2007

Risk assessment for invasive exotic plants using multi-criteria risk models

Lindsey Marie Shartell
Michigan Technological University

Follow this and additional works at: https://digitalcommons.mtu.edu/etds
Part of the Forest Sciences Commons

RISK ASSESSMENT FOR INVASIVE EXOTIC PLANTS

USING MULTI-CRITERIA RISK MODELS

By

LINDSEY MARIE SHARTELL

A THESIS

Submitted in partial fulfillment of the requirements

for the degree of

MASTER OF SCIENCE IN FOREST ECOLOGY AND MANAGEMENT

MICHIGAN TECHNOLOGICAL UNIVERSITY

2007

This thesis, Risk Assessment for Invasive Exotic Plants using Multi-Criteria Risk Models, is hereby approved in partial fulfillment of the requirements for the Degree of MASTER OF SCIENCE IN FOREST ECOLOGY AND MANGEMENT.

School of Forest Resources and Environmental Science

Signatures

Thesis Advisor____
Linda M. Nagel

School Dean____
Margaret R. Gale

Date___________________________________

Table of Contents

Chapter 3. A Multi-Criteria Risk Model for Garlic Mustard (*Alliaria petiolata*) Invasion Across the Upper Peninsula of Michigan

Acknowledgements

This research could not have been completed without the help of numerous individuals. Thank you to Linda Nagel for giving me the opportunity to work on these projects and for advising on all aspects of grad school, research, and writing. Thank you to my committee members Andrew Storer, Chris Webster, and Barry Solomon. And thank you to Mike Hyslop for his interest in these projects and his support with GIS, GPS, and computing.

Thank you to everyone who assisted in the field, especially my main crew, Jess Barden and Michelle Freeman, but not forgetting Kurt Doran, Kasey Cornwell, and Jennie Lund. Thank you to the National Park Service and U. S. Forest Service for funding and assistance with field work. And thank you to Greg Corace, reviewing and providing helpful comments on my thesis drafts.

Thank you to Michigan Tech and the School of Forest Resources and Environmental Science for providing a top-notch place to learn, and for keeping me funded. And last but not least, thank you to my family and friends for continued their support and encouragement.

Thesis Abstract

Invasive exotic plants have altered natural ecosystems across much of North America. In the Midwest, the presence of invasive plants is increasing rapidly, causing changes in ecosystem patterns and processes. Early detection has become a key component in invasive plant management and in the detection of ecosystem change. Risk assessment through predictive modeling has been a useful resource for monitoring and assisting with treatment decisions for invasive plants.

Predictive models were developed to assist with early detection of ten target invasive plants in the Great Lakes Network of the National Park Service and for garlic mustard throughout the Upper Peninsula of Michigan. These multi-criteria risk models utilize geographic information system (GIS) data to predict the areas at highest risk for three phases of invasion: introduction, establishment, and spread. An accuracy assessment of the models for the ten target plants in the Great Lakes Network showed an average overall accuracy of 86.3%. The model developed for garlic mustard in the Upper Peninsula resulted in an accuracy of 99.0%. Used as one of many resources, the risk maps created from the model outputs will assist with the detection of ecosystem change, the monitoring of plant invasions, and the management of invasive plants through prioritized control efforts.

Chapter One

Thesis Introduction

Invasive plants pose a significant threat to natural ecosystems throughout much of North America because of their ability to alter natural ecosystem patterns and processes. Nearly all introductions of invasive plants are associated with humans (Baker 1984). Westbrooks (1998) blamed population growth, which leads to increased disturbance, overuse of the land, and increased international travel, all of which facilitate the movement and success of invasive plants. Coblentz (1990) put forward the three most threatening human-induced problems: inappropriate resource use, pollution, and the introduction of exotic organisms. Of these, he suggested that the introduction of exotic organisms is the most difficult to correct (Coblentz 1990). Invasive species are thought to be one of the most critical threats to biodiversity (Pimm *et al.* 1995). Therefore the areas of greatest concern are natural ecosystems where much of the biodiversity remains intact. These areas often represent unique habitats, and contain many threatened and endangered species. As land is developed for human use and natural areas become rarer, the land set aside for protection and management such as National Wildlife Refuges, National Parks, National Forests, State Forests, and other conservation lands will be of great importance.

