image represented 73% of the mosaic surface, November 2006 image represented 17% and the April image represented 10% of the mosaic surface (Fig. 4). The same cutline was used for the multispectral and panchromatic mosaic. Cutlines were feathered 200 m (656 ft). A cubic convolution resampling algorithm was used for the mosaic process. The two mosaic images (multispectral and panchromatic) were then merged to produce a 60 cm image (from the panchromatic dataset) with 4 bands of spectral data (from the multispectral dataset). The Resolution Merge tool in Erdas Imagine 9.0 was used for this process. Data were combined using a principal components method and a cubic convolution resampling method. The output image was produced as an unsigned 16 bit *.img* file. The spectral merge image was then resampled to 1.0 m (3.28 ft) resolution and projected to UTM Zone 13, GRS 1980 Spheroid and NAD 83 Datum in ArcGIS 9.1 to facilitate additional analyses.

Vegetation Indices

Spectral enhancements to satellite images can be used to improve interpretability, reduce information redundancy, and extract information from the data which is not readily visible in its raw form. Normalized Difference Vegetation Index (NDVI) and Tasseled Cap Transformation are spectral enhancements that were conducted on the final spectral merged image. The NDVI is a ratio between measured reflectivity in the red and near infrared portions of the electromagnetic spectrum (e.g., band 4 – band 3 / band 4 + band 3). The contrast between vegetation and soil is maximized in these bands, with bare soils resulting in low NDVI values, and dense vegetation resulting in high NDVI values (Navular 2007). Tasseled Cap Transformation is similar to a principle component analysis, in that it uses mathematical equations to transform the original *n* multispectral bands into three bands related to soil brightness, vegetation greenness, and soil/vegetation wetness (Kauth and Thomas 1976). Tasseled Cap coefficients used for image transformation were obtained from Navular (2007).

Land Cover

Muldavin et al. (1997) modeled the land cover in HAFB by analyzing 2 m (6.6 ft) aerial photographs and 30 m (98 ft) Landsat 5 Thematic Mapper (TM) imagery corresponding to Fall 1993. The resulting digital dataset consisted of 22 land cover classes at a 2 m (6.6 ft) spatial resolution. Muldavin et al. (1997) tested the land cover model against 120 independent field samples, and concluded the spatial model was approximately 90% accurate in representing landscape conditions. All 22 land cover classes were used in the predictive habitat analyses.

Extent and Spatial Resolution of Spatial Datasets

All spatial datasets were resampled to 1.0 m (3.28 ft) spatial resolution to facilitate analyses. Further, all datasets needed to have the same spatial extent (area of data). As such, we clipped the satellite imagery datasets to the extent provided by the land cover model. All nine datasets (four spectral bands from the Quickbird image, three Tasseled Cap Transformation bands, NDVI dataset, and the land cover dataset) were extracted into Generic ASCII raster format for data analyses by Maxent software.

Analytical Approach to Construct Predictive Habitat Models

Deductive Habitat Models

There are a variety of analytical methods that could be used to construct predictive models for invasive species, ranging from simple overlays to statistically driven models. Boolean approaches employ overlay rules (where individual layers are added, subtracted, or multiplied together). These approaches have been widely used in wildlife habitats relationships models (e.g., GAP models) which describe resources and conditions present in areas where a species persists and reproduces or otherwise occurs. These modeled relationships predict, and spatially depict, areas of potentially suitable habitat. While these types of models are informative, they lack inferential abilities, and should be tested with independent data.

Sample sizes may restrict the type of analytical approach used. For example, inductive models are driven by statistical relationships of environmental variables associated with known occurrences. As such, sample sizes greatly influence the statistical relationships. Phillips et al. (2004) recommended 50-100 samples for optimal inductive models generated by the program

Maxent. Descriptive or deductive models are driven by associations with environmental variables, and thus are not influenced by low sample sizes. As such, we used a deductive approach to produce predictive habitat models for plant species with ≤ 25 plant occurrence records. Typical variables that may be used in deductive models for invasive plants include variables associated with the plants presence, establishment, and spread. We restricted our deductive habitat models to only the land cover model produced by Muldavin et al. (1997). Other variables associated with the plants presence, establishment, and spread were used in the risk assessments.

We reviewed plant habitat associations in the scientific literature, and provided each land cover class a numeric rank from 0 to 1.0, based on our perception of the importance of the land cover class to the ecology of the plant (Table 8). Our 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.

Deductive Model Performance

We evaluated the performance of the deductive models by using the locations of known plant occurrences. Due to limited sample sizes, we summarized only the percent agreement of known plant locations in predicted suitable habitats versus non-suitable habitats.

Table 8. Numeric rank provided to land cover classes modeled by Muldavin et al. (1997) on Holloman Air Force Base used to create deductive habitat models.

Land Cover Description	Russian Knapweed	Russian Olive	Siberian Elm
Creosote bush Shrubland	0.00	0.00	0.00
Golf Course	0.67	1.00	1.00
Development / Ground Disturbance	1.00	0.67	0.67
Airfield	0.33	0.33	0.33
Rosemarymint Dune Shrubland	0.00	0.00	0.00
Barren Duneland	0.00	0.00	0.00
Fourwing Saltbush Shrubland with Honey Mesquite	0.00	0.00	0.00
Fourwing Saltbush / Gyp Dropseed Shrubland	0.00	0.00	0.00
Fourwing Saltbush / Alkali Sacaton Shrubland	0.00	0.00	0.00
Sparse Fourwing Saltbush Shrubland	0.00	0.00	0.00
Gyp Grama Interdune Grassland	0.00	0.00	0.00
Gyp Dropseed Grassland	0.00	0.00	0.00
Alkali Sacaton Grassland	0.00	0.00	0.00
Rock Outcrop	0.00	0.00	0.00
Saltcedar Woodland	0.00	0.67	0.67
Urban Vegetation	0.67	1.00	1.00
Wetland	0.67	1.00	1.00
Surface Water	0.00	0.00	0.00
Barren Alkaline Playa	0.00	0.00	0.00
Pickleweed Shrubland	0.00	0.00	0.00
Semi-riparian Alkali Sacaton Grassland	0.00	0.00	0.00
Semi-riparian Honey Mesquite Shrubland	0.67	1.00	1.00

Inductive Habitat Models

Maxent software version 3.2.1 (http://www.cs.princeton.edu/~schapire/maxent/) was used to create inductive predictive habitat models. 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, Dudík et al. 2007). A description of how the program functions can be found in the software tutorial, help section, and from Phillips et al. (2004; 2006; 2009) and Phillips and Dudik (2008). Program outputs include a logistic probability surface with values ranging from 0 to 1, and tabular and graphical representations of model performance and variable contribution.

Species occurrences were used for the sample locations, and the Quickbird datasets provided the environmental variables measured at each sample point. We used the default settings for the regularization multiplier, iterations, convergence threshold, and the number of background points. 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.

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 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, and provides a tabular output of variable percent contribution.

We first created a model that included all nine variables (global or full model). This model was considered to have the most flexibility in fitting the data but may have low precision (White 2001). We then excluded all variables that contributed < 3% to the full model, and recreated the spatial model. We then tested the resulting model for variable correlation. We used the point intersect tool in ArcGIS to obtain model variable values under each plant occurrence point. These values were tested for correlation using Pearson's correlation coefficient calculated from Proc Corr in Statistical Analysis Software (SAS). Correlated variables ($r \geq 0.8$) were removed from the final model by retaining the variable with the highest model contribution. This approach provided 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 parameters (over-fitting) (Burnham and Anderson 1992).

Inductive 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 9). 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 9). 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 habitats (Phillips et al. 2006).

Table 9. Confusion matrix to derive measurements of model performance.

		Actual	
		Present	**Absent**
Predicted	**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- Omission Rate
Specificity = d/(b+d) = 1- Commission Rate

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 (sensitivity) rather than species absence (specificity). As such, a threshold weighted towards sensitivity would be selected.

For the inductive models, we randomly withheld 12-20% of the occurrence points for each target species to assess model performance. 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.

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).

Early Detection Models Based on Risk Assessments

Our approach to build early detection models was based on a risk analysis approach similar to Landis (2004) in that we incorporated a system of numerical ranks and weighting of spatial factors that influence the distribution of our target invasive plant species. We used five spatial variables for our risk analyses: 1) invasive occurrence locations, 2) predicted species habitat suitability, 3) vectors and pathways, 4) disturbance, and 5) soil moisture.

Numeric Ranks of Risk Variables

Our risk variables included a variety of classes (attributes, distances, or landscape conditions) that may interact to influence the suitability of the study area in terms of invasive plant establishment and persistence. As such, we provided numeric ranks that ranged from 0 (least likely to enhance invasive plant establishment or persistence) to 1.0 (most likely to enhance invasive plant establishment or persistence) for each attribute, distance, or landscape condition. Numeric ranks were assigned variable class by ordering classes in order of their likelihood to enhance invasive plant establishment or persistence. The number of classes per variable was divided by 1.0 to provide a numeric interval or distance between classes. For example, if a variable contained 13 classes, then the distance between numeric ranks for each class was 0.08, with a minimum value of 0.08 and a maximum value of 1.0. Although numeric ranks are not derived by quantified data, they were selected using our best professional judgment. Numeric should be evaluated with empirical data and updated as information becomes available. These numeric ranks may not be appropriate for other study areas or species.

Invasive Plant Occurrences

The probability of invasion success increases with greater propagule pressure or number of introduction attempts (Rejmanek 2000). Inherently, existing plant populations provide a steady propagule source for neighboring areas. Animals or people may inadvertently transport propagules from these populations in any direction. Thus, we used locations of known invasive species populations to represent species introduction points. We created five circular 50 m (164 ft) intervals around all target invasive plant species coordinates through a simple buffering operation in ArcGIS 9.2. The actual distance that propagules may travel is unknown, but assumed to be variable by species, locations, habitats, and seasons. Therefore, these distance intervals should be considered conservative estimates.

We provided each buffered plant location interval and all areas outside of the buffered plant locations a numeric rank between 0.17 and 1.0 (Table 10). Intervals closer to the plant coordinate were provided higher ranks due to increased likelihood of lateral expansion of the established population through seed dispersal or vegetative reproduction. A numeric rank of 0.17 was provided to all areas outside of the buffered plant locations, recognizing that all areas have some likelihood of plant occurrence.

Table 10. Numeric ranks given to buffer intervals around known invasive plant locations on Holloman Air Force Base.

Buffer Distance			
m	**(ft)**	**Description**	**Numeric Rank for all Target Plants**
50	(164)	Distance < 50 m of a known plant location.	1.00
100	(328)	Distance between 50 -100 m of a known plant location.	0.83
150	(492)	Distance between 100 - 150 m of a known plant location.	0.67
200	(656)	Distance between 150 - 200 m of a known plant location.	0.50
250	(820)	Distance between 200 - 250 m of a known plant location.	0.33
0	(0)	Distance > 250 m from a known plant location.	0.17

Predictive Species Habitat Models

Areas that host favorable environmental conditions for target plants are more likely to have successful invasions (Rejmanek 2000). As such, we used the inductive predictive habitat models derived through Maxent analyses and the deductive habitat models in our risk assessment. The inductive habitat surfaces represented logistic probabilities with values ranging from 0 to 1.0. The logistic values were not recoded or grouped for risk analyses. Likewise, the deductive habitat surfaces contained values ranging from 0 to 1.0, although these values did not represent logistic probabilities.

Vectors and Pathways

Potential distributional vectors and pathways used by invasive plants to expand their range in HAFB were identified through literature review and by consulting with plant experts. Propagule dispersal by wind, water, and human transportation were the primary vectors identified. Likewise, waterways, roads, runways and taxiways, utility and gas corridors, fence lines, and other human development areas tended to be the common pathways for our target invasive plant species. We were unable to create spatial datasets of the vectors that accurately portrayed potential distributional movements for these plants.

In many 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). Likewise, utility and gas corridors and fence lines were often associated with roads that were used for maintenance. As such, we used spatial datasets of roads (paved, gravel, and dirt), air support facilities (runways and taxiways), arroyos, playas, and canals to represent common pathways for HAFB.

We buffered paved roads and air support facilities 50 m (164.0 ft), in 5 m (16.4 ft) intervals. Likewise, dirt and gravel roads were buffered 20 m (65.6 ft), in 5 m (16.4 ft) intervals. The actual distance that propagules may travel along pathways was unknown, but assumed to be variable by species, locations, habitat, and pathway type. Therefore these distance intervals should be considered conservative estimates. The larger buffer distances for paved roads and air support facilities represented greater anthropogenic use, and thus a wider potential swath for propagule dispersal.

Arroyos, playas, and canals were not buffered, as we assumed propagule dispersal may occur via water dispersal (down slope) or by animals using arroyos as movement corridors (movement up or down arroyo). However, animals that cross waterways may also act as dispersal vectors. Waterway datasets were compiled using National Hydrography Dataset (NHD) (http://edc.usgs.gov), spatial datasets provided by HAFB, and by digitizing Digital Ortho Quads. Roads and air support facilities datasets were provided by HAFB.

We provided each potential pathway and their distance interval with numeric rank between 0.14 and 1.0 (Table 11). Because each target plant may react differently to potential pathways and their buffers, each plant received a different ranking. For example, African rue's primary pathway is along paved roads. Thus, areas < 5 m (16.4 ft) of a paved road received a numeric rank of 1.0. Conversely, five-horn smotherweed potential pathways are tied closely to soil moisture. Thus, arroyos, playas, and canals received a numeric rank of 1.0 for this species. Typically, distance intervals closer to the potential pathway were provided higher ranks due to increased likelihood of propagule expansion. A numeric rank of 0.07 was provided to all areas outside of the buffered pathways, recognizing that all areas have some likelihood of serving as a dispersal pathway or hosting dispersal vectors.

Table 11. Numeric ranks given to buffer intervals around potential pathways on Holloman Air Force Base.

Potential Pathway	Distance from Pathway m	(ft)	Numeric Ranks Given to Potential Pathways: African Rue, Russian Thistle, Malta Starthistle, Russian Olive, Siberian Elm	Five-horn Smotherweed, Saltcedar, Russian Knapweed
Paved Roads and Air Support Facilities				
	< 5	(<16.4)	1.00	0.93
	5 - 10	(16.4 - 32.8)	0.79	0.71
	10 - 15	(32.8 - 49.2)	0.71	0.64
	15 - 20	(49.2 - 65.6)	0.50	0.43
	20 - 30	(65.6 - 98.4)	0.36	0.29
	30 - 40	(98.4 - 131.2)	0.29	0.21
	40 - 50	(131.2 - 164.0)	0.21	0.14
Dirt and Gravel Roads				
	0	(0)	0.93	0.86
	0 - 5	(0 - 16.4)	0.86	0.79
	5 - 10	(16.4 - 32.8)	0.64	0.57
	10 - 15	(32.8 - 49.2)	0.57	0.50
	15 - 20	(49.2 - 65.6)	0.43	0.36
Arroyos, Playas, and Canals				
	0	(0)	0.14	1.00
Areas Not Included in Above Categories				
	0	(0)	0.07	0.07

Disturbances

Disturbance along roads by vehicle traffic and maintenance activity (e.g., road grading, ditch clearing, and trimming of overhanging vegetation), is often the source of repeated introductions (Gelbard and Belnap 2003). The frequency and intensity of disturbances can influence the abundance, species, and likelihood of invasive plant establishment and persistence. Increasing disturbance intensity may remove native individuals or species, and alter available resources necessary for native plant recolonization. Further, invasive plants are favored when disturbance frequency is greater than the rate of competitive exclusion.

Spatial datasets of roads (paved, gravel, and dirt), parking areas (paved, gravel, and dirt), air support facilities (runways, taxiways, and parking areas), and buildings were obtained from HAFB. Disturbance frequency and intensity was estimated for each of disturbance type by: 1) visually assessing disturbed areas, 2) reviewing 2004 and 2007 aerial photos, and 3) consultation with HAFB staff. Four different categories of disturbance frequency and intensity were estimated: 1) regular interval disturbance of moderate or high intensity, 2) regular interval disturbance of low intensity, 3) infrequent disturbance events of moderate or high intensity, and 4) infrequent disturbance events of low intensity. Examples of regular interval, moderate or high intensity disturbances included roads or parking areas that received daily traffic, received yearly maintenance in the form of grading (road surface) or grading and mowing roadsides. Typically, areas around buildings that receive yearly maintenance were considered as having regular disturbances that were of moderate or high intensity. Likewise, examples of regular interval, low intensity disturbance included roads that received at least daily traffic but did not receive yearly maintenance, or receive roadside maintenance. Infrequent, moderate or high disturbances included roads or areas that received less than monthly traffic, but were well maintained (roadbed and/or roadside). Infrequent, low intensity disturbances included closed roads, or roads that receive less than monthly traffic and are not maintained (roadbed and/or roadside).

In addition, we digitized areas based or large patch disturbances using 2007 Quickbird imagery. Disturbance frequency and intensity were estimated into two classes: 1) regular interval disturbances, and 2) infrequent interval disturbances. Regular interval disturbances tended to have total vegetation removal, or “moonscapes.” Thus, these areas were considered as having moderate or high disturbance intensity. Almost all of HAFB has been historically disturbed. There was evidence of tire tracks and other disturbance events across the entire base. Thus, areas outside of the “regular frequency disturbances” were classified as having infrequent disturbances. These areas still retain vegetation, although their disturbance intensity varied.

We expanded (buffered) disturbance types between 0-15 m (0-49.2 ft) to represent a potential area of influence. 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). The actual area of influence distance is unknown, but assumed to be greater for frequent, moderate or high intensity disturbances than infrequent, low intensity disturbances. As such, we created a 15 m (49.2 ft) buffer around paved roads because these areas receive greater traffic and have more frequent maintenance intervals than two-track dirt roads, which received a 5 m (16.4 ft) buffer (Table 12).

Generally, disturbances that occur on a regular basis and have moderate to high intensity are more likely to favor invasive plant establishment than disturbances with infrequent and low intensities. As such, we provided each disturbance type with numeric rank between 0.08 and 1.0 (Table 12). Because each target plant may react differently to potential disturbances, each plant received a different ranking. For example, regular and high intensity disturbance along roads favor the establishment of African rue. Thus, areas < 15 m (49.2 ft) of a paved road received the highest numeric rank of 1.0. Five-horn smotherweed establishment is heightened by area disturbances that have regular intervals and moderate or high intensity. Thus, large areas with regular disturbances received higher ranks than paved roads. We provided a numeric rank of 0.08 to areas across HAFB that had infrequent disturbances of varying intensity, recognizing that even very infrequent disturbed areas have some likelihood of plant establishment.

Table 12. Numeric ranks given to disturbance types on Holloman Air Force Base.

	Disturbance		Buffer Distance		Numeric Ranks Given to Disturbance Types				
Disturbance	**Frequency**	**Intensity**	**m**	**(ft)**	**African Rue**	**Five-horn Smotherweed, Russian Thistle**	**Russian Olive, Siberian Elm**	**Malta Starthistle**	**Saltcedar**
Paved Roads									
	Regular	Moderate or High	15	(49)	1.00	0.77	0.92	1.00	1.00
Paved Parking									
	Regular	Moderate or High	10	(33)	0.62	0.54	0.85	0.85	0.85
Air Runways and Taxi									
	Regular	Moderate or High	15	(49)	0.85	0.69	0.46	0.62	0.92
Air Parking									
	Regular	Moderate or High	10	(33)	0.46	0.46	0.77	0.46	0.77
Dirt or Gravel Roads									
	Regular	Moderate or High	15	(49)	0.92	0.92	0.69	0.92	0.62
	Regular	Low	5	(16)	0.31	0.31	0.62	0.31	0.31
	Infrequent	Moderate or High	10	(33)	0.38	0.38	0.23	0.38	0.38
	Infrequent	Low	5	(16)	0.15	0.15	0.15	0.15	0.15
Dirt or Gravel Parking									
	Regular	Moderate or High	10	(33)	0.54	0.85	0.54	0.77	0.54
	Infrequent	Moderate or High	10	(33)	0.23	0.23	0.31	0.23	0.23
Buildings									
	Regular	Moderate or High	10	(33)	0.69	0.62	1.00	0.69	0.69
Area Wide (Path type) Disturbances									
	Regular	Moderate or High	0	(0)	0.77	1.00	0.38	0.54	0.46
	Infrequent	Varied	0	(0)	0.08	0.08	0.08	0.08	0.08

Soil Moisture

Soil moisture properties may enhance the likelihood of plant establishment, especially in desert environments. We created a soil moisture dataset by merging two datasets: 1) soils data provided by Natural Resources Conservation Service, and 2) locations of arroyos, playas, and canals provided by the National Hydrography Dataset (http://edc.usgs.gov). We created four soil moisture classes: 1) very low to low, 2) moderate, 3) high, and 4) very high (Table 13). Soil attributes (texture, water capacity, and drainage) described by Derr et al. (1981) and Natural Resources Conservation Service (2008) were linked to the spatial soil data layers. Soil moisture classes were given a numeric rank between 0.25 and 1.0 (Table 13).

Soils with higher water holding capacity were considered to have properties that would enhance the likelihood of plant establishment, and thus were given higher ranks. A numeric rank of 0.25 was provided to areas with low or very low soil water holding capacity, recognizing that these areas also host some likelihood of plant occurrence and establishment.

Table 13. Numeric ranks given to soil moisture classe for all target invasive plant species on Holloman Air Force Base.

Soil Moisture Class	Numeric Rank	Soil Name or Land Form Description	Soil Drainage	Water Runoff
Very High	1.0			
		Water	Undefined	Undefined
		Arroyos	Undefined	Undefined
		Gypsic Aquisalids, Playas	Poorly drained	Medium
		Canal	Undefined	Undefined
High	0.75			
		Yesum, Calcareous-Gypsic Haplosalids complex	Well drained	Low
		Reclaimed Area	Undefined	Undefined
		Gypsic Haplosalids 35 to 90% slope	Well drained	Very high
		Bomber-Tornado Complex	Well drained	High
		Borrow Pits and Dumps	Undefined	Undefined
Moderate	0.50			
		Yesum-Nasa Complex	Excessively drained	Very low
		Typic Calcigypsids-Lithic Calcigypsids-Rock Outcrop Complex	Well drained	Medium
		Yesum Sandy Loam	Excessively drained	Very low
		Yesum Clay Loam	Well drained	Medium
		Matador Complex	Moderately well drained	Undefined
		Corvus-bucklebar	Undefined	Very low
Very Low or Low	0.25			
		Parabolic Dunes-Firebee Association 0 to 35% slope	Well drained	Undefined
		Firebee-Parabolic Dunes Association 0 to 35% slope	Well drained	Undefined
		Transverse Dunes 5 to 90% slope	Well drained	Undefined
		Transformer-Barchan Duneland Association 0 to 60% slope	Well drained	Undefined
		Nasa-Corvus Complex	Well drained	Very low
		Alamogordo-Nasa-Corvus Complex	Well drained	Very low
		Corvus Loam	Well drained	Medium

Numeric Weights of Risk Variables

Our numeric weights of risk variables represent a relative importance of each variable to the establishment and spread of our invasive plant species (Table 14). As such, the sum weights of all risk variables are equal to 1.0. Weights are not quantitatively supported, and should be revised as information becomes available. Likewise, these numeric weights may not be appropriate for other study areas or species.

Table 14. Numeric weights given to risk variables.

Risk Variable	Numeric Weight
Invasive Plant Occurrences	0.30
Predictive Species Habitat Models	0.30
Vectors and Pathways	0.20
Disturbances	0.15
Soil Moisture	0.05

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 predicted suitable habitats as the two main characteristics that influence the establishment and spread of our target invasive plant species in HAFB. 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 (0.3) (Table 14). When these two variables spatially co-exist, their combined rank (added together) could account for greater than half the total possible score in the risk models.

Pathways for the spread of invasive plants and disturbed areas 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 concentrate human activity increase the likelihood of successful plant invasions. Therefore, we gave pathways and disturbances a weight of 0.20 and 0.15, respectively (Table 14).

Likewise, soil moisture regimes can greatly influence the likelihood of successful plant invasions, especially in desert environments (Brooks and Pyke 2001). Although many invasive plants are drought tolerant, areas with enhanced soil moisture availability increases the likelihood of plant establishment. Because we view this variable as an attribute to enhance habitat suitability, but not limit habitat suitability, we gave it a weight of 0.05 (Table 14).

Analytical Approach to Construct Early Detection Models

The numerical ranks and weights allowed us to combine multiple stressors, habitats, and species occurrences into a spatial model that represented areas at risk of invasion by nonnative species. To create the risk surface, we multiplied each risk variable by its weight, and then added all variables together (additive Boolean overlay), which resulted in a risk surface with values ranging from 0 to 1.0. We grouped risk scores into classes of low, moderate, and high risk (EPA 1998). We considered values > 0.50 to be areas of high risk to invasion by nonnative plants; values from 0.25-0.50 were areas of moderate risk to invasion; and values < 0.25 were considered to be areas of low risk to invasion. We averaged all eight plant risk models to estimate key areas at risk to multiple plant species incursion.

Boles Wells Water System Annex

Invasive plant predictive models and associated risk assessments were not conducted on BWWSA because spatial datasets were not available. Quickbird imagery was not obtained for the BWWSA, and the vegetation layer for the northern well area was not available. As such, we provide a description of areas to monitor in the BWWSA based on areas of disturbance, locations of known invasive plants, and potential pathways for dispersal.

RESULTS AND DISCUSSION

Sample Locations

We surveyed historic invasive plant locations and systematically searched roads, parking lots, disturbed sites, arroyos, springs, and other water sources on HAFB from May through August 2007. This sampling approach optimized our chance of detecting invasive plant populations, and yielded 879 invasive plant locations on HAFB (Table 15). Additional survey effort would likely detect other target plants or plant populations. We randomly sampled 159 (18%) of the invasive plant occurrence locations, characterizing vegetation cover. In addition, we sampled 295 random sites across HAFB (Fig. 5). We found our target invasive plants at 22 (8%) of these random sites (Table 16). Of the grouped land cover classes, communities classified as saltcedar woodlands (80%) and disturbed habitats (37%) yielded the greatest frequency of occurrence of invasive plants. We did not detect our invasive plant species in dune, wetland or playa, or creosote bush communities.

Plant Canopy Cover Estimates at Invasive Plant Locations and Random Sites

Our target invasive plants were found in areas that yielded a low mean percent native vegetation cover (< 10%) and a high percent mean unvegetated surface (> 60%) (Table 17), which are typical characteristics in areas with high to moderate intensity disturbance.

Table 15. Number of invasive plant occurrence locations recorded and randomly sampled on Holloman Air Force Base, New Mexico.

Target Invasive Plant	Number of Invasive Plant Occurrence Locations Recorded	Number of Locations Randomly Sampled	Percent Sampled
Russian knapweed	1	0	0.0
Five-horn smotherweed	61	20	32.8
Malta starthistle	25	11	44.0
Russian olive	4	1	25.0
African rue	250	44	17.6
Russian thistle	117	31	26.5
Saltcedar	409	50	12.2
Siberian elm	12	2	16.6
Total Number of Samples	**879**	**159**	**18.1**

Table 16. Number of randomly sampled transects sampled that had at least one occurrence of the eight target invasive plants on Holloman Air Force Base.

Grouped Land Cover Class	Number of Transects Sampled	Number of Transects with Target Invasive Plants	Frequency of Occurrence (%)
Creosote Bush Shrubland	5	0	0
Development / Ground Disturbance	38	14	36.8
Dune	46	0	0
Fourwing Saltbush Complex	80	1	1.3
Grassland	74	2	2.7
Rock Outcrop	5	0	0
Saltcedar Woodland	5	4	80.0
Wetland/Playa	6	0	0
Xeric Riparian	36	1	2.8
Total Number of Samples	**295**	**22**	**7.5**

Five-horn smotherweed and African rue populations were found in areas with < 6% native vegetation cover and a > 70% unvegetated surface (Table 17). Likewise, Russian thistle populations were found in areas with approximately 6% native plant cover, 34% of non-native plant cover, and approximately 60% unvegetated surface area. Although Malta starthistle was found in disturbed areas, native plant cover was approximately 18% with approximately 55% unvegetated surface area. Cover estimates of invasive tree/shrub species (e.g., Russian olive, Siberian elm, and saltcedar) naturally yielded high non-native plant cover estimates (> 35%), and moderate amounts of unvegetated surface areas (~ 47%) (Table 17).

Communities represented by the grouped land cover classes yielded unvegetated surface area estimates ranging from 58 to 97%, with an average unvegetated surface area across all communities of approximately 78% (Table 17).

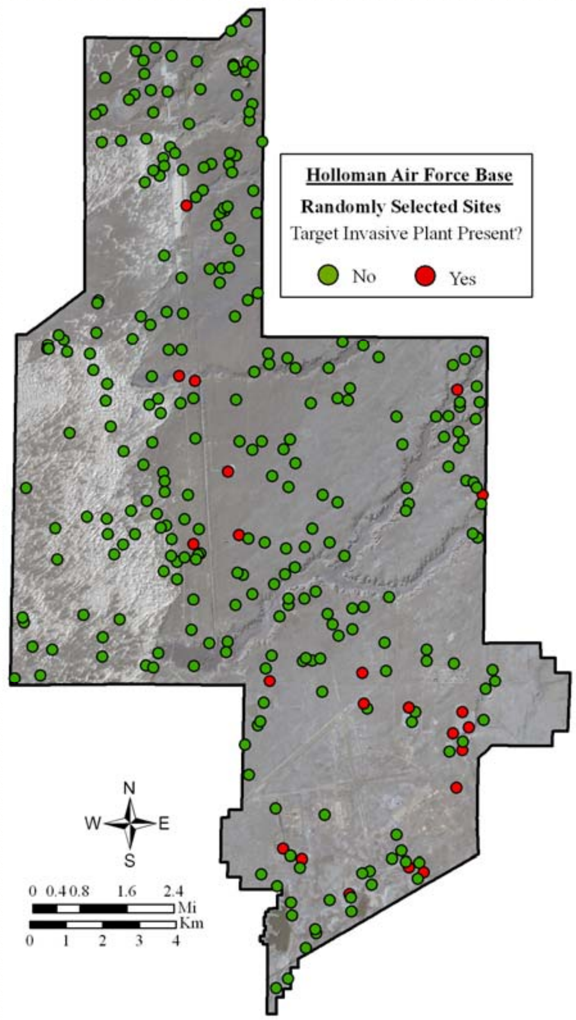

Figure 5. Location of randomly selected sites (n=295) sampled in Holloman Air Force Base.

Table 17. Percent of native vegetation, invasive plant, and unvegetated surface area along 25 m transects at sites occupied by target invasive plants and random sites selected in landscape communities on Holloman Air Force Base. Mean percentages are accompanied by standard errors in parentheses.

		Native Vegetation Percent Cover (se)				**Target**	**Unvegetated Surface Area Percent (se)**		
	Sample Size	**Annual Forbs**	**Perennial Forbs**	**Perennial Grass**	**Shrubs/ Trees**	**Invasive Plants**	**Litter**	**Bare Ground**	**Gravel / Rock**
Invasive Plant Occurrences Sampled									
Five-horn Smotherweed	20	0.6 (0.26)			1.7 (1.09)	15.8 (2.10)	0.68 (0.41)	71.8 (6.20)	9.3 (4.94)
Malta Starthistle	11	4.4 (1.16)		5.5 (2.21)	8.0 (3.12)	27.4 (2.74)	6.7 (3.2)	17.0 (5.71)	31.0 (8.21)
Russian Olive	1			5.6 (--)		48.0 (--)	3.2 (--)	43.2 (--)	
African Rue	44	2.8 (0.51)	0.1 (0.10)	1.4 (0.41)	1.6 (0.43)	22 (1.85)	2.6 (0.60)	38.9 (5.36)	30.7 (4.60)
Russian Thistle	31	1.2 (0.33)		1.9 (0.75)	3.1 (1.58)	33.9 (2.45)	0.8 (0.26)	42.1 (5.08)	17.4 (4.74)
Saltcedar	50	2.1 (0.46)	0.1 (0.05)	4.5 (0.95)	8.5 (1.07)	36.9 (3.04)	4.9 (0.55)	39.3 (2.80)	3.7 (1.64)
Siberian Elm	2			10.4 (0.80)	1.6 (1.60)	39.6 (6.00)		40.0 (12.80)	8.4 (4.40)
All Target Plants	**159**	**2.1 (0.24)**	**0.1 (0.03)**	**2.7 (0.41)**	**4.5 (0.58)**	**29.0 (1.37)**	**3.0 (0.36)**	**42.3 (2.37)**	**16.4 (2.07)**
Grouped Land Cover Classes Sampled									
Creosote Bush	5	2.7 (2.72)		2.2 (2.24)	24.8 (5.01)		1.4 (0.64)	68.8 (2.43)	
Developed	38	5.5 (0.93)		3.6 (0.89)	5.1 (1.36)	2.8 (0.84)	6.5 (1.26)	64.3 (4.88)	12.2 (4.20)
Dune	46	1.4 (0.30)	0.1 (0.07)	5.6 (1.08)	6.8 (0.88)		1.8 (0.44)	84.2 (1.87)	0.2 (0.17)
Fourwing Saltbush	80	6.8 (0.73)	0.2 (0.22)	10.6 (1.22)	9.3 (0.62)	< 0.1 (0.01)	14.4 (1.13)	58.3 (1.31)	0.4 (0.43)
Grassland	74	6.7 (0.77)		9.3 (1.34)	7.9 (0.72)	0.1 (0.08)	7.8 (0.78)	68.2 (1.42)	
Rock Outcrop	5	3.8 (1.53)		1.4 (1.44)	3.0 (1.42)		0.8 (0.62)	31.0 (9.93)	59.9 (15.24)
Saltcedar	5				4.3 (3.75)	37.3 (17.87)	14.2 (8.04)	44.2 (17.42)	
Wetland	6	2.0 (2.00)		0.4 (0.40)	1.1 (0.92)		15.7 (15.57)	80.8 (15.26)	
Xeric Riparian	36	8.8 (1.88)		8.3 (2.94)	7.3 (1.69)	< 0.1 (0.02)	5.2 (2.05)	70.3 (4.92)	
All Classes	**295**	**5.7 (0.41)**	**0.1 (0.06)**	**7.6 (0.65)**	**7.7 (0.43)**	**1.0 (0.41)**	**8.2 (0.63)**	**67.0 (1.27)**	**2.7 (0.79)**

Invasive plant cover was low (<3%) in all grouped land cover classes except saltcedar communities. Randomly sampled grouped land cover classes yielded over twice the amount of native vegetation cover than areas sampled with invasive plant occurrences (Fig. 6). The lower overall percent unvegetated surface area and native vegetation cover at invasive plant areas sampled likely indicates that invasive plants are colonizing unvegetated areas and displacing native vegetation on HAFB.

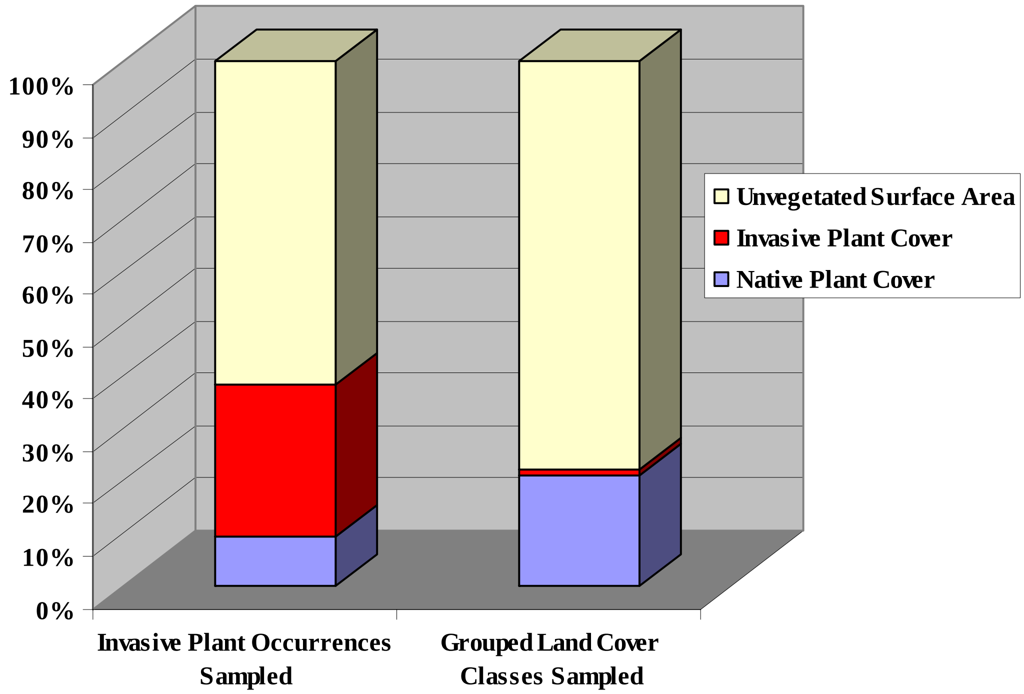

Figure 6. Unvegetated surface area, invasive, and native plant cover percentages at invasive plant occurrence locations and grouped land cover classes sampled on HAFB.

We used a point-intercept transect method that recorded the first layer of plant canopy (first foliar intercept) at each point. The first foliar intercept method has been reported to be accurate and efficient, requiring less time and less sample points than other sampling methods to obtain total vegetation cover estimates, although its efficacy decreases when measuring individual species cover when plants are intermingled or plant boundaries are not distinct (Heady et al. 1959, Winkworth et al. 1962, Bonham 1989, Hofmann and Ries 1990). Accuracy of the point-intercept transect method depends on the length of transects, and the number of points per transect.