Not all exotic plants that disperse from their native habitat to a new location become invasive. Those that do become invasive tend to spread rapidly once established. This rapid rate of spread is thought to be possible due to a release from native competitors, predators, pathogens, and diseases. This hypothesis, referred to as the ‘enemy release hypothesis’, is based on the idea that no natural enemies exist in the invaded range to limit their reproduction and spread as they do in the native range (Keane and Crawley 2002). While this may explain in part why species become invasive, there are a number of other factors that contribute to the ability of an exotic plant to become invasive. Callaway and Ridenour (2004) suggest the ‘novel weapons hypothesis’. This hypothesis suggests that the success of invasive plants relies on the possession of biochemicals that are used to gain increased competitive ability (Callaway and Ridenour 2004). Another explanation for the success of invasive plants is their ability to take advantage of disturbed areas, whether natural or anthropogenic (Underwood *et al.* 2004). It is also thought that invasive plants possess common life history traits that favor population growth. Some of these characteristics include early maturation, abundant seed production, long life of seeds in the soil, adaptation for spread, and production of toxins that suppress the growth of other plants (Van Driesche 2002). The possibility of using life history traits to identify which species will become invasive has been tested using various combinations of traits (Rejmanek and Richardson 1996, Sutherland 2004). Rejmanek and Richardson (1996) tested 10 life history traits and found that invasiveness in woody plants could be predicted by seed mass, juvenile period, and seed crop interval. However, other studies have not shown such clear relationships. Sutherland (2004) found that lifespan, life form, habitat, being armed and toxicity to other plants were common characteristics among weeds, but was unable to use these characteristics to differentiate between non-invasive and invasive plants. These results were similar to those of Goodwin *et al*. (1999), who were not able to predict invasiveness from the biological characteristics of life form, stem height, and flowering period. Furthermore, this does not address the crucial issue of managing plants that are invasive and are already established and spreading.

The primary dilemma with invasive plants is their ability to out-compete native plants for growing space and other resources, thus reducing biodiversity (Coblentz 1990, Pimm *et al*. 1995). Of increasing concern, however, is their capability to cause major changes in ecosystem and landscape structure. In addition to reducing biodiversity, invasive plants can alter resource availability, disturbance regimes, water flow, and other natural processes. They can have direct effects on all components of an ecosystem. Invasive plants in Florida were found to alter geomorphology, hydrology, biogeochemistry, and disturbance patterns (Gordon 1998). A study of 56 invasive plants showed effects on soil nutrient cycling such as increased biomass, increased nitrogen availability, and altered nitrogen fixation rates (Ehrenfeld 2003). Invasive plants also stress wildlife by altering food sources, nesting sites, and cover habitat, or by attracting other non-native species (Graham 2002). Of most concern to humans, however, is the economic loss associated with managing and controlling invasive plants. It is estimated that invasive species cause losses of $138 billion each year (Pimentel *et al.* 2005). Furthermore, the presence and management of invasive plants can impinge on recreational activities and the aesthetics of natural areas.

Rejmanek (2000) suggested that three management approaches for managing invasive plants exist: 1) prevention or exclusion, 2) early detection or rapid assessment, and 3) control, containment, or eradication. Too often invasive plant management is reactive; it is only of interest once a problem has established (Peterson and Vieglais 2001). This necessitates a control, suppression, or eradication approach. Although control is imperative once a species is established, preventing establishment would be more effective. However, as noted earlier, prevention or exclusion may be unfeasible, therefore early detection and rapid assessment is the most promising approach to locating initial introductions of invasive plants for feasible control treatments. An increasing interest in the use of predictive modeling to monitor invasions makes this approach practical. These models can be used to predict spread from known invasions, to determine the possible distribution of a species, and to predict the probability of invasion.