We considered survey efficiency (time required to set up transect and conduct measurements), vegetation composition and density, and spatial resolution of our remotely sensed data in selecting transect length and point measurement distance. Long transects require greater amount of setup and survey time, and they increase the amount of variability associated with comparisons between ground data and landscape data. Likewise, long transects tend to underestimate species cover, especially when points are widely spaced, although estimates of total cover are fairly robust regardless of transect length (Bonham 1989). Our transect length (25 m; 82 ft) and point distance (20 cm; 8 in) minimized our chance of missing plant species that were present. Further, we established and surveyed transects in < 20 min, which allowed for a greater number of samples to be collected in a smaller time window.

In a related study (same methods and set of randomly sampled locations), Kamienski (2008) compared the first foliar intercept method to a multi-strata sampling approach that measured the top canopy cover, middle (understory), and ground-cover at each point intercept. The study results were limited due to low sample sizes (n=20) and temporal differences in measurements between the two methods. However, Kamienski (2008) reported > 0.80 correlation between the two methods, indicating that the first foliar intercept method adequately represented the structural composition and total plant cover. Kamienski (2008) results may be a function of the limited plant species structural composition and canopy overlap at the sampled sites on HAFB, however; they do provide support for the plant cover estimates provided in this report.

Predictive Habitat Models

The number of occurrence points used to generate individual species models ranged from 20-360 (Table 18). Small sample sizes (< 15 occurrences) occurred with Russian knapweed, Russian olive, and Siberian elm. As such, deductive (descriptive) habitat models were created for these species. Inductive predictive habitat models were created for five-horn smotherweed, Malta starthistle, African rue, Russian thistle, and saltcedar using Maxent software.

Performance of Deductive Models

The performance of the deductive models could not be adequately determined because of low sample sizes. However, we assessed model performance by using locations of known plant occurrences. We only found one Russian knapweed location on HAFB. This location was in the Boles Wells Water System Annex area. We were unable to locate the historical occurrence of plants found in Hay Draw (Reiser et al. 2000). As such, we did not assess model performance for Russian knapweed (Fig. 7). We had four occurrences of Russian olive on HAFB. All four occurrences coincided with suitable habitat predicted with the deductive model (Fig. 8). Likewise, we had 12 occurrences of Siberian elm on HAFB. Eleven (92%) of these occurrences were in suitable habitat predicted with the deductive model (Fig. 9). However, model performance metrics are not robust due to low sample sizes and because these models incorporated only one environmental variable (land cover). Models constructed with few parameters and low sample sizes have a high degree of bias (Franklin et al. 2001). These models should be updated as additional information becomes available.

Table 18. Number of georeferenced target invasive plant occurrences used to construct predictive habitat models with Maxent software on Holloman Air Force Base.

Common Name	Number of Occurrences	Number of Training Samples	Number (Percent) of Test Samples	
Russian knapweed	1	-	-	
Five-horn smotherweed	61	51	10	(16%)
Malta starthistle	25	20	5	(20%)
Russian olive	4	-	-	
African rue	250	213	38	(15%)
Russian thistle	117	97	20	(17%)
Saltcedar	409	360	49	(12%)
Siberian elm	12	-	-	
Total	**879**	**741**	**122**	

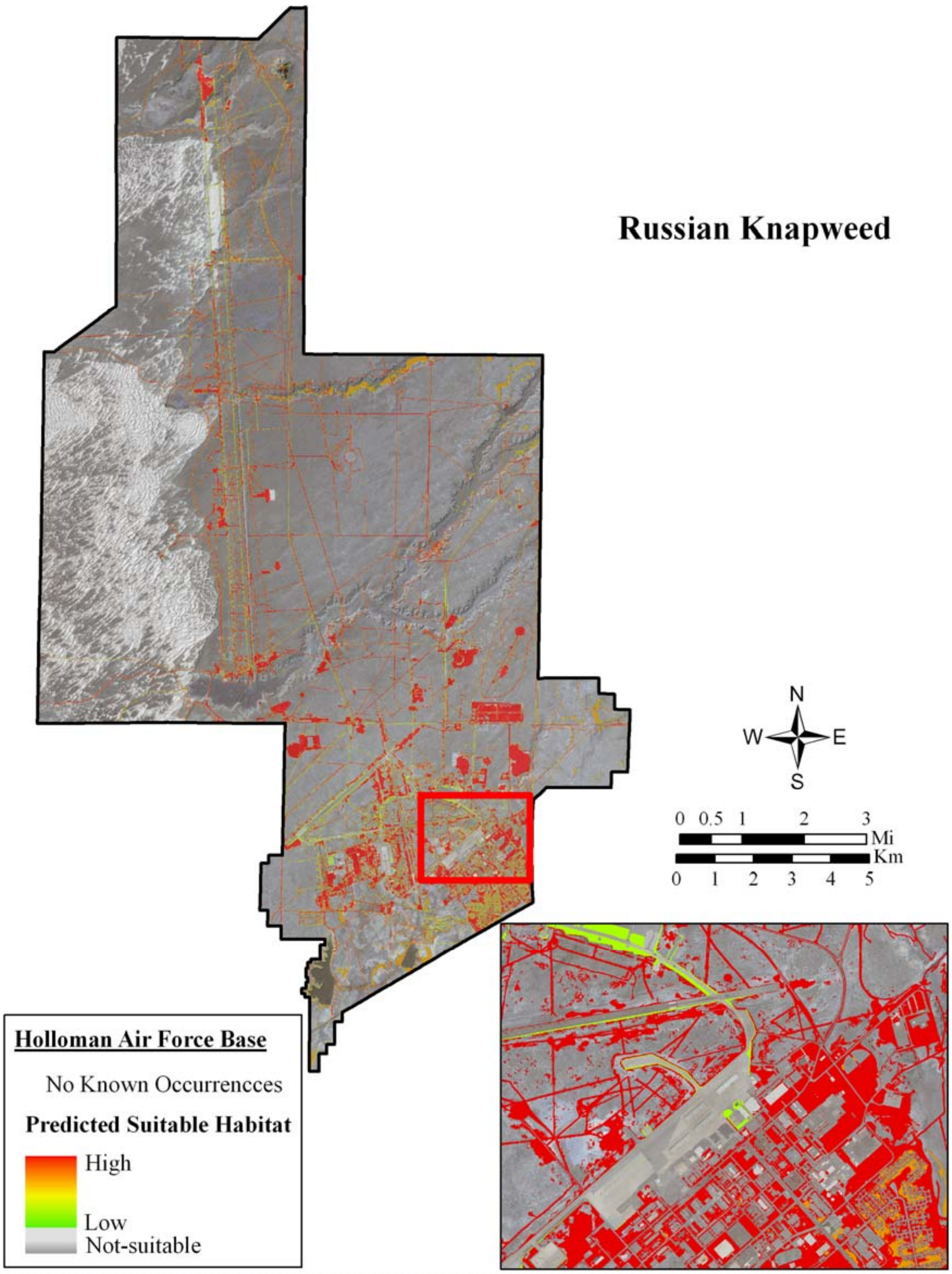

Figure 7. Potential distribution of Russian knapweed suitable habitat on Holloman Air Force Base estimated from the land cover map produced by Muldavin et al. (1997).

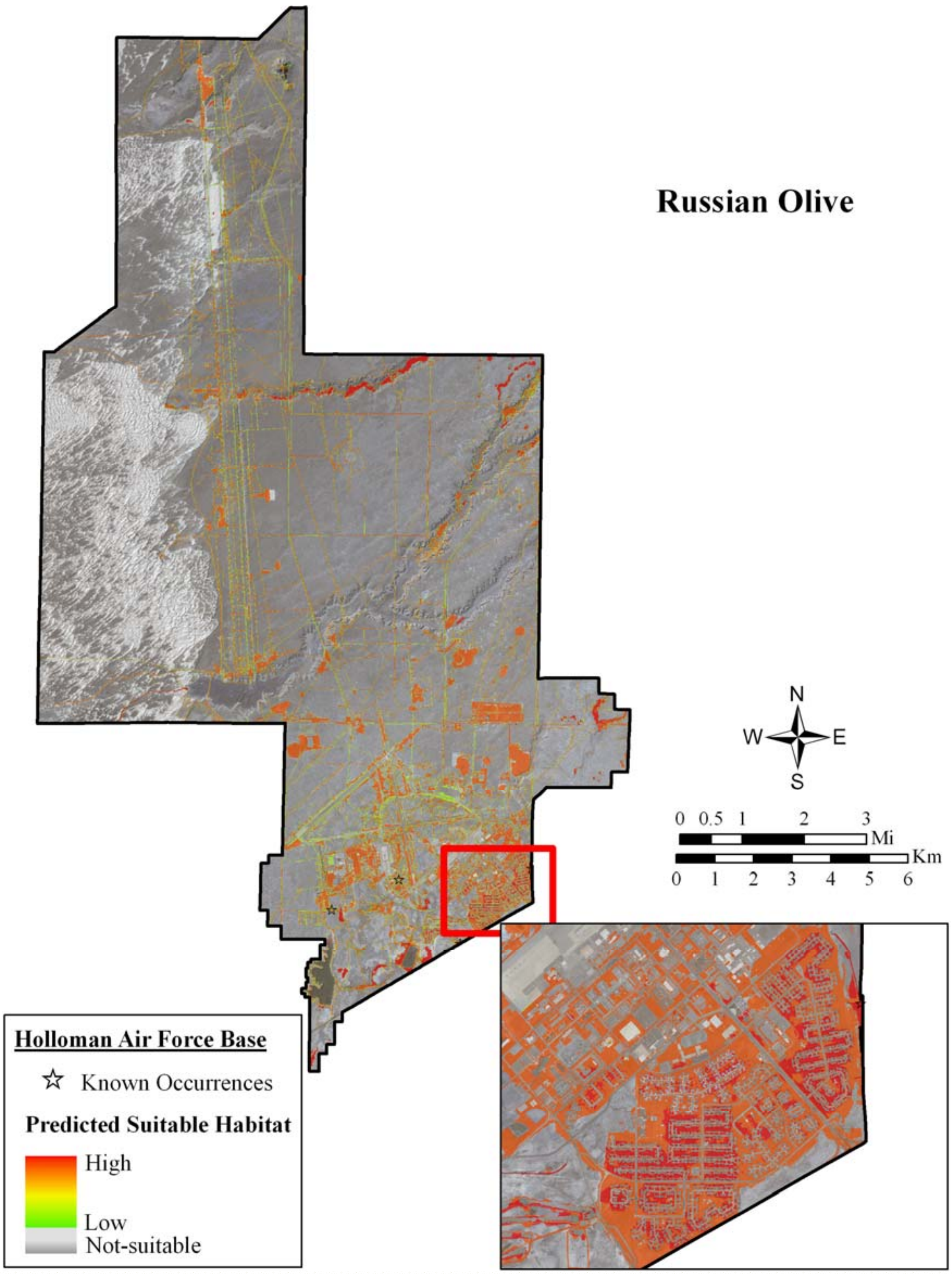

Figure 8. Potential distribution of Russian olive suitable habitat on Holloman Air Force Base estimated from the land cover map produced by Muldavin et al. (1997).

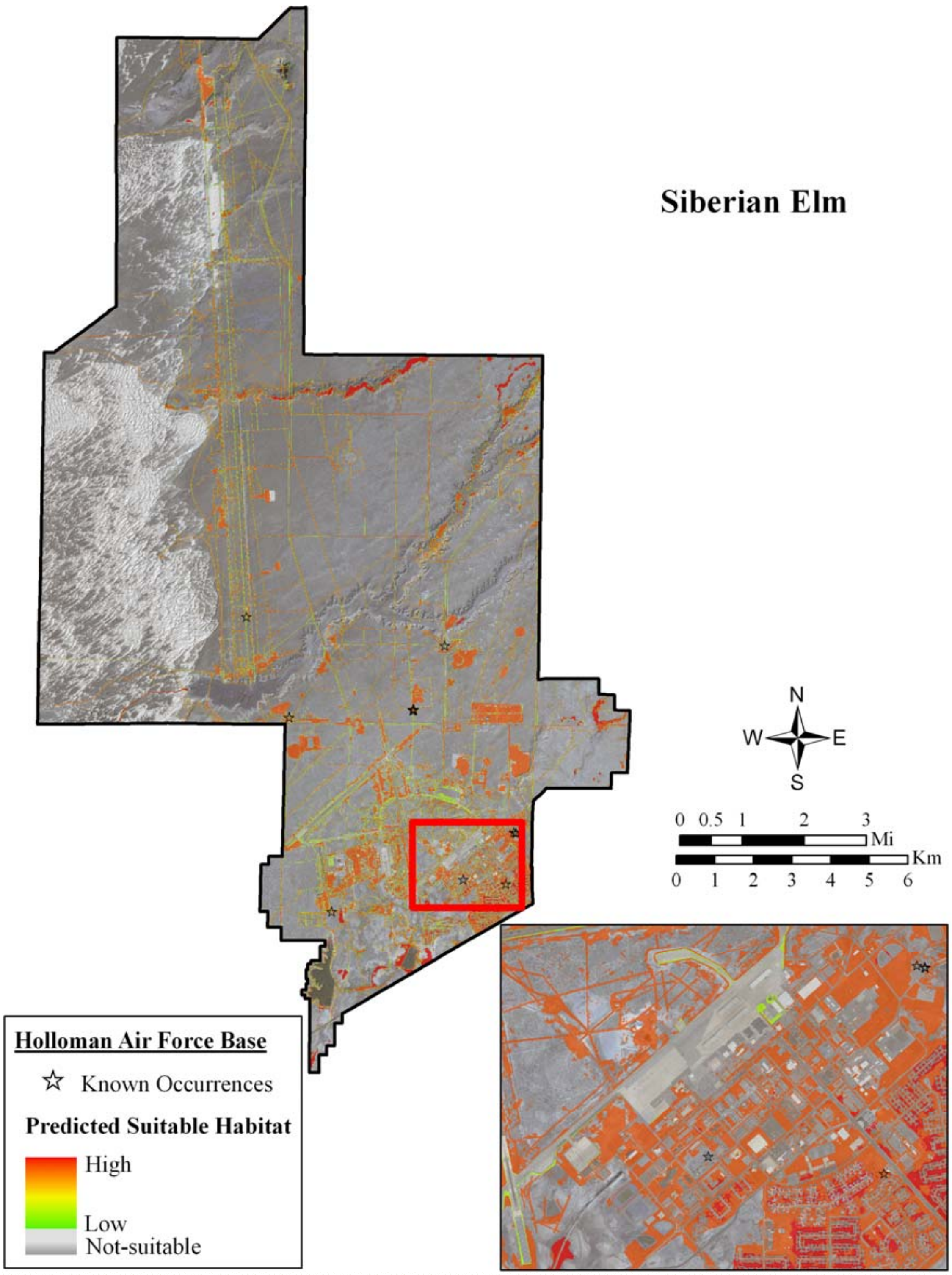

Figure 9. Potential distribution of Siberian elm suitable habitat on Holloman Air Force Base estimated from the land cover map produced by Muldavin et al. (1997).

Performance of Inductive 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 verses predicted area provides insight to model performance. An optimal model would have an omission rate close to the predicted omission (diagonal line).

Visual interpretations of ROC and omission curves in Figures 10-14 indicate that all of the plant models performed better than a random prediction. Training AUC values ranged from 0.91 to 0.99, and test AUC values ranged from 0.93 to 0.98, indicating all models yielded a high discriminate ability to detect potential suitable habitat for each target species (Table 19).

Five-horn smotherweed, Malta starthistle, and saltcedar models yielded the highest AUC values ($\geq$ 0.97), highest model gain values ($\geq$ 2.0) and the lowest error of omission ($\leq$ 16%; P<0.01) (Table 19). Model AUC and gain values for these three plants indicate that the likelihood that a random positive plant occurrence and a random negative location were accurately predicted approximately 97% of the time, and the average sample likelihood was exp(2.0) ≈ 11 times higher than that of a random background pixel. However, model assessment values for five-horn smotherweed and Malta starthistle may not be robust due to the low number of test samples (Table 18). Model predictions may be improved by locating and including additional samples in the modeling effort. Nevertheless, our results indicate that useful predictive models may be created with < 50 occurrence locations with Maxent.

African rue and Russian thistle predictive models yielded slightly lower AUC values (0.91 and 0.93, respectively), lower gain values (≈ 1.2), and moderate errors of omission (37% and 27%, respectively). These two plants were widely distributed across HAFB, and were likely actively spreading. African rue could be found in most disturbed areas on HAFB, and there were

Table 19. Performance measurements for invasive species predicted habitat models in Holloman Air Force Base.

	Threshold Independent				Threshold Dependent			
	Training Samples		Test Samples		Equal Test Sensitivity and Specificity			
Common Name	AUC[1]	Gain[2]	AUC[1]	SD	Logistic Threshold	Training Omission Rate	Test Omission Rate	P-value
Five-horn smotherweed	0.968	2.028	0.968	0.014	0.185	0.06	0.10	9.10E -9
Malta starthistle	0.978	2.467	0.976	0.008	0.288	0.15	0.0	3.03E -7
African rue	0.906	1.212	0.942	0.012	0.362	0.24	0.13	1.86E-41
Russian thistle	0.932	1.388	0.928	0.014	0.298	0.12	0.15	7.11E-12
Saltcedar	0.987	2.449	0.983	0.006	0.283	0.05	0.06	0.00 E -0

[1] AUC = Area under the curve derived from receiver operating characteristic (ROC) curves for each plant.
[2] Regularized model gain derived from training samples.

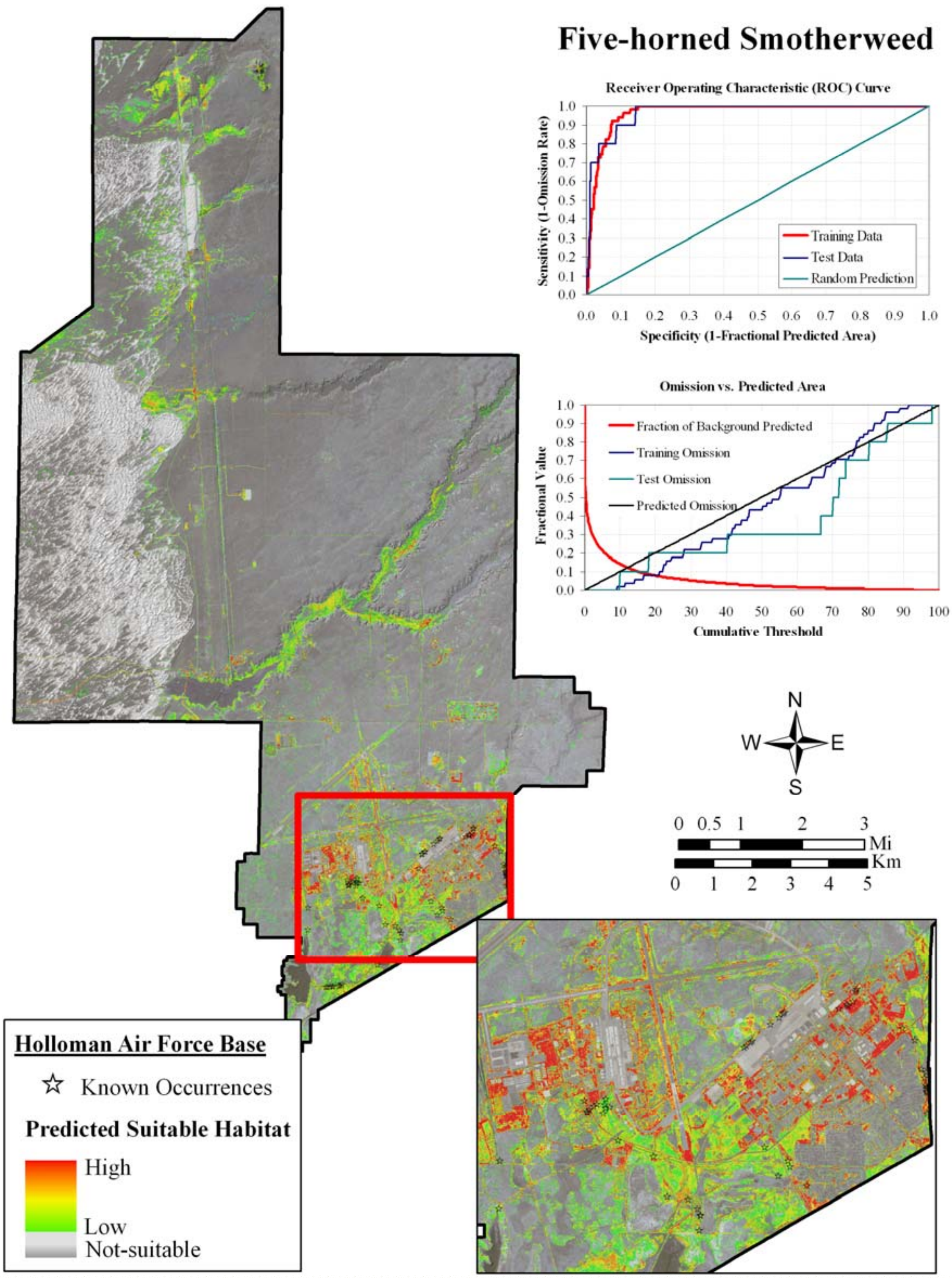

Figure 10. Potential distribution of five-horn smotherweed suitable habitat on Holloman Air Force Base using maximum entropy of 51 presence locations. Independent model evaluation was conducted using 10 test samples.

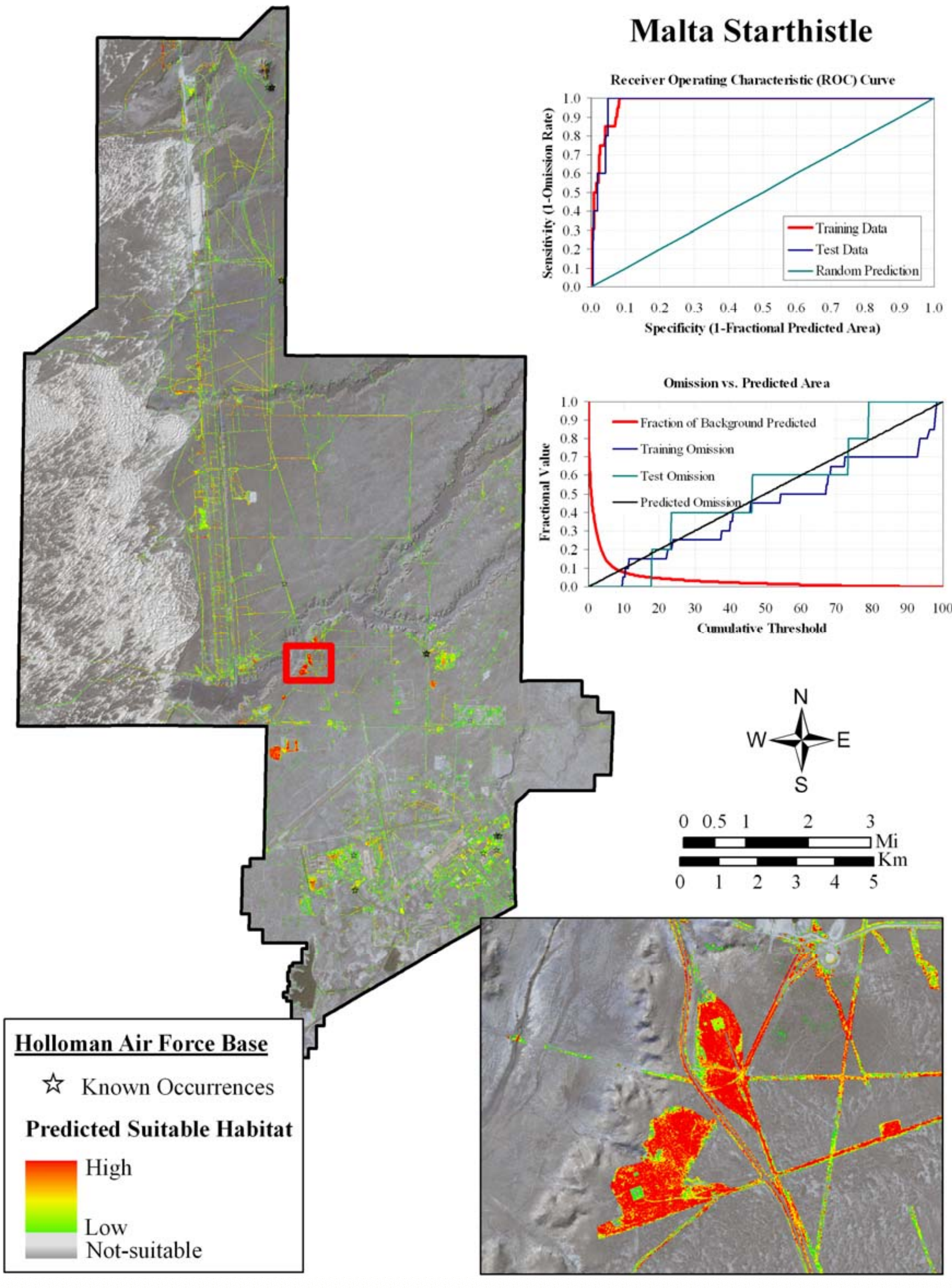

Figure 11. Potential distribution of Malta starthistle suitable habitat on Holloman Air Force Base using maximum entropy of 20 presence locations. Independent model evaluation was conducted using 5 test samples.

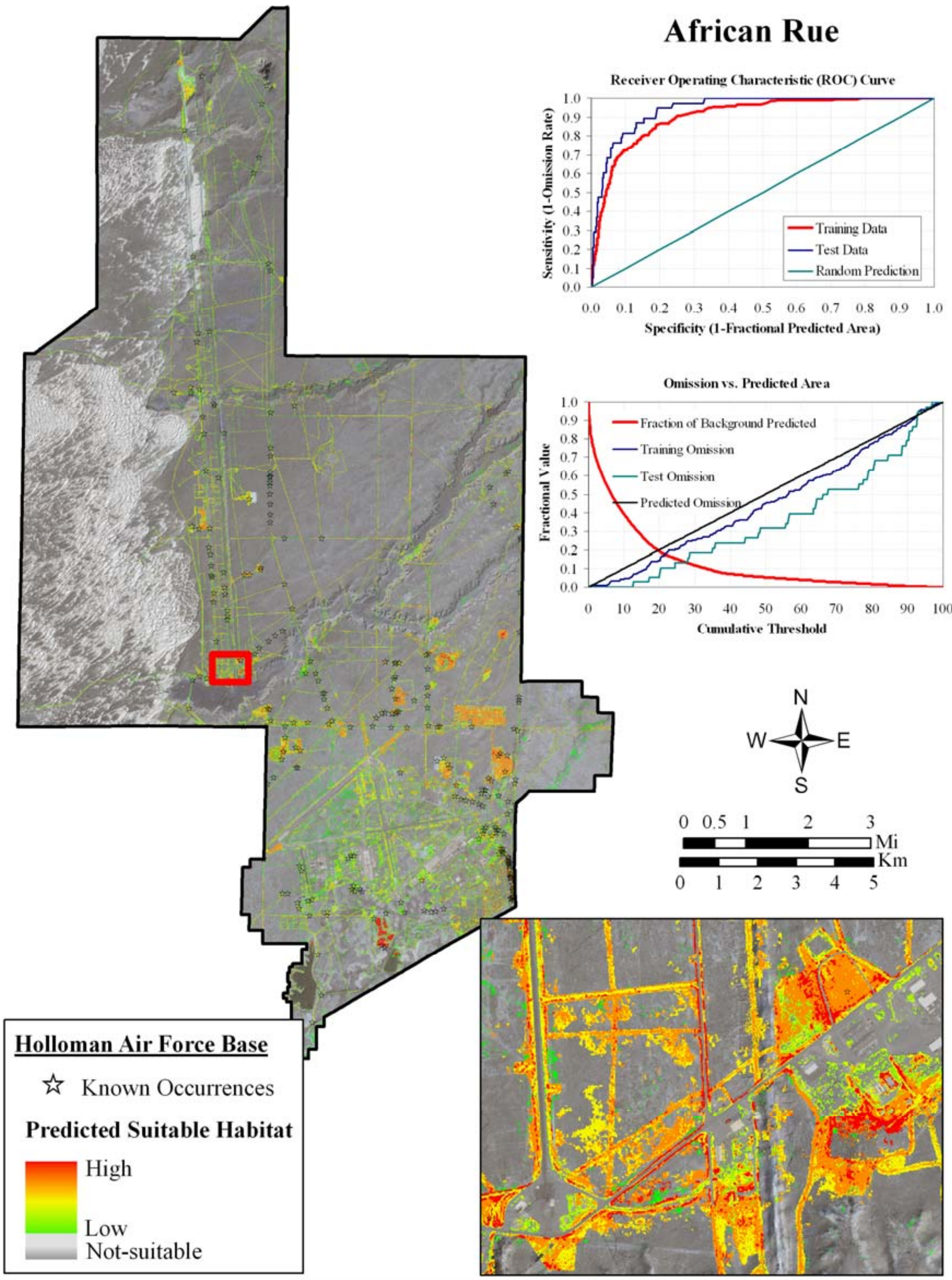

Figure 12. Potential distribution of African rue suitable habitat on Holloman Air Force Base using maximum entropy of 213 presence locations. Independent model evaluation was conducted using 38 test samples.

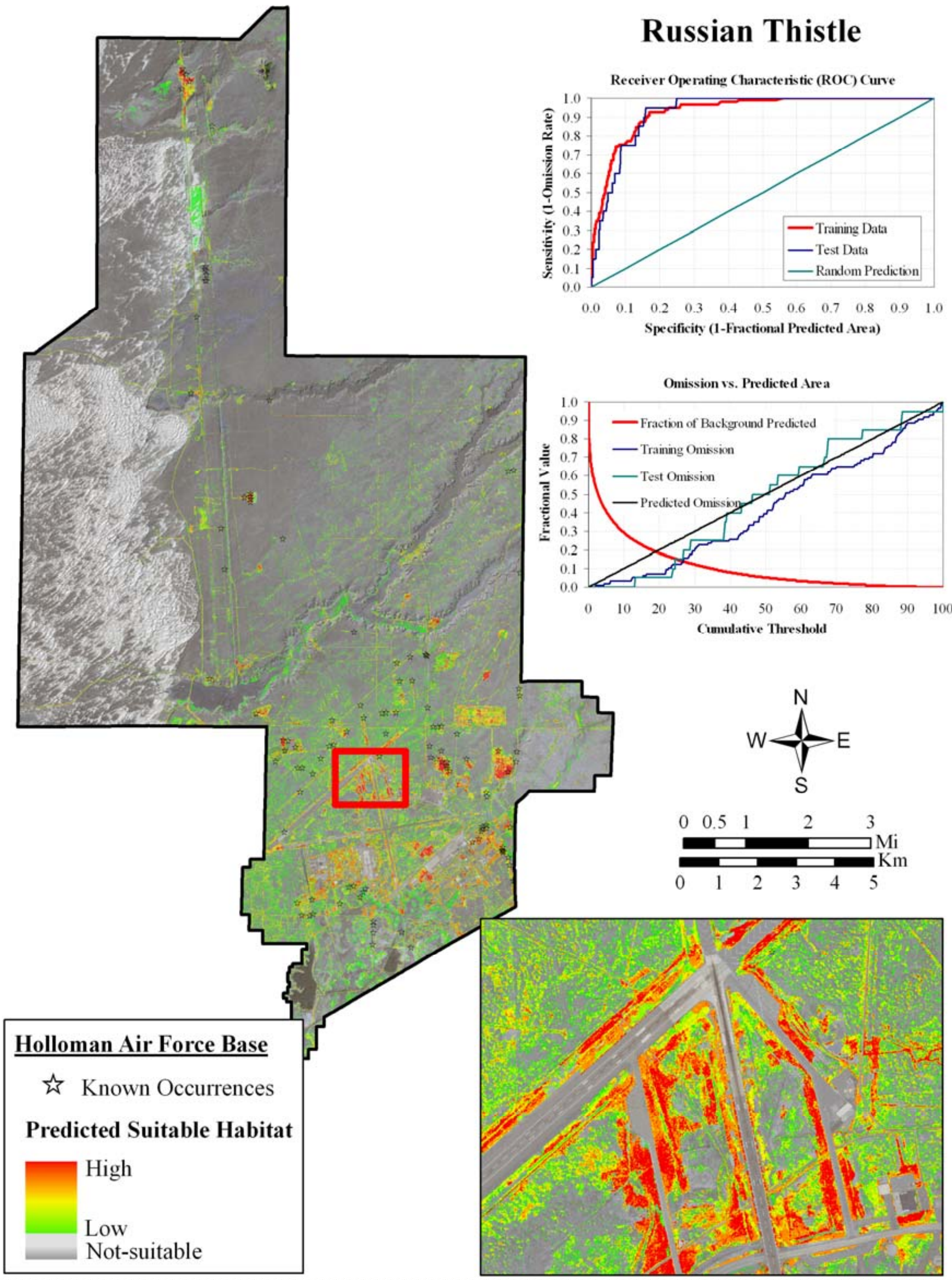

Figure 13. Potential distribution of Russian thistle suitable habitat on Holloman Air Force Base using maximum entropy of 97 presence locations. Independent model evaluation was conducted using 20 test samples.

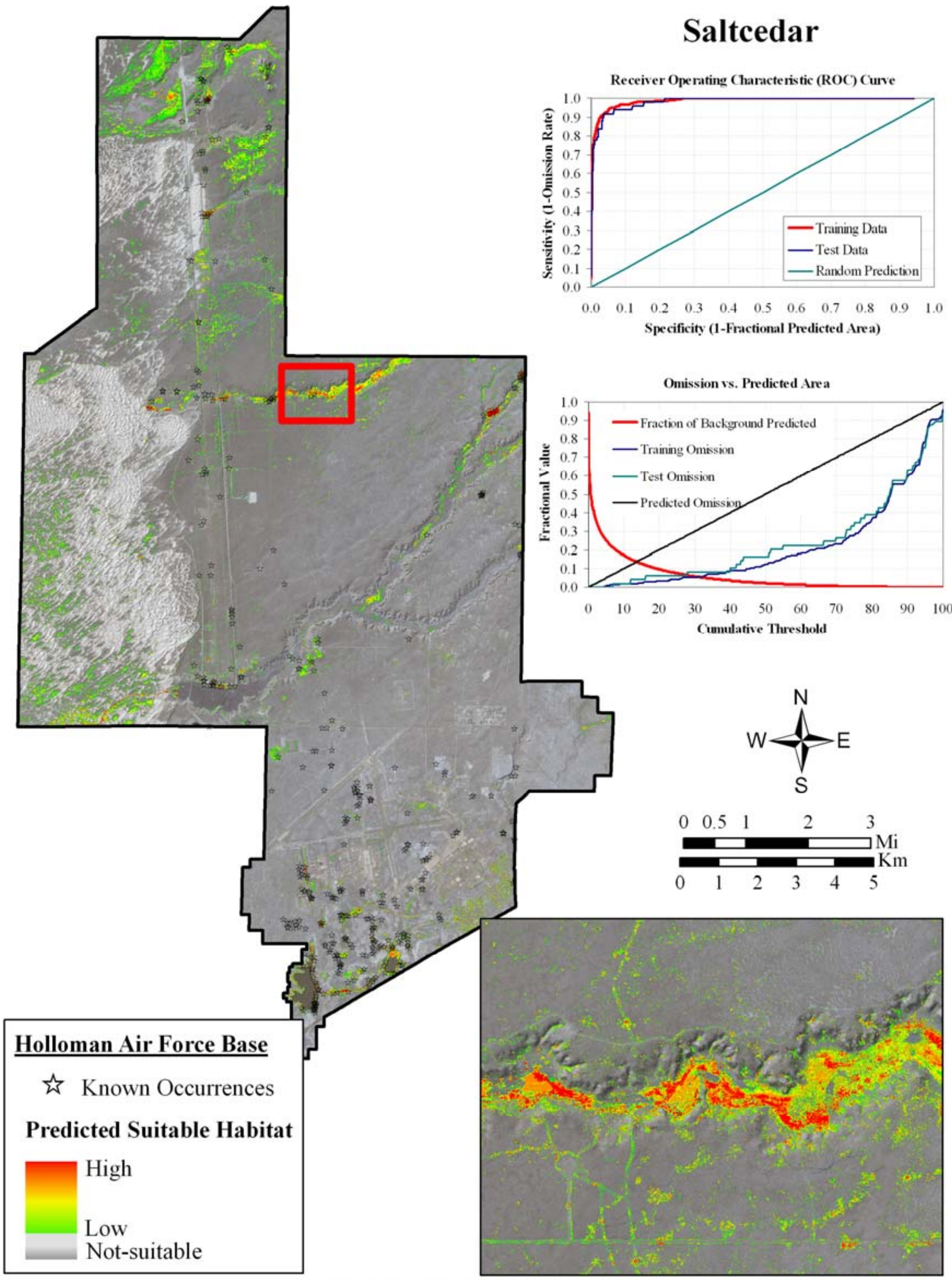

Figure 14. Potential distribution of saltcedar suitable habitat on Holloman Air Force Base using maximum entropy of 360 presence locations. Independent model evaluation was conducted using 49 test samples.

indications that this plant was spreading into four-wing saltbush communities (Reiser et al. 2000). Since our African rue predictive model was greatly influenced by our land cover dataset (Table 20), inaccuracies in mapping four-winged saltbush habitats and disturbed areas would contribute to model omission errors. Likewise, areas that were disturbed after the land cover dataset was created would contribute to model omission error. These inconsistencies could account for the moderate error of omission observed in the African rue predictive model as well as the other plant models.