Previous predictive modeling has focused on spatial models of spread and ecological niche modeling. Modeling of invasive plant spread is motivated not only by the negative effects of these plants, but also by the unique opportunity to watch a species expand through a new ecosystem (Higgins and Richardson 1996). For this reason, numerous models of spread have been created for invasions of insects, plants, animals, and diseases. Higgins and Richardson (1996) designated three types of models of spread, with the most useful being spatial-mechanistic, which is a combination of the other two types, simple-demographic and spatial-phenomenological. Spatial-mechanistic models are based on ecological parameters and characteristics of the initial invasion, and can predict both population density and area invaded over time (Higgins and Richardson 1996). Predictive models such as reaction-diffusion (RD) models and integrodifference equation (IDE) models are examples of spatial-mechanistic models. RD and IDE models both assume that the landscape is spatially homogenous and from there, predict population growth and dispersal over time (With 2002). Models of spread depend on a known invasion from which the plant will spread. Not only are these models strictly concerned with spread from established populations, they do not consider long-distance dispersal (movement over 100 m). Long-distance dispersal of seeds can be difficult to measure and is not entirely understood, however, it is a critical factor in the introduction of invasive plants (Shigesada *et al.* 1995, Higgins and Richardson 1999, Cain *et al.* 2000). Additionally, since the results of these models predict future invasion based on current conditions, they may be unrealistic in a changing landscape or a changing climate (With 2002).

Determining the possible distribution of an invasive plant relies heavily on ecological niche modeling. Ecological niche modeling uses data on environmental characteristics of a species' native range and/or of areas currently invaded to determine similar suitable habitats (Peterson and Vieglais 2001). These environmental characteristics can simply be related to species presence using logistic regression, however, more recently, improved techniques have been developed and utilized. One popular method is the Genetic Algorithm for Rule-set Production (GARP, Stockwell and Peters 1999). GARP inputs species location records and environmental data, analyzes this information with a machine-learning-based analytical program, and from this, determines rules for presence (Stockwell and Peters 1999). These rules are compared to points sampled randomly from a set study area, and a potential distribution map is created (Stockwell and Peters 1999). The GARP method has been shown to accurately predict the potential distribution of invasive plants (Peterson *et al.* 2003, Underwood *et al.* 2004). Peterson *et al.* (2003) predicted the invasive potential of four plants, garlic mustard (*Alliaria petiolata*), sericea lespedeza (*Lespedeza cuneata*), Russian olive (*Elaeagnus angustifolia*), and hydrilla (*Hydrilla verticillata*), using the GARP method and found the predicted area to be highly coincident with the areas of known invasions. Underwood *et* *al*. (2004) used the GARP method to obtain a 76% correct prediction of the areas at highest risk for invasion by non-native plants within Yosemite National Park in California. Unfortunately, many invasive plants have a large potential distribution and, within small study areas, are not limited by the environmental characteristics used. The GARP method predicts areas suitable for species survival and can be used to calculate probability of invasion, but it does not include information on dispersal, disturbances, or other stochastic events, all of which can greatly affect the suitability of a site to invasion (Underwood *et al.* 2004).

A method similar to ecological niche modeling is invasion risk assessment. This method utilizes environmental characteristics of invaded areas to predict areas at risk for future invasion. One method, recursive modeling, creates a tree diagram that relates predictors with each other and with the dependent variable. Using recursive modeling to determine important environmental characteristics is favorable since environmental characteristics are often related. Formal Inference-based Recursive Modeling (FIRM) was used for risk assessment of invasive *Pinus* species in South Africa (Rouget *et al.* 2004). FIRM is advantageous because it can handle larger datasets, can utilize both categorical and continuous predictors, and can uncover complex relationships between predictors (Hawkins 1999). There are, however, some limitations to using recursive modeling for invasion risk assessment. The method requires ample data on current invasion locations and environmental characteristics for both invaded (presence) and non-invaded (absence) locations. Data limited to presence only may not identify the environmental characteristics that are important and may only represent habitats with a high probability of introduction rather than the full extent of suitable habitat.

Furthermore, small-scale variations in environmental characteristics, in particular from disturbances and human activity, may not be represented in the data, but strongly influence where an invasive plant will establish.