We found Russian thistle occurring in large clumps and individual plants in a variety of habitat types across HAFB; including disturbed areas, fields, roadsides, and desert communities. Field sampling techniques may have failed to detect Russian thistle plants in other habitat types. For example, Russian thistle was detected at only one out of the 257 random sites sampled in land cover classes other than developed or disturbed. As such, additional sampling effort in other habitat classes may have been necessary to better quantify habitat associations and improve model predictions. Likewise, these predictive models may be improved by using presence/absence modeling approaches.

Model Variable Selection

Predictive models that incorporate the largest number of possible variables are considered to have the most flexibility in fitting the observed data (White 2001). However, a large number of input variables typically increase the amount of variability which decreases model precision, effectively decreasing our ability of accurately predicting suitable habitat. Conversely, models with few parameters are typically more precise because all the data are being used to estimate 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). Phillips et al. (2004) echoed the need for parsimonious models that are derived with Maxent. The authors noted that models with a larger number of variables tend to overfit small training sets, although models that have a large number of variables may provide more accurate predictions for large training sets.

Table 20. Relative percent contribution of environmental variables to each invasive plant predictive habitat model on Holloman Air Force Base.

		Relative Percent Contribution							
						Vegetation Indices			
		Spectral Bands					Tassel Cap		
Common Name	**Number of Model Variables**	**Band 1**	**Band 2**	**Band 3**	**Band 4**	**NDVI**	**Green**	**Wet**	**Land Cover**
Five-horn smotherweed	3	7.1				46.8			46.1
Malta starthistle	3			11.8				8.1	80.1
African rue	3				6.5	6.5			87.0
Russian thistle	4		20.1			9.7		12.7	57.5
Saltcedar	3			84.3			4.6		11.1

We used analyses provided in Maxent and correlation analysis to evaluate and reduce the number of model variables. We first analyzed all nine environmental variables and removed variables that appeared to provide little contribution to model predictions. We then eliminated correlated variables which reduced the amount of variables used in each plant model to 3-4 variables. All models contained one band of spectral data, a vegetation index, and the land cover class dataset (Table 20). However, each plant model was represented by a different set of variables and different percent contributions. Land cover class yielded high percent contribution ($\geq$ 80%) for Malta starthistle and African rue, and moderate percent contribution (between 45-60%) for five-horn smotherweed and Russian thistle. Land cover class only contributed to 11% of the saltcedar model. The Tassel Cap soil/vegetation brightness index was not used for any of the final plant models.

Our results underline the need to evaluate multiple environmental variables when creating multiple species predictive models. Likewise, analyses provided in Maxent when combined with correlation analyses can provide effective parsimonious models.

Amount and Distribution of Predicted Suitable Habitats

Maxent outputs include a logistic surface with values ranging from 0-1. 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 19). Values greater than the logistic threshold indicate suitable habitats, while thresholds less than the logistic 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 on HAFB.

Our results indicate that Malta starthistle and saltcedar may have the most limited potential distribution on HAFB, with approximately $\leq$ 6% of the base surface modeled as potential habitat (Table 21). Malta starthistle was first detected in 2005-2006, and rapidly spread from one known location near the garbage disposal area, to populations scatted along roadsides and parking facilities throughout the base, including as far north as Tula Peak. Modeled potential habitat for Malta starthistle was fairly restricted to roadways and moderately disturbed sites near primary roadways (Fig. 11). The distribution of known occurrences and modeled distribution would implicate vehicles as the primary source for the transfer of propagules (seeds) on HAFB (Roche and Roche 1991). Roadside mowing of plants with mature seeds may further disperse populations.

Muldavin et al. (1997) mapped 131 ha (323 ac) of saltcedar on HAFB. This mapped distribution was based on analyses that identified populations that existed in 1997, but not the amount of suitable habitat area for saltcedar. Holloman Air Force Base engaged in an active saltcedar control effort in 2005-2006, greatly reducing the amount of saltcedar on the base by our 2007 field survey. We used > 400 locations of saltcedar for our predictive model (Table 18). These locations were widely distributed across the base. Our predictive model identified approximately 1,323 ha (3,270 ac) of potential suitable habitat on HAFB for saltcedar (Table 21, Fig 14). Most of the potential distribution was located in drainages that bisect the base. Likewise, suitable habitat occurred in lowland habitats and next to water sources in the southern part of the base.

Our analyses predicted approximately 2,128 ha (5,257 ac) of suitable habitats on HAFB for Russian knapweed, five-horn smotherweed, Russian olive, and Siberian elm (Table 21). As previously noted, Russian knapweed has been found in Hay Draw, but was not present during our field surveys. Suitable habitats for Russian knapweed were modeled primarily around commercial and housing areas in the southern part of the base, and in Hay Draw (Fig. 7). Russian olive and Siberian elm were found primarily around old ranch sites and developed areas, perhaps planted as a shade tree. As such, modeled suitable habitat for Russian olive and Siberian elm was primarily around commercial and housing areas in the southern part of the base (Figs. 8, 9). High suitable habitat for five-horn smotherweed was predicted in the southern part of the base around the developed areas and wetlands. Moderate suitable habitat was modeled in the lowlands and drainages and along some roadways throughout the base (Fig. 10).

Our analyses indicated that Russian thistle and African Rue may have the widest potential distribution across HAFB. We estimated approximately 3,120 ha (7,710 ac; 15%) of suitable area for Russian thistle on HAFB (Table 21, Fig. 13). High suitable habitat occurs along roadsides and disturbed areas through the base; it was especially prevalent south of Lost River where potential habitat occurred in fourwing saltbush and grassland communities. Potential suitable habitat for African Rue was widespread and encompassed approximately 15% of the base (Table 21, Fig. 12). Roadsides, parking lots, and disturbed areas were modeled as high potential habitat across all of HAFB. This was consistent with habitat associations described by Parker and Reiser (1997) and our observations of existing populations on HAFB. Given the moderate errors of omission with the Russian thistle and African Rue predictive models (37% and 27%, respectively), our estimates of the amount of suitable habitat for these species may be conservative.

Table 21. Amount of predicted suitable habitat for each invasive plant species on Holloman Air Force Base.

	Not-suitable Habitat			Suitable Habitat		
Common Name	**Hectares**	**Acres**	**Percent**	**Hectares**	**Acres**	**Percent**
Deductive Models						
Russian knapweed	19,225	47,505	90.4	2,044	5,050	9.6
Russian olive	19,095	47,185	89.8	2,173	5,370	10.2
Siberian elm	19,095	47,185	89.8	2,173	5,370	10.2
Inductive Models						
Five-horn smotherweed	19,141	47,297	90.0	2,128	5,257	10.0
Malta starthistle	20,216	49,955	95.1	1,052	2,600	4.9
African rue	18,003	44,487	84.6	3,265	8,067	15.4
Russian thistle	18,148	44,845	85.3	3,120	7,710	14.7
Saltcedar	19,945	49,284	93.8	1,323	3,270	6.2

Early Detection Models Based on Risk Assessments

Our assessments were designed to evaluate the risk of our target invasive plants either becoming established, or expanding their distribution. Our approach yielded semi-quantitative, non-probabilistic, categorical landscape risk models which were similar to other ecological relative risk approaches (Andersen et al. 2004b, Landis 2004). The ranks and weights provided to risk variables were not based on quantified data. As such, 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. Nevertheless, our risk assessment approach allowed us to combine species occurrences, multiple stressors, and potential habitats, into a spatial model that represented areas at risk of invasion by nonnative species. Additional information could easily be incorporated into these risk models, including areas that host sensitive plant, wildlife, cultural, or other natural resources.

Areas at Risk of Invasion

Areas at risk of invasive plant incursion occurred throughout HAFB (Figs 15-22). Areas modeled where our target invasive plants may first establish and then spread are the residential and commercial areas in the southern part of the base, areas around installation and test track infrastructures, and areas around paved and gravel roadsides and parking lots.

Although Russian knapweed is not a current problem on HAFB, approximately 1,310 ha (3,237 ac) were considered to be at high risk for Russian knapweed incursion or spread, with an additional 2,345 ha (5,794 ac) at moderate risk (Table 22). Modeled areas at high risk of knapweed incursion were primarily in the southern part of the base around commercial and residential areas (Fig. 15). Likewise, Hay Draw and areas next to the test track were considered at high risk of Russian knapweed occurrence.

Approximately 4,739 ha (11,712 ac; 22%) of HAFB were considered at high and moderate risk to five-horn smotherweed incursion, which represents most of the area around residential and commercial areas, along roadsides, and wetlands and arroyos (Fig. 16). These areas also correspond to areas at risk for other target invasive plants. For example, high risk areas were primarily modeled around known smotherweed populations. Some of these areas also hosted populations of African rue and Russian thistle. The competitive advantage of five-horn smotherweed, or the other target invasive plants, over native plants or other invasive plants is currently unknown. Similarly, our analyses indicated that approximately 21% of HAFB was at high and moderate risk for Malta starthistle incursion (Table 22). The distribution of modeled risk areas was primarily disturbed areas around residential and commercial development in the southern part of the base, and roadsides and disturbed areas in the middle to north part of the base (Fig. 17). The modeled risk distribution is more extensive than the current distribution of starthistle on the base. Malta starthistle was first detected near the garbage disposal site on Vandergrift road in 2005 (J. Dye, personal comm.). During our surveys, we found additional plant populations in the commercial/military area in the southern part of the base, along the Tula Peak road, and on the Tula Peak overlook. This plant appears to be rapidly spreading and may

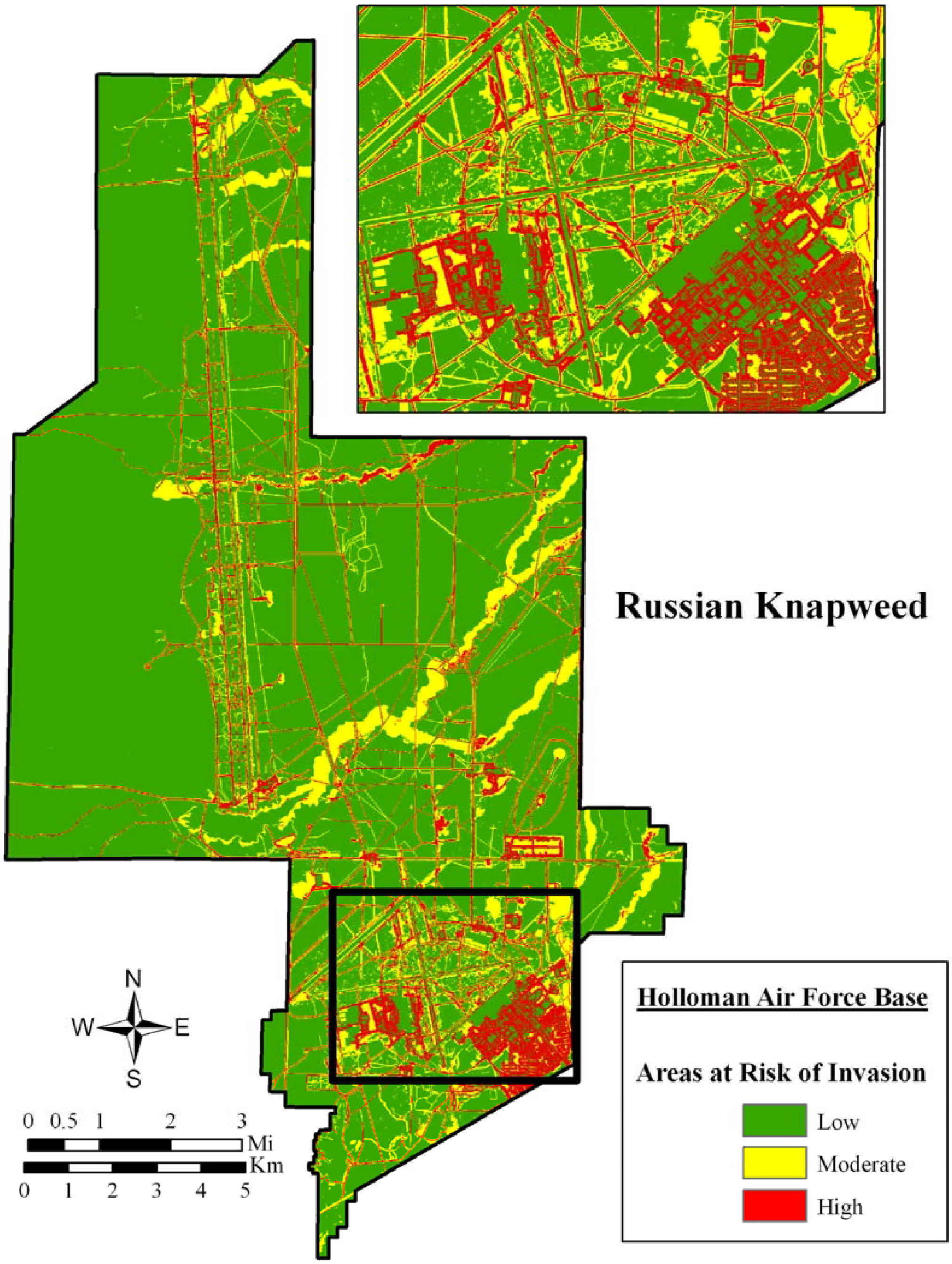

Figure 15. Areas at risk of Russian knapweed incursion or spread on Holloman Air Force Base. The likelihood of Russian knapweed incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

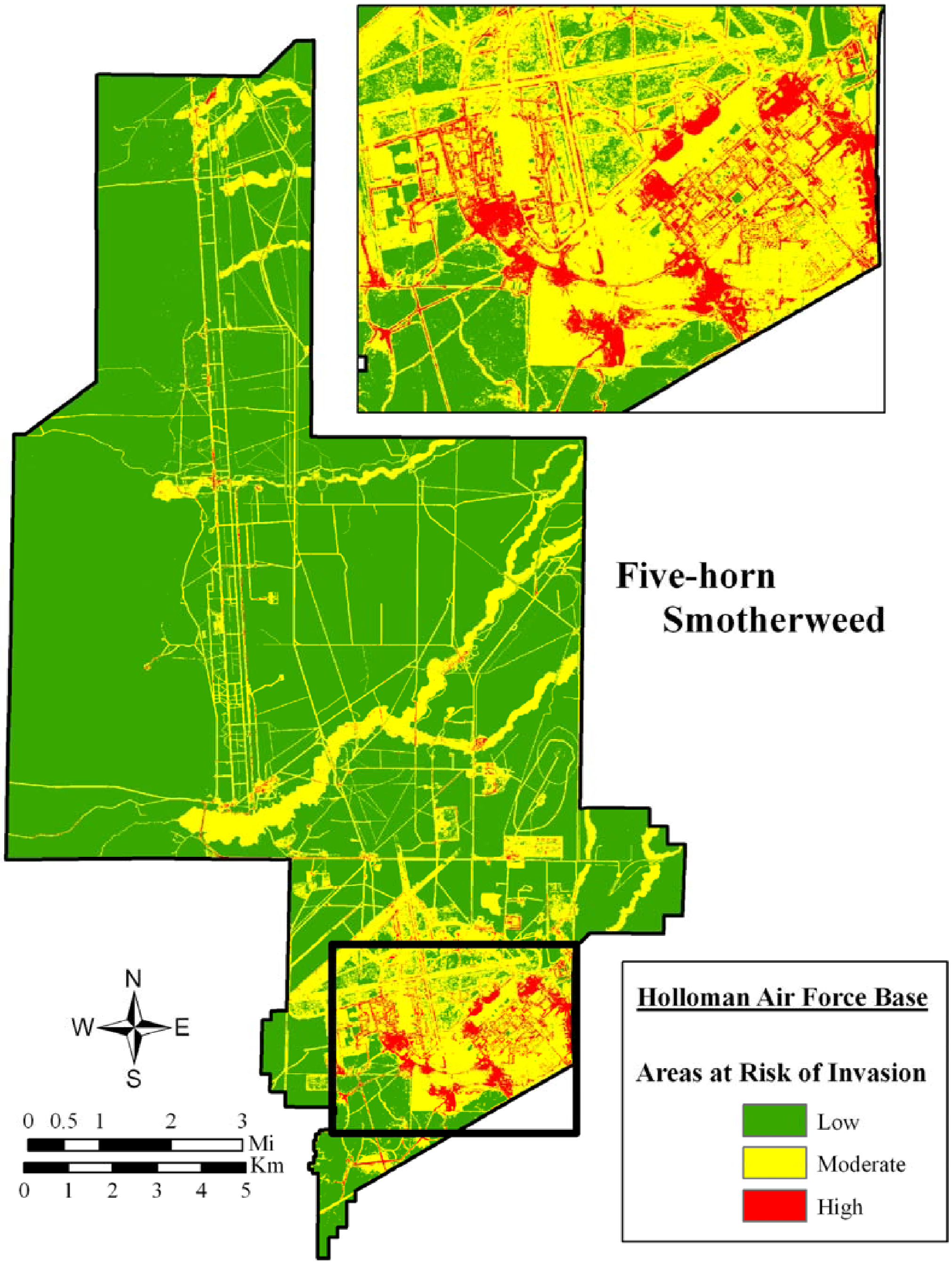

Figure 16. Areas at risk of five-horn smotherweed incursion or spread on Holloman Air Force Base. The likelihood of five-horn smotherweed incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

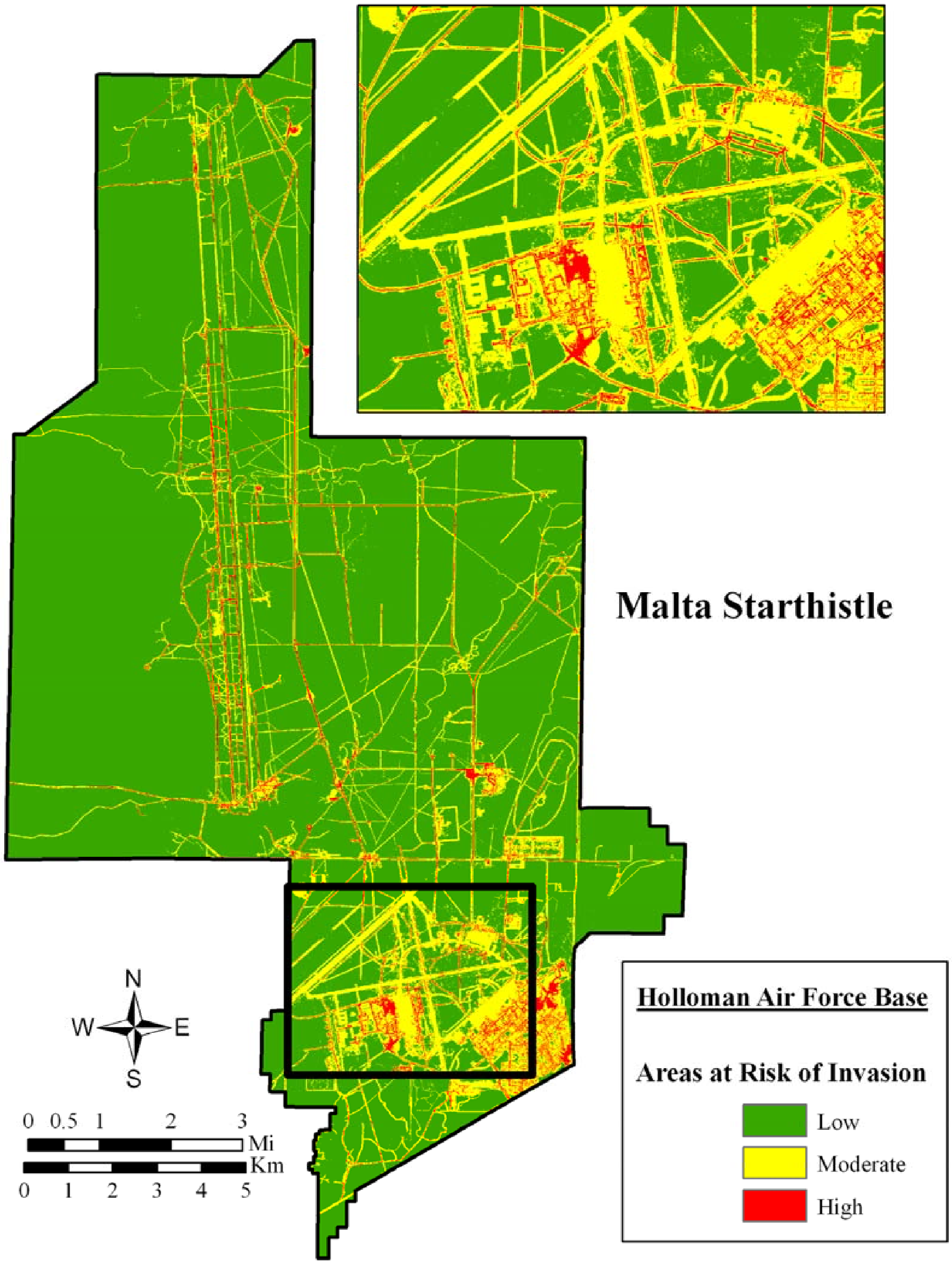

Figure 17. Areas at risk of Malta starthistle incursion or spread on Holloman Air Force Base. The likelihood of Malta starthistle incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

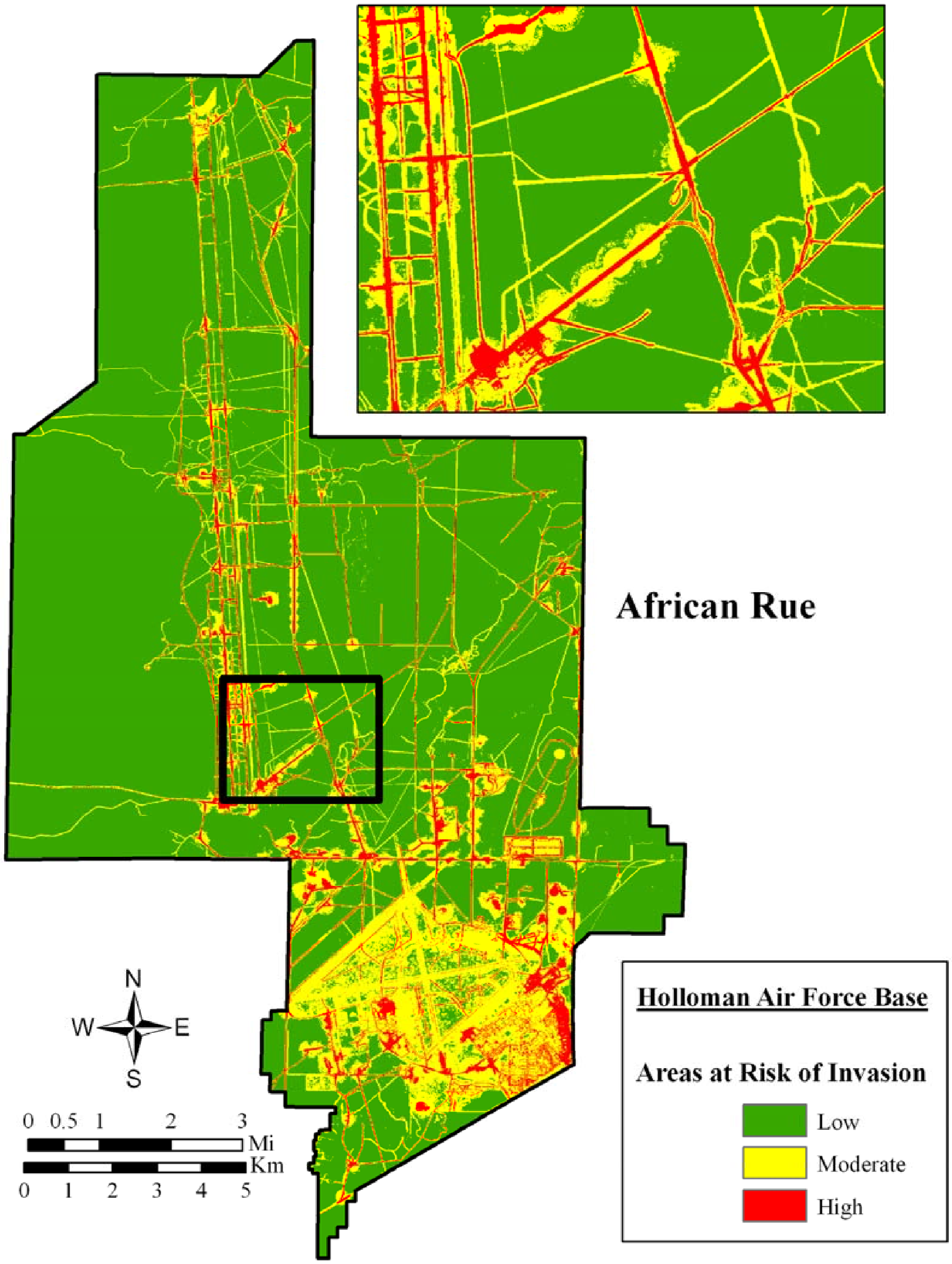

Figure 18. Areas at risk of African rue incursion or spread on Holloman Air Force Base. The likelihood of African rue incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

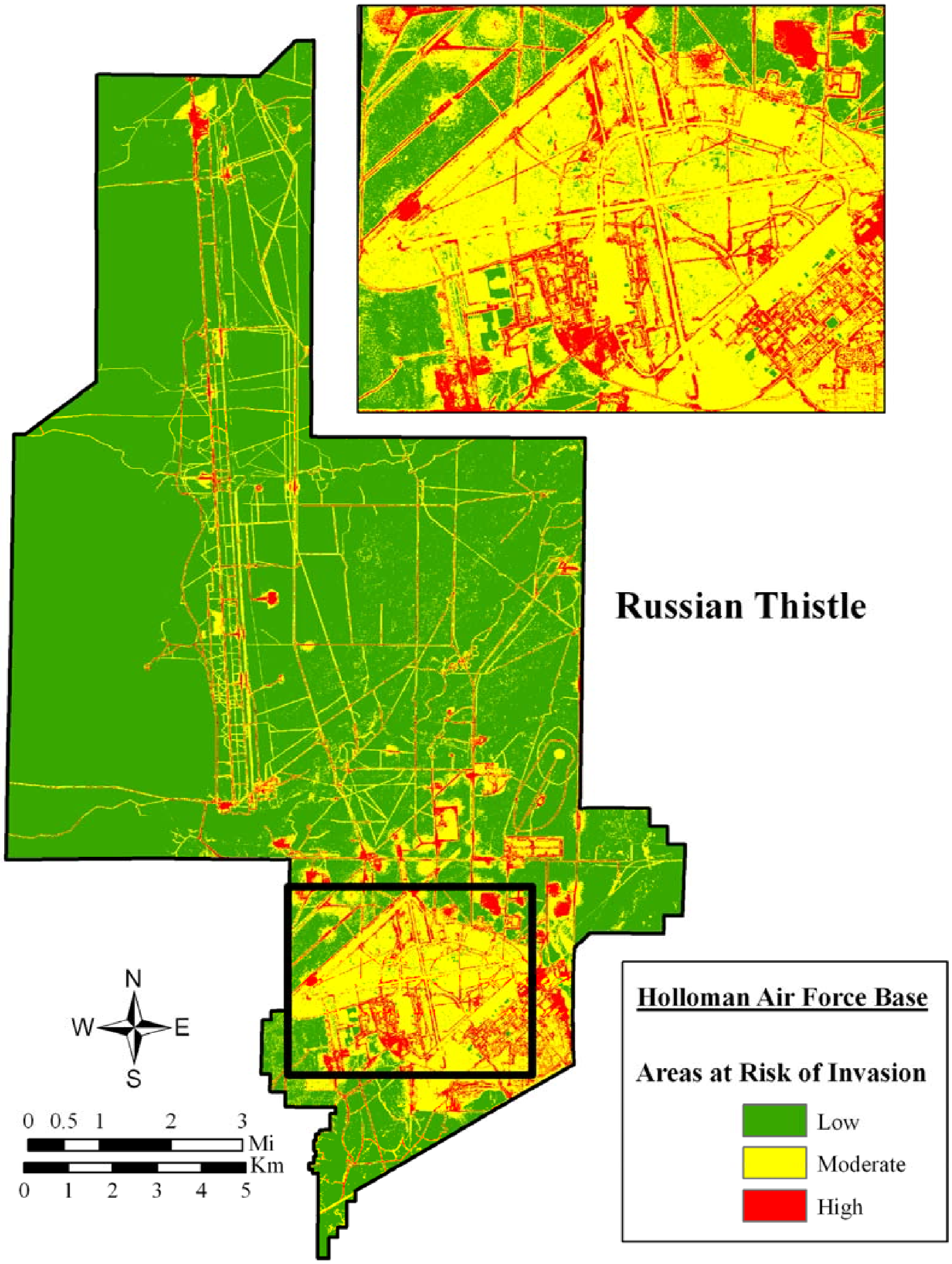

Figure 19. Areas at risk of Russian thistle incursion or spread on Holloman Air Force Base. The likelihood of Russian thistle incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

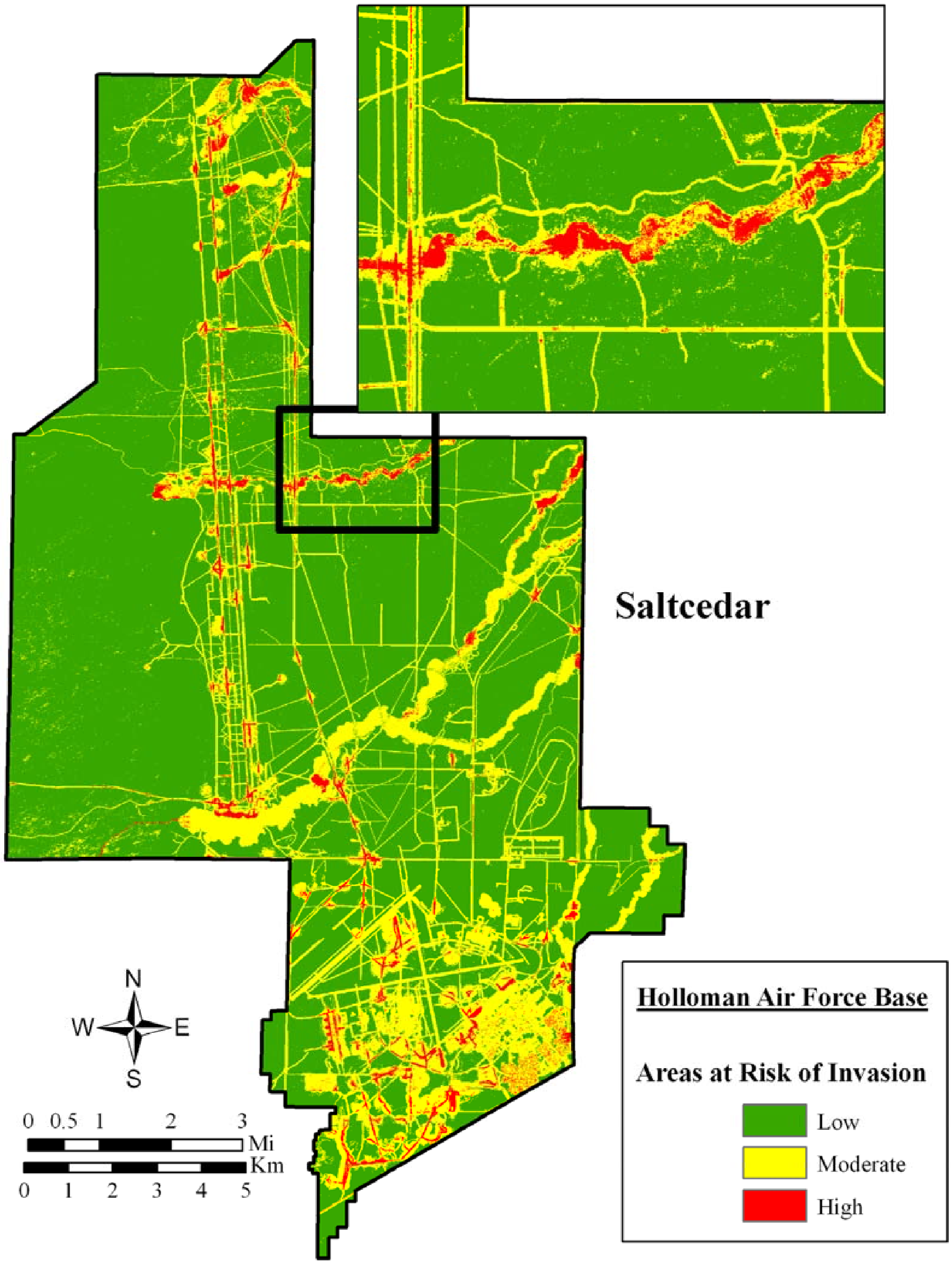

Figure 20. Areas at risk of saltcedar incursion or spread on Holloman Air Force Base. The likelihood of saltcedar incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

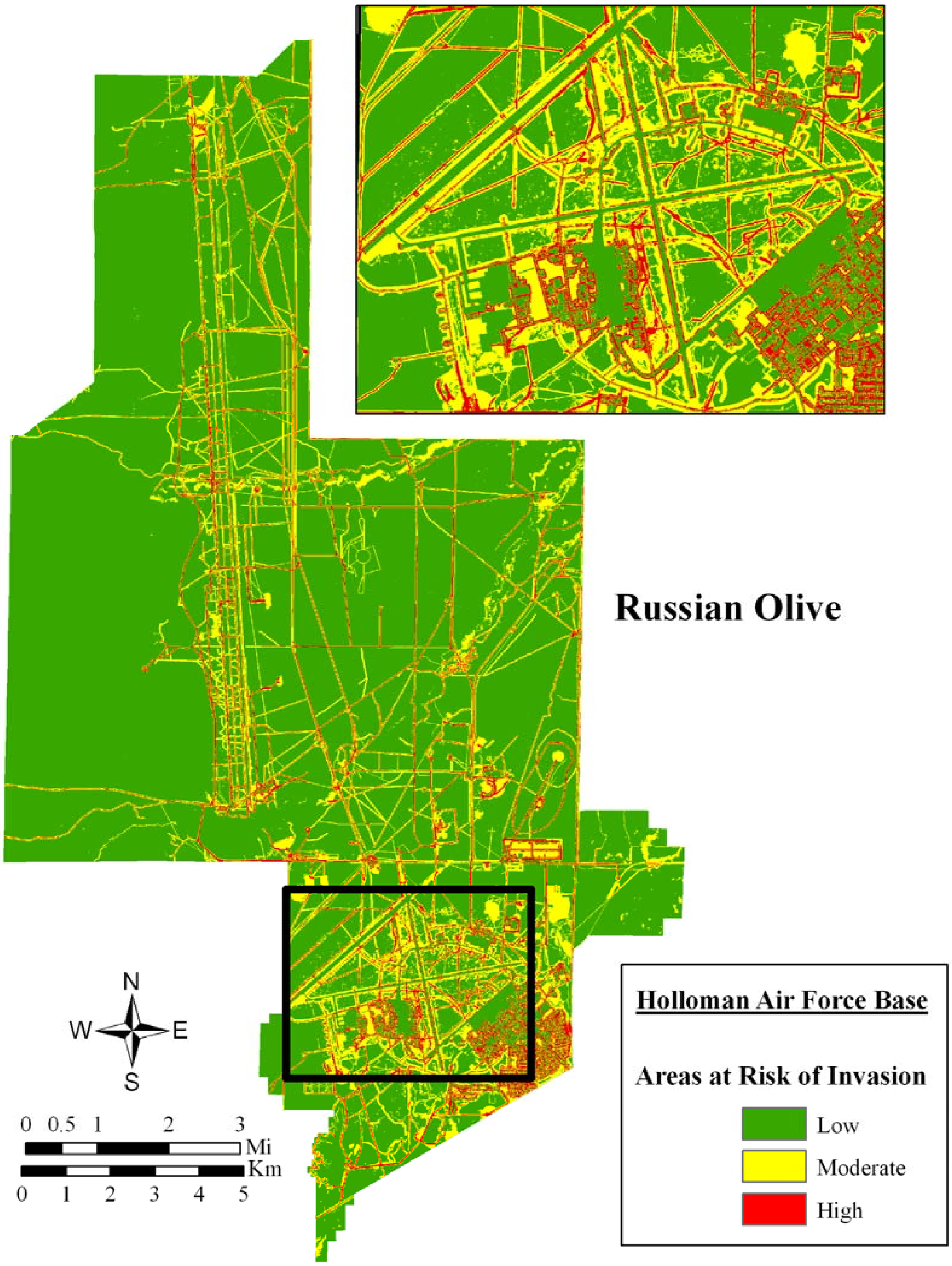

Figure 21. Areas at risk of Russian olive incursion or spread on Holloman Air Force Base. The likelihood of Russian olive incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