More recently, multi-criteria decision analysis has played a role in invasive species management (Born *et al.* 2005, Cook and Proctor 2007). Despite mostly being used to aid complex cost-benefit decisions, when developed as a multi-criteria model within a geographic information system (GIS) this analysis method can be used to combine multiple layers of data across a landscape. These models combine information about criteria to generate a single value that can be assessed with less difficulty (Eastman 1999). The spatial output from the models can then be used to create risk maps. A detailed method for creating multi-criteria risk models and risk maps was developed for forest insects and diseases (Krist *et al.* 2007). By altering these methods slightly, multi-criteria risk models and risk maps can be developed for invasive plants as well.

Previous modeling attempts have rendered some success; nevertheless a number of recommendations have been made that should increase the precision and accuracy of invasion predictions. Underwood *et al.* (2004) suggested that the incorporation of human disturbances, vectors of spread, natural disturbances, and propagule pressure, although difficult to model, would result in more precise predictions. They also noted their concern with using data from invasive plants in the initial phases of invasion as the full extent of suitable habitat is not yet realized. Dark (2004) also observed the importance of human disturbances in predicting invasion. In a model developed for invasive plants in California, road density had a significant influence on the number of invasive plants present in an area (Dark 2004). Another consideration when assessing model function is knowledge of management and control activity that would limit the natural spread of an invasive plant. Higgins *et al.* (1999) advised not to overlook areas where invasive species have been removed or treated when ground-truthing predictive models.

The development of a multi-criteria risk model utilizing these recommendations would be beneficial. The improvement of monitoring and control efforts is necessary to combat the increasing population of invasive plants. The model would be useful in areas with populations already established and in areas where the invasive plant is not yet present. The model predictions could be used to identify new populations through more efficient detection, and could also reduce movement by allowing land managers to focus control on high-risk populations and locations. The development of a predictive model is extremely important if invasive plants are to be managed effectively. Monitoring can focus on areas with a high risk for introduction and establishment, and populations in areas with high risk of establishment and spread can promptly receive priority for control.

Literature Cited

Baker, H. G. 1984. Patterns of plant invasion in North America *in* Mooney, H. A. and J. A. Drake (Eds). *Ecology of biological invasions in North America and Hawaii*. Springer-Verlag, New York.

Born, W., F. Rauschmayer, and I. Brauer. 2005. Economic evaluation of biological invasions-a survey. Ecol Econ 55:321-336.

Cain, M. L., B. G. Milligan, and A. E. Strand. 2000. Long-distance dispersal in plant populations. Am J Bot 87:1217-1227.

Callaway, R. M. and W. M. Ridenour. 2004. Novel weapons: invasive success and the evolution of increased competitive ability. Front Ecol Environ 2:436-443.

Coblentz, B. E. 1990. Exotic organisms: A dilemma for conservation biology. Conserv Biol 4:261-265.

Cook, D. and W. Proctor. 2007. Assessing the threat of exotic plant pests. Ecol Econ 63:594-604.

Dark, S. J. 2004. The biogeography of invasive alien plants in California: an application of GIS and spatial regression analysis. Diversity Distrib 10:1-9.

Eastman, J. R. 1999. Multi-criteria evaluation and GIS. *in* Longley, P. A., M. F. Goodchild, D. J. Maguire, and D. W. Rhind (Eds). 1999. *Geographical Information Systems*. Wiley, New York.

Ehrenfeld, J. G. 2003. Effects of exotic plant invasions on soil nutrient cycling processes. Ecosystems 6:503-523.

Goodwin, B. J., A. J. McAllister, and L. Fahrig. 1999. Predicting invasiveness of plant species based on biological information. Conserv Biol 13:422-426.

Gordon, D. R. 1998. Effects of invasive, non-indigenous plant species on ecosystem processes: lessons from Florida. Ecol Appl 8:975-989.

Graham, K. L. 2002. Human influences on forest wildlife habitat *in* Wear, D. N. and J. G. Greis (Eds). Southern forest resource assessment. Gen. Tech. Rep. SRS-53. U.S. Department of Agriculture, Forest Service, Asheville, North Carolina.

Hawkins, D. M. 1999. FIRM: formal inference-based recursive modeling. University of Minnesota, St. Paul, MN.

Higgins, S. I. and D. M. Richardson. 1996. A review of models of alien plant spread. Ecol Model 87:249-265.

Higgins, S. I. and D. M. Richardson. 1999. Predicting plant migration rates in a changing world: the role of long-distance dispersal. Am Nat 153:464-475.

Higgins, S. I., D. M. Richardson, R. M. Cowling, and T. H. Trinder-Smith. 1999. Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. Conserv Biol 13:303-313.

Keane, R. M. and M. J. Crawley. 2002. Exotic plant invasions and the enemy release hypothesis. Trends Ecol Evol 17:164-170.

Krist, F. J. Jr., F. J. Sapio, and B. M. Tkacz. 2007. Mapping risk from forest insects and diseases. U.S. Department of Agriculture Publication. FHTET-2007-06

Peterson, A. T. and D. A. Vieglais. 2001. Predicting species invasions using ecological niche modeling: new approaches from bioinformatics attack a pressing problem. BioScience 51:363-371.

Peterson, A. T., M. Papes, and D. A. Kluza. 2003. Predicting the potential invasive distributions of four alien plant species in North America. Weed Sci 51:863-868.

Pimentel, D., R. Zuniga and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecol Econ 52:273-288.

Pimm, S. L., G. J. Russell, J. L. Gittleman, and T. M. Brooks. 1995. The future of biodiversity. Science 269:347-350.

Rejmanek, M. 2000. Invasive plants: approaches and predictions. Austral Ecol 25:497-506.

Rejmanek M. and D. M. Richardson. 1996. What attributes make some plant species more invasive? Ecology 77:1655-1661.

Rouget, M., D. M. Richardson, S. J. Milton, and D. Polakow. 2001. Predicting invasion dynamics of four alien *Pinus* species in a highly fragmented semi-arid shrubland in South Africa. Plant Ecol 152:79-92.

Shigesada, N., K. Kawasaki, and Y. Takeda. 1995. Modeling stratified diffusion in biological invasions. Am Nat 146:229-251.

Stockwell, D., and D. Peters. 1999. The GARP modeling system: problems and solutions to automated spatial prediction. Int J Geogr Inf Sci 13:143–158.

Sutherland, S. 2004. What makes a weed a weed: life history traits of native and exotic plants in the USA. Pop Ecol 141:24-39.

Westbrooks, R. 1998. Invasive Plants, Changing the Landscape of America: Fact Book. Federal Interagency Committee for the Management of Noxious and Exotic Weeds (FICMNEW), Washington, D.C. 109 pages.

Underwood, E. C., R. Klinger, and P. E. Moore. 2004. Predicting patterns of non-native plant invasions in Yosemite National Park, California, USA. Diver Distrib 10:447-459.

Van Driesche, R. 2002. Introduction. *in* Van Driesche, R. Biological Control of Invasive Plants in the Eastern United States. USDA Forest Service Publication FHTET-2002-04.

With, K. A. 2002. The landscape ecology of invasive spread. Conserv Biol 16:1192-1203.

Chapter Two

Development of Multi-Criteria Risk Models for Invasive Plant Species in the Great Lakes Network of the National Park Service

Abstract

Invasive plants impair the long-term health of natural ecosystems by changing characteristics such as species composition, water flow, and nutrient availability. For this reason, invasive plants are one of the "Vital Signs" monitored annually by the National Park Service (NPS). Multi-criteria risk models were developed to assist with the detection of ten target invasive plants in the Great Lakes Network of the National Park Service. The models function by inputting biological, environmental, and anthropogenic data, assigning risks, and weighting the parameters to calculate a risk value at three phases of invasion: introduction, establishment, and spread. An accuracy assessment of the models showed an average overall accuracy of 86.3%. Broken down by phase the models correctly predicted a high risk for 77.5% of known invasions for introduction, 90.3% for establishment, and 91.2% for spread. These results were consistent with those obtained from data collected during field sampling at two National Parks. Used as one of many resources, the risk maps created from the model output can assist with the detection of ecosystem change through improved monitoring methods.

Introduction

The National Park Service (NPS) uses the annual monitoring of "Vital Signs" to detect ecosystem change within the National Parks. The NPS defines a Vital Sign as "a physical, biological, chemical element or process that: indicates the health of a park ecosystem, responds to natural or anthropogenic stresses in a predictable or hypothesized manner, or has high value to the park or the public" (Route and Elias 2007). Vital Signs guide the NPS by indicating ecological changes that create problems and will require management or further research (Route and Elias 2007). Invasive exotic plants are a clear indicator of ecosystem change. Invasive plants have the ability to impair the long-term health of a natural ecosystem by altering species composition, resource availability, structure, and function, among other things. The NPS ranks plant and animal exotics as the highest priority Vital Sign for the Great Lakes Network (Route and Elias 2007).

Monitoring for invasive plants in the National Parks is a daunting task. The Great Lakes Network of the NPS is made up of nine National Parks totaling 471,264 hectares. The time and resources needed to accomplish the in-depth monitoring required to detect individual invasions or small populations of invasive plants in a large area such as a National Park are often not available. The use of models to predict the locations where invasive plants are likely to occur would assist with monitoring and management. This information would allow monitoring to focus on high-risk areas, and when compared with known invasions would indicate the areas that should be given priority for control treatments.

Predictive modeling can be used to estimate distribution and spread of established invasive plants. The distributions of four invasive plants were predicted across North

America using ecological niche modeling creating maps of suitable habitat (Peterson *et al*. 2003). The distribution of invasive plants in California was used to determine significant predictors for sites most likely to be invaded (Dark 2004). Models of invasive plant spread have also been developed to explain the process of invasion and spread (Higgins and Richardson 1996). Predictive models that assist in monitoring for new introductions of invasive plants would be of great use. At Yosemite National Park, the potential invasion pattern was predicted for a set of non-native species (Underwood *et al*. 2004). These predictions were used to target monitoring and control efforts to high-priority areas within the Park (Underwood *et al.* 2004). Multi-criteria models have rarely been used to assess the risk of plant invasions, however they have the potential to play an important role (Cook and Proctor 2007). Multi-criteria models can assist with making complex management decisions (Cook and Proctor 2007). Used to guide invasive plant monitoring, these models can generate a single risk value from multiple criteria, which can be used to create risk maps that will guide monitoring (Eastman 1999).

The overall goal of this research project was to produce multi-criteria risk models that utilize geographic information system (GIS) data to determine the areas at greatest risk for invasive plant species in the Great Lakes Network (GLKN) of the National Park Service (NPS). This work provides a basis for monitoring invasive plants to detect ecosystem change within National Parks. The resulting risk maps will be used as a resource to assist the management of invasive plants through improved monitoring and control efforts. The main objectives were to: 1) obtain and/or create GIS data for nine National Parks and ten target invasive plants, 2) develop multi-criteria risk models that predict invasion at three phases, introduction, establishment, and spread, and 3) test and refine the models using known invasions and field sampling.

Methods

Region of Study

The National Park Service divides its Parks into 32 management networks. This study focused on the Great Lakes Network (GLKN). The GLKN manages nine National Parks (Table 1): Apostle Islands National Lakeshore (APIS), Grand Portage National Monument (GRPO), Indiana Dunes National Lakeshore (INDU), Isle Royale National Park (ISRO), Mississippi National River and Recreation Area (MISS), Pictured Rocks National Lakeshore (PIRO), Saint Croix National Scenic River (SACN), Sleeping Bear Dunes National Lakeshore (SLBE), and Voyageurs National Park (VOYA). These parks encompass a total area of 471,264 ha, and are located in the Great Lakes states of Indiana, Michigan, Minnesota, and Wisconsin (Figure 1). This project focuses on four common habitat types found within the Parks: coastal dune, marsh/bog, woodland, and grassland.

Target Invasive Plants

Within the GLKN, ten species were identified as target invasive plants: baby's breath (*Gypsophila paniculata* L.), common buckthorn (*Rhamnus cathartica* L.), common reed (*Phragmites* spp.), garlic mustard (*Alliaria petiolata* [Bieb] Cavara and Grande), glossy buckthorn (*Frangula alnus* P. Mill.), honeysuckle (*Lonicera* spp.), leafy spurge (*Euphorbia esula* L.), multiflora rose (*Rosa multiflora* Thunb. ex Murr.), purple loosestrife (*Lythrum salicaria* L.*)*, and spotted knapweed (*Centaurea biebersteinii* DC.) (Table 2, Appendix 1). These ten plant species are thought to present the highest risk to natural ecosystems and pose the greatest challenge to management within the GLKN. The species identified are diverse as each invades different habitat types and has unique biological and environmental requirements. However, all of the ten species can be found in one or more of the common habitat types associated with the GLKN.

Model Development

To develop the multi-criteria risk models, the invasion process was broken into three phases: introduction, establishment, and spread. Definitions of each phase were developed to determine which factors would be used to predict invasion at each phase. The definitions were adapted from several previous definitions (see Williamson 1996, Richardson *et al*. 2000, Kolar and Lodge 2001, Sakai *et al*. 2001). The invasion process was considered to follow an S-shaped population growth curve (Figure 2). Introduction was defined as the arrival of a species in an area where it was not currently present. Introduction involves the dispersal of seeds to an area as well as successful germination of the seeds and survival of seedlings. Establishment follows introduction and was defined as the development of a free-living, reproducing population of a species. Establishment requires the survival of plants past the seedling stage, combined with successful reproduction. Most factors affecting establishment are environmental characteristics of the landscape, but other factors such as fire regime, human land use, and disturbance play roles as well. Spread was defined as the increase in population size of an established population. Spread involves short- and long-distance dispersal of seeds and survival and growth of the established population. A population can grow by increasing the number of individuals or by increasing the size of the area invaded. The potential for population spread can be greater in large areas of connected suitable habitat or on disturbed land.

The risk of invasion for a given area varies by phase since each phase has different requirements. An initial list of risk factors affecting invasion was created for each phase based on a literature review for each species (for bibliography see Appendix 2). The preferences and requirements of each invasive plant were determined for each risk criteria. Important criteria were incorporated into the models as parameters. Criteria were unimportant if there was little variation within Parks, or if the invasive plant had no requirements or preferences discernable from the literature for that criterion. Furthermore, to be used in the model, the criteria needed to have obtainable or creatable spatial data across the nine National Parks. The NPS provided all existing GIS data layers for the National Parks in the GLKN. This generally consisted of spatial data layers for Park boundary, hydrology, hypsography, roads, railroads, trails, vegetation, recreation features, land use, and disturbance features. Additionally, Soil Survey Geographic (SSURGO) data was obtained from the Natural Resources Conservation Service for each Park. When not available, coarser scale State Soil Geographic (STATSGO) data was used. From the soil data, drainage and average pH were determined for the soil types present. All spatial data were converted to or created in coverage or shapefile format, and included relative attribute data indicating source, date, and species if applicable. A common datum and coordinate system was used within each Park, but not across Parks.

An additional parameter used in the model was the presence of known invasions. Most of the National Parks monitor invasive plants and develop GIS data each year based on known locations of target species. The NPS provided any GIS data available for the target invasive plants. These data occasionally included information on areas where invasions occurred and had been treated. Such data were included as known locations since these sites were known to be suitable for invasion, and because it was not possible to determine whether treatment had been successful. Every species had invasion data for at least one Park and many species were recorded at multiple parks (Table 3). Only one park, GRPO, did not have any GIS data on invasive plant invasions. Despite this, it is likely that some of these plants are present at this site. The known invasion data were only used after model development to test the initial accuracy of the models and did not influence criteria selection or the assignment of risks to parameters.

Using the lists of significant criteria, a unique risk model was developed for each invasive plant using ModelBuilder