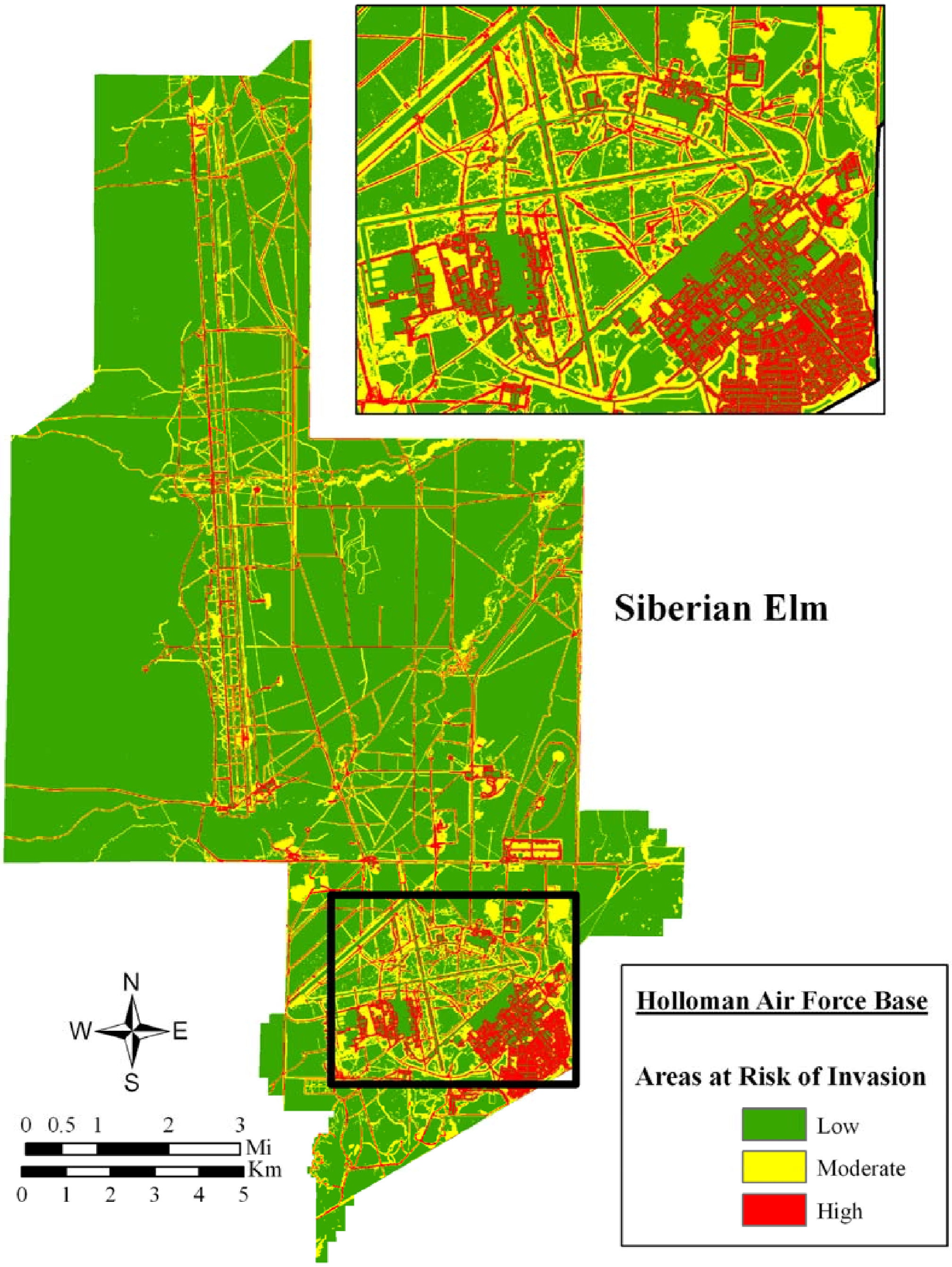

Figure 22. Areas at risk of Siberian elm incursion or spread on Holloman Air Force Base. The likelihood of Siberian elm incursion was estimated from spatial analyses of current plant locations, predicted suitable habitats, vectors and pathways, and disturbances.

Table 22. Amount of area at risk to invasion by target invasive plants on Holloman Air Force Base.

	Amount of Area at Risk to Plant Invasion								
	High Risk			Moderate Risk			Low Risk		
Plant	**Hectares**	**Acres**	**Percent**	**Hectares**	**Acres**	**Percent**	**Hectares**	**Acres**	**Percent**
Russian knapweed	1,310	3,237	6.2	2,345	5,794	11.0	17,614	43,524	82.8
Five-horn smotherweed	498	1,229	2.3	4,242	10,482	19.9	16,529	40,843	77.7
Malta starthistle	434	1,074	2.0	2,711	6,699	12.7	18,123	44,782	85.2
Russian olive	949	2,345	4.5	2,851	7,045	13.4	17,468	43,165	82.1
African rue	879	2,173	4.1	3,683	9,101	17.3	16,706	41,281	78.5
Russian thistle	878	2,170	4.1	4,037	9,976	19.0	16,353	40,408	76.9
Saltcedar	474	1,172	2.2	4,472	11,051	21.0	16,322	40,332	76.7
Siberian elm	965	2,384	4.5	2,879	7,115	13.5	17,424	43,056	81.9
All Target Plants	628	1,552	3.0	3,125	7,723	14.7	17,515	43,280	82.4

pose to be the next invasive plant issue on HAFB. Malta starthistle is a prolific seed producer and can rapidly displace native vegetation and create a monoculture (DiTomaso and Gerlach 2000, Callaway et al. 2003).

African rue is currently well established on HAFB. Large populations of African rue have been found along all road right-of-ways, the High Speed Test Track, Space Command testing area, 46 Test Group office areas, and many portions of the flightline infield on HAFB (Reiser et al. 2000). African rue colonization began in disturbed areas on HAFB, but it rapidly spread into the native grasslands, shrublands, and roadways within the dune area, and has been found in the Lake Holloman Wildlife Refuge Area (Reiser et al. 2000). This plant is a prolific seed producer, and spreads proximate to the parent plant. Tilling, mowing, and blading along roadsides have likely increased its spread (Parker and Reiser 1997). Approximately 22% (4,562 ha; 11,274 ac) were modeled at high and moderate risk of African rue incursion (Table 22). High risk areas included roadsides along all the primary roads, open parking lots, and other disturbed areas (Fig. 18). Moderate risk areas radiate out from high risk areas, and include infrequently used dirt roads, and areas around air support facilities, and native vegetation communities in close proximity to disturbed areas. The spread of African rue into native communities will likely result in diminished biodiversity as native plant communities are displaced. The effect of African rue incursion on vertebrate communities has not been investigated, and is poorly understood.

Russian thistle can be found in a wide variety of habitat types on HAFB, including disturbed areas, fields, roadsides, grasslands and desert communities. Our analyses indicated that approximately 23% (4,915 ha; 12,145 ac) of the base is at high and moderate risk of incursion by Russian thistle (Table 22). The modeled risk distribution of moderate and high areas indicated that only the dune portion of the base was not at risk by this plant (Fig. 19). Moderate at risk patches were modeled throughout other native vegetation communities, although large patches of areas at risk were primarily around open or disturbed areas/lots, residential and commercial areas, roadsides, and air support facilities (Fig. 19). The interior airfield area, roadsides, and open areas were modeled at moderate risk of Russian thistle occurrence, and thus could influence HAFB missions if plants become established in these areas, especially when mature plants break away from their root system and move across the runways during wind events. Due to the mode of seed dispersal and the variety of areas suitable for establishment, control of Russian thistle on HAFB may be difficult and costly.

Saltcedar on HAFB was originally planted as a windbreak above Carter Draw and other former ranch sites, and was planted by HAFB near the test track for dune stabilization over 30 years ago (Reiser et al. 2000). Saltcedar has since moved aggressively into drainage bottoms and lowland depressions at the southern end of the base, and along drainage bottoms and isolated depressions across most of HAFB. Our risk assessment indicate that approximately 23% (4,946 ha; 12,223 ac) are at moderate and high risk of saltcedar incursion (Table 22). Areas modeled at high risk of saltcedar occurrence were also widespread throughout the base. Arroyos, depressions, and wet areas were considered at high risk while disturbed areas, residential and commercial areas, roadsides, and air support facilities were modeled at moderate risk of saltcedar occurrence (Fig. 20). Although HAFB has greatly reduced the amount of saltcedar, the large seed bank and availability of suitable habitats will require continued monitoring and management on the base.

Russian olive and Siberian elm have similar distributions and predicted risk distributions on HAFB. Our risk assessments indicate that approximately 18% (~3,800 ha; 9,390 ac) are at moderate and high risk of Russian olive or Siberian elm incursion (Table 22). These plants were primarily introduced for horticultural purposes. Thus, their current and predicted risk distributions are closely tied to residential and commercial areas (Figs. 21, 22). However, these plants may occur in other disturbed areas as propagules are spread by human activities.

We averaged all eight plant risk models to estimate key areas at risk to multiple invasive plant species incursion. Approximately 18% (3,753 ha; 9,274 ac) are at moderate and high risk of incursion by multiple species (Table 22). Key areas of concern include the northern and southern end of the Test Track, Reagan Draw, Hay Draw, Lost River, Malone Draw, Ritas Draw, Dillard Draw, and areas surrounding the airfield in the southern part of HAFB (Fig. 23). Invasive plant incursion into these areas could impact native biodiversity and influence the mission needs of HAFB origanizations and units.

We calculated the proportion of communities represented by the grouped land cover classes at moderate and high risk of invasive plant incursion or spread. Our results indicate that xeric riparian habitats may have approximately 5-times greater risk to invasive plant incursion than upland habitat types on HAFB (Table 23). Approximately 48% of riparian habitats and 20% of wetland habitats are at moderate to high risk of invasive plant incursion or spread. These habitats have already been heavily impacted by saltcedar (Reiser et al. 2000), and host favorable conditions for other species of concern. Wetland habitats in the southern part of the HAFB are also popular recreational areas that concentrate human activity and disturbance. Approximately 9% of the grassland and fourwing saltbush communities on HAFB are at moderate to high risk of invasive plant incursion or spread, while < 1% of dune and creosote bush communities are at risk (Table 23).

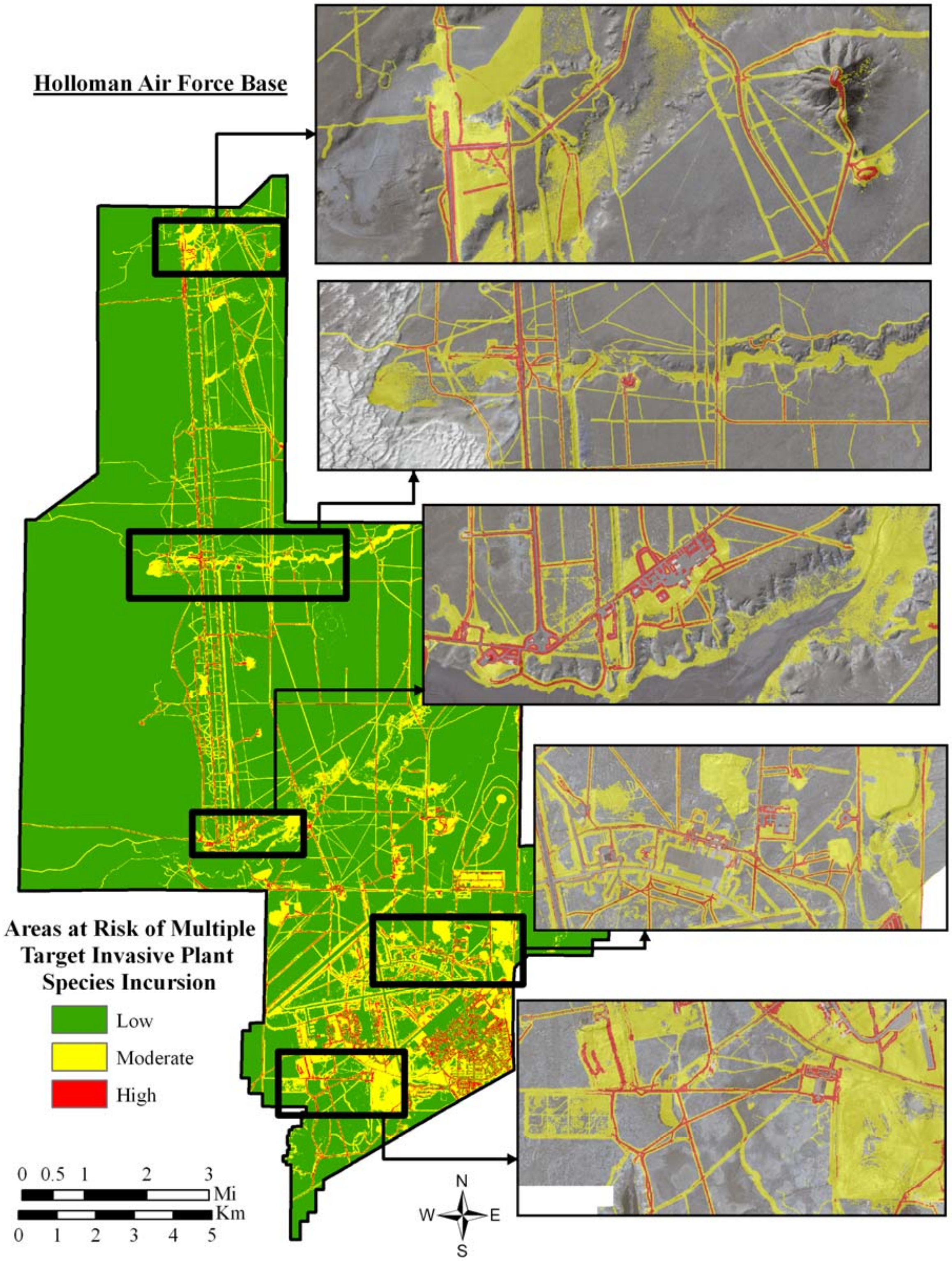

Figure 23. Areas at risk of multiple invasive plant species incursion or spread on Holloman Air Force Base.

Table 23. The amount of each grouped land cover class at moderate to high risk of invasive plant incursion or spread on Holloman Air Force Base.

	Approximate Amount of Land Cover Class on HAFB			Amount of Area at Moderate to High Risk to Plant Invasion		
Grouped Land Cover Class	**Hectares**	**Acres**	**Percent**	**Hectares**	**Acres**	**Percent**
Xeric Riparian	918	2,267	4.3	436	1,077	47.5
Wetland/Playa	302	745	1.4	61.164	151	20.3
Grassland	5804	14,341	27.3	495.4897	1,224	8.5
Fourwing Saltbush Complex	7973	19,701	37.5	751.2992	1,856	9.4
Dune	3476	8,590	16.3	27.3715	68	0.8
Creosote Bush Shrubland	147	364	0.7	10.2521	25	7.0

The buffer distances provided to risk variables were 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 developed area. We also considered the scale of our datasets. For example, our orthorectified images were derived from ground control points from a 15 cm (6 in) digital aerial photo. Root mean square error for our orthorectified images was ≤ 0.58 m (1.89 ft) for the multispectral images and ≤ 0.34 m (1.12 ft) for the panchromatic images. We also derived our spatial locations of plants by matching field observations and GPS tracking with the same 15 cm (6 in) aerial photos. The spatial accuracy of our datasets was very precise, and not typical for similar studies. As such, our buffer distances should have been consistent with our data and should have encompassed the area influenced by each model variable. Nevertheless, buffer distances provided to risk variables should be evaluated and updated as information becomes available. Likewise, buffer distances may not be appropriate for other study areas or species.

Boles Wells Water System Annex

As previously noted, there was insufficient data to conduct risk assessments for the BWWSA. Muldavin et al. (2006) noted five locations that hosted African rue and one location that hosted saltcedar in the southern portion of the BWWSA. These plants were typically found in highly disturbed sites such as the wellheads and along roads (Muldavin et al. 2006). This survey did not include the Boles Wells Well Field, located at the north end of the BWWSA.

During our 2007 surveys, we located Russian thistle, African rue, Russian knapweed, saltcedar, and Siberian elm. Most plants were in disturbed areas in the Boles Wells Well Field area (Fig. 24). As such, areas around the Boles Wells Well Field were at high risk of additional invasive plant incursion. Roadsides and disturbed areas should be monitored for plant establishment and population expansion.

These invasive plant species are spread by soil disturbances in close proximity to existing populations. Therefore, a strong indicator of future invasion in the BWWSA may be determined by monitoring soil disturbance or areas that are periodically disturbed.

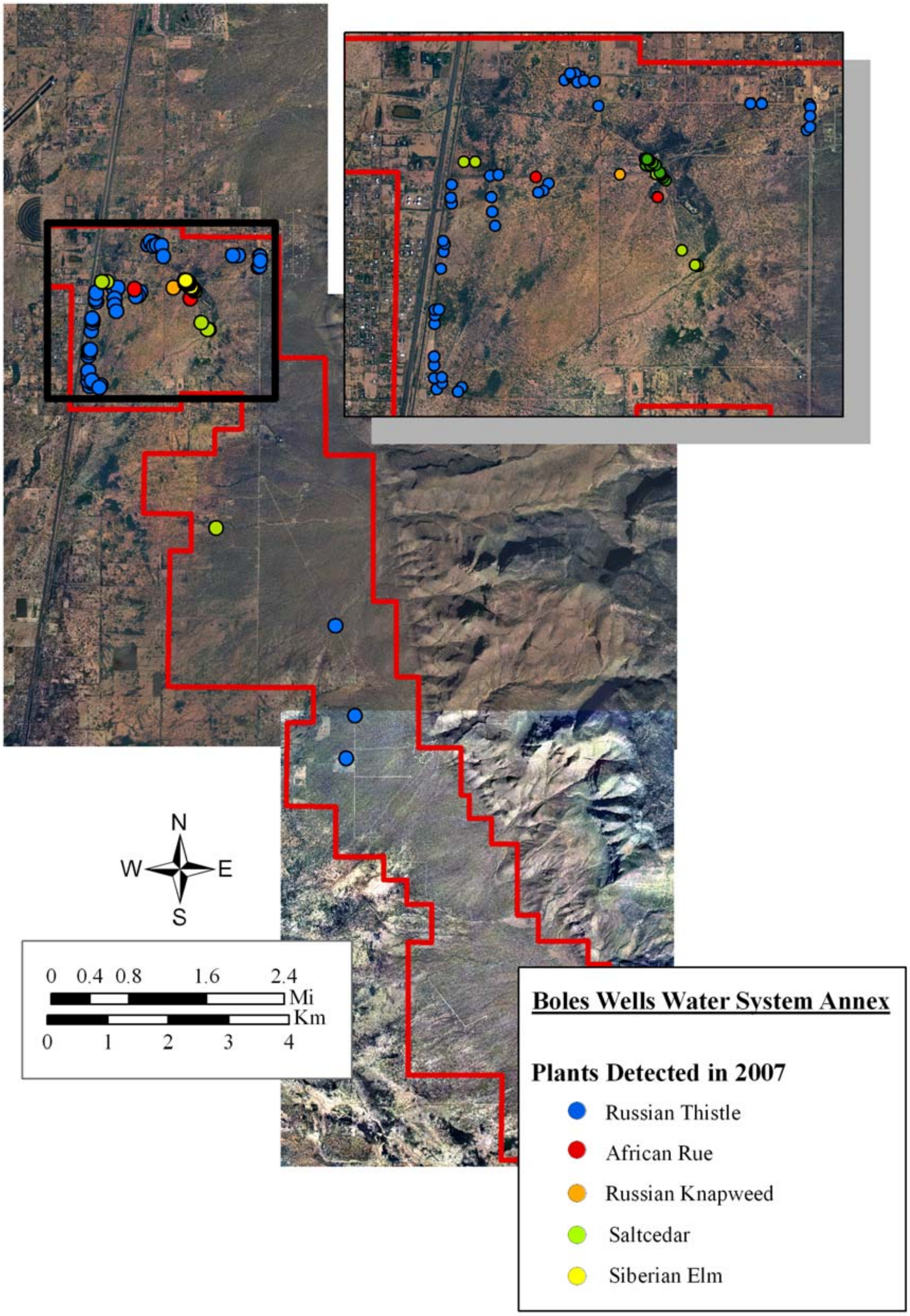

Figure 24. Target invasive plants detected in 2007 in the Boles Wells Water System Annex, Holloman Air Force Base.

CONSERVATION IMPLICATIONS

The Use of Remotely Sensed Data

Successful invasive plant management strategies rely on the early detection of species, effective plant control efforts, and assessment and monitoring of plants and landscapes (Johnson 1999, Smith et al. 1999, Rejmanek 2000, Masters and Sheley 2001). Successful eradication or control efforts are dependent on the detection of new populations (early detection), or in terms of control, the detection of an existing population beginning to spread (Rejmanek and Pitcairn 2002). Assessment and monitoring of plants and landscapes allows managers to develop scientifically defensible predictions of the status and trends of invasive species incursion or spread, to evaluate management practices, and to inform necessary modification. Successful monitoring programs provide the foundation for effective natural resource management and conservation. Monitoring establishes a method for evaluating the success of meeting desired management and conservation outcomes, detecting shifts in distribution or changes in habitat, and documenting regulatory compliance.

Remote sensing tools have the potential to provide information for early detection of invasive plants and for assessing and monitoring plants and landscapes. Remotely sensed images have a number of features which make them ideal for predicting areas at risk for invasive plant species incursion across large landscapes. Remote sensors can collect data beyond the visible electromagnetic spectrum, which allows better detection of vegetation. Remote sensors also allow for regional or landscape analyses that would be cost-prohibitive using ground-based visits. Landscape 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). Integrating remotely sensed data with other spatial datasets enhances our abilities to model invasive plants. However, 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 (sensors that sample the electromagnetic spectrum in narrow continuous increments) and high spatial resolution sensors (e.g., spatial resolutions < 5 m; 16.4 ft). While hyperspectral data facilitate detection of individual plants, hyperspectral data have approximately 75 times greater data volume than an equivalent area using Landsat ETM+ (Thenkabail et al. 2004).

Our research indicates that multispectral, high spatial resolution sensors (e.g., Quickbird imagery), show promise in detecting invasive plants, although these sensors are 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.

Plant Invasion Stage

Our research focused on modeling the potential distribution of suitable habitats for eight invasive plant species that were in various stages of plant invasion (Hobbs and Humphries 1995, Williamson 1996, Masters and Sheley 2001) (Fig. 25). From a plant eradication or control standpoint, early detection efforts are best directed at invasive plants that are in the introduction to establishment plant invasion phase, e.g., Russian knapweed, Malta starthistle, Russian olive, Siberian elm, and five-horn smotherweed on HAFB. Unfortunately, these early invasion phases may also yield the greatest challenges for conservation modeling. We did not locate Russian knapweed during our 2007 surveys on the main base, and our surveys resulted in ≤ 12 occurrence locations for Russian olive and Siberian elm. Thus, we used a deductive model approach to estimate potential habitat areas for these three plant species. Likewise, we had limited occurrence locations (n=25) of Malta starthistle, which greatly limited the number of test locations used for model performance. Greater research is needed to determine the effects of low sample size on model performance and to evaluate the efficacies of using deductive models for plants that are in the early invasion stage. Nevertheless, our research indicates that deductive habitat models are still useful for providing insight into potential habitat areas, especially when combined with other risk data layers.

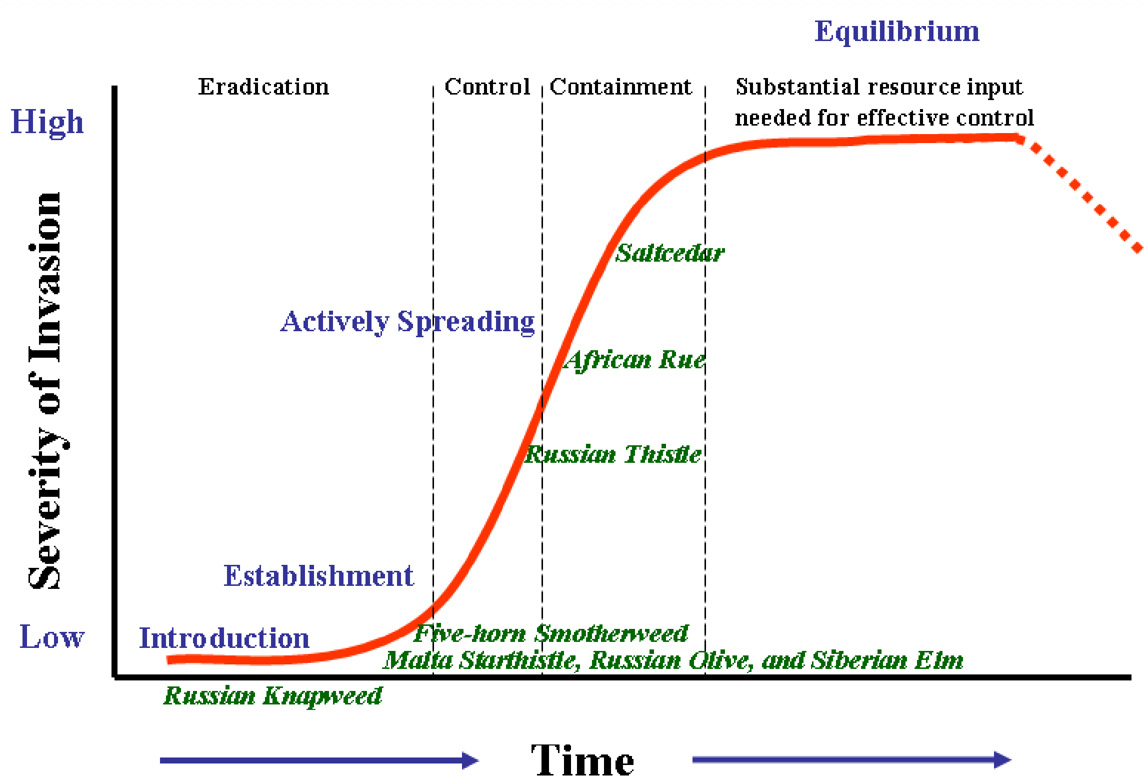

Figure 25. Graph of plant invasion stages redrawn from Hobbs and Humphries (1995), Williamson (1996), and Masters and Sheley (2001) for our target invasive plants on Holloman Air Force Base.

Although five-horn smotherweed is not listed on New Mexico's noxious weed list, Russian knapweed and Malta starthistle are considered Class B noxious weeds in the State. Control efforts for these species across the state are directed towards preventing the spread of these species into new areas (Office of the Secretary 2009). However, eradication of Russian knapweed and Malta starthistle on HAFB may be possible with immediate control efforts directed on known populations followed by an active monitoring program in at risk areas. Russian olive and Siberian elm are considered widespread in the state (Class C noxious weed), but are relatively contained to the southern part of the base and they do not appear to be actively spreading. Nevertheless, to avoid future incursion into areas at risk, consideration should be given to eradication efforts of known Russian olive and Siberian elm trees.

Although Russian thistle is not listed on New Mexico's noxious weed list, it is actively spreading on HAFB (Fig. 25). Control efforts for this plant may be difficult given the habitat generalist nature and the specialized seed dispersal mechanism of this plant. Likewise, African rue is actively spreading on HAFB, and considered a Class B noxious plant in New Mexico (Office of the Secretary 2009). Eradication of these species is also unlikely. Containment (limiting the spread of invasive plants to existing populations) of plant populations should be the management priority.

Saltcedar is also considered a Class C noxious plant in New Mexico, as it is widespread across the state. Saltcedar has aggressively spread into depressions across the entire base. HAFB exerted substantial control efforts on the base in 2007, which greatly reduced the amount of saltcedar on the base. However, a considerable amount of areas are still occupied by saltcedar, on and adjacent to the base. The eradicated and current populations have produced a substantial seed bank in and within close proximity to suitable areas. As such continued saltcedar monitoring and management on the base is warranted.

Presence-only Predictive Habitat Models

Often, land managers record spatial data on the presence of invasive plants within their management area. However, few land managers have the budget or time to collect spatial data on areas where invasive plants are absent. As such, reliable modeling approaches using presence-only datasets can be a powerful tool for land managers. We used a maximum entropy (program Maxent) approach that statistically derived a probability distribution based on presence-only data for our modeling endeavors. This approach estimates the broadest target probability distribution which should have minimized errors of omission. From a 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. Program Maxent is a user friendly free-ware program, although documentation on the interpretation of the program's output is somewhat limited. 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 verses GARP outputs, and concluded that Maxent models outperformed GARP models.

Reliable invasive plant 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. Our invasive plant predictive models were created using Quickbird spectral information derived from georeferenced plant locations on HAFB. This approach would include species distribution data, landscape structures and statistical patterns. Based on model performance, our proof-of-concept approach worked well for all species. Nevertheless, modeling approaches that employ presence/absence analyses may further help differentiate invasive communities from native communities, which may improve model performance.

Early Detection Risk Models

Knowledge of invasive plant distributions, invasion dynamics, environmental characteristics, and weed dispersal processes is important for developing invasion risk assessment models (Masters and Sheley 2001). Our research has demonstrated that remotely sensed and GIS spatial datasets that incorporate these key elements can aid in the development of useful invasion risk models. These risk models provide land managers with early detection tools, a means to evaluate current and future control needs, and a means to prioritize conservation efforts.

Our habitat models and risk surfaces allow land managers to prioritize species for monitoring. For example, our results indicated that African rue and Russian thistle have considerably more potential habitat than the other species analyzed (Fig. 26). However, when areas at risk are considered, then saltcedar and five-horn smotherweed are also important plants to monitor, as they could impact a large amount of the landscape.

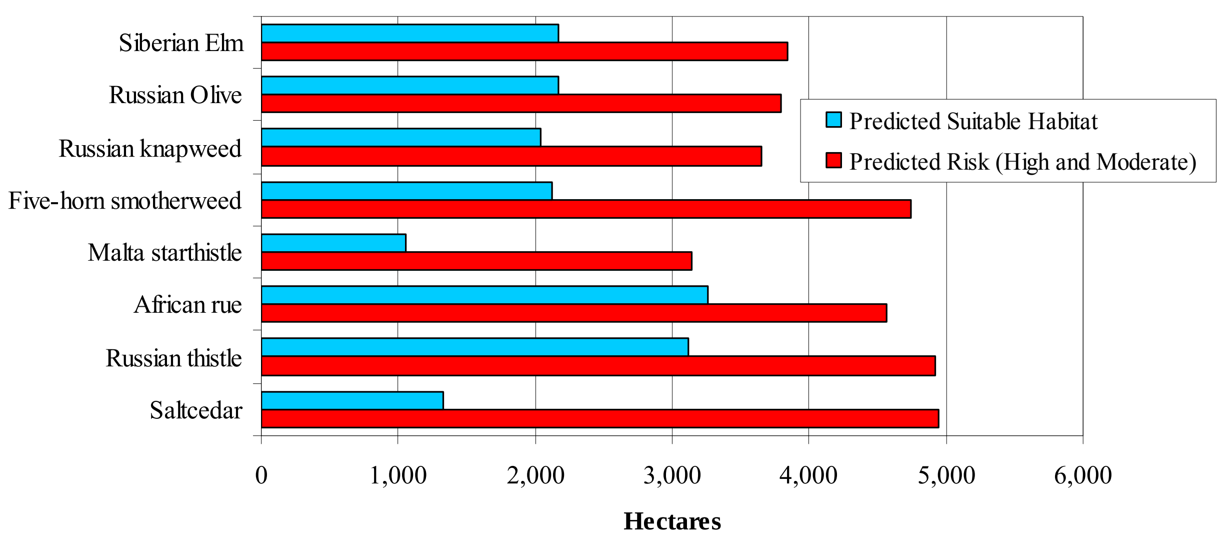

Figure 26. Amount of predicted suitable habitat and area at risk for invasive plants on Holloman Air Force Base.

Our risk surfaces also allow land managers to prioritize landscape units to monitor. For example, approximately 9% of fourwing saltbush and grassland habitats, 20% of wetland habitats, and 48% of xeric riparian habitats are at moderate to high risk of invasive plant incursion or spread (Fig. 27). Riparian, wetland, and grassland habitats on HAFB support diverse plant and vertebrate community (Reiser et al. 2000), which may be threatened by invasive plant incursion. Monitoring activity directed at moderate to high risk areas could assist in sustaining native biodiversity.

Management considerations such as threatened or endangered species or mission readiness could also be used to further prioritize species and landscape units. For example, the White Sands pupfish (*Cyprinodon tularosa)* is listed as a threatened species by the State of New Mexico (New Mexico Department of Game and Fish 2006). Pupfish habitat includes shallow, alkaline springs and streams. Pupfish are currently present on HAFB in Lost River (Mehlhop et al. 1998), but important habitat for the pupfish includes all stream channels of Malone Draw and Lost River on and a corridor 100 m (330 ft) buffer on both sides of the stream channel (Reiser et al. 2000). Large saltcedar populations in this buffer area may influence water quantity and quality, and negatively affect pupfish populations. As such, monitoring efforts for saltcedar incursion could be prioritized in moderate to high risk areas near Malone Draw and Lost Creek (Fig. 28).

The use of appropriate buffers around model variables ensures that inferences represent locations of influential features within a reasonable degree of data precision. However, determining an appropriate influence distance for each model variable without quantified data is difficult. As with most studies, the actual distance that a variable influenced the presence/absence of our target plants was unknown, and designing a study to quantify this area was beyond the scope of our project. Therefore, we created buffer distances that seemed logical for our plant species in our study area.

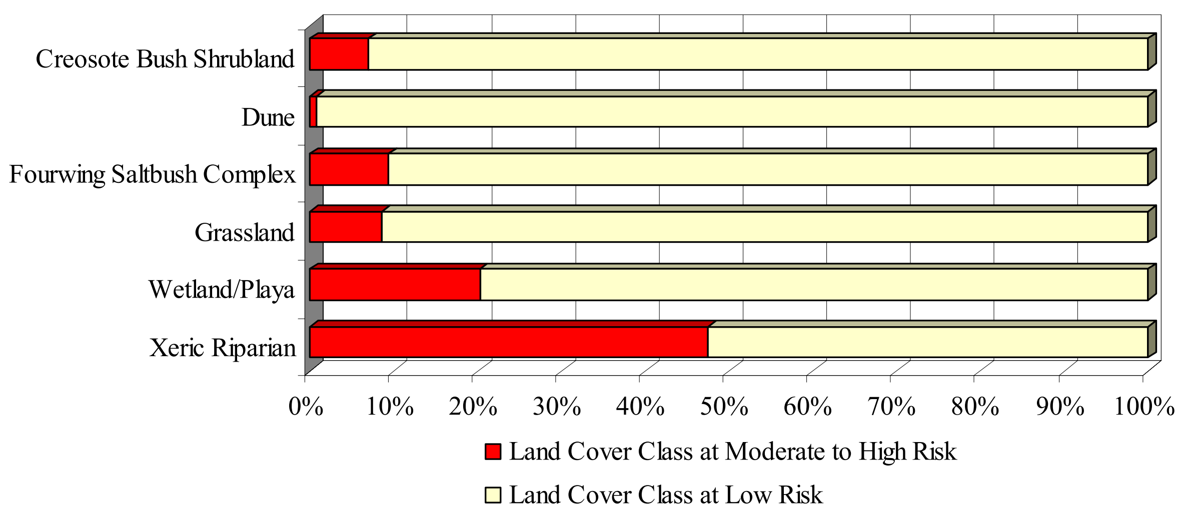

Figure 27. Percent of each habitat type on Holloman Air Force Base at moderate to high risk of invasive plant incursion or spread.

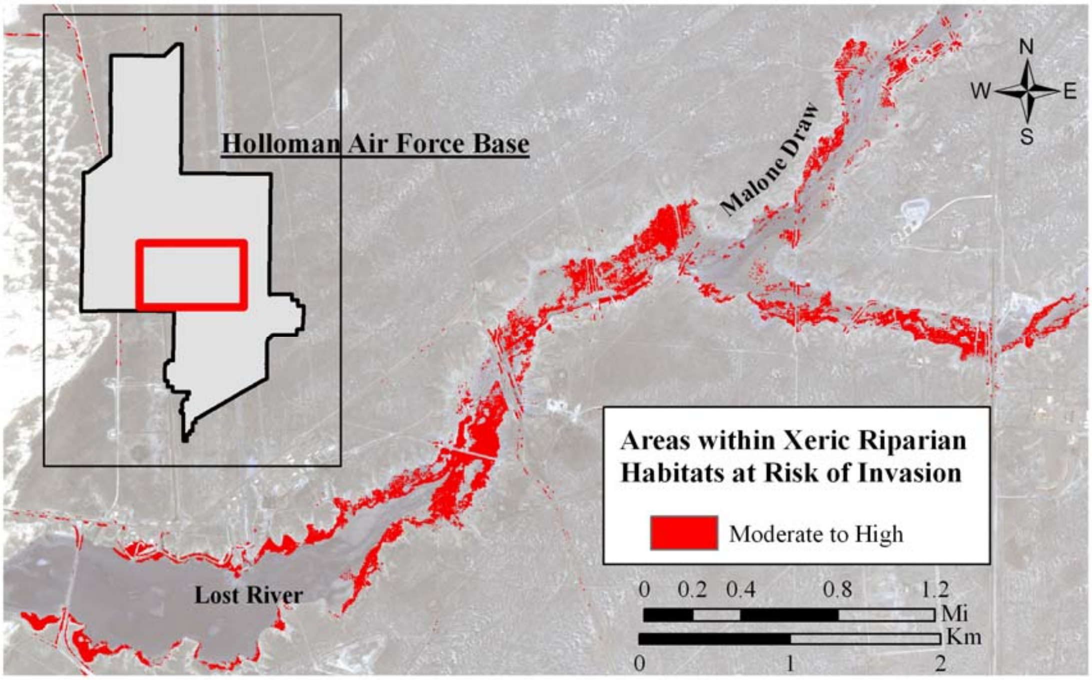

Figure 28. Candidate monitoring areas within xeric riparian habitats that host moderate to high risk of invasive plant incursion. Invasive plant incurion in these areas may influence water quantity and quality, and negatively affect White Sands pupfish populations on Holloman Air Force Base.

Efficacy of Early Detection Models

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 non-native 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 represent an effective approach to assess landscapes for risk to incursion of invasive plants. These models can inform 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.

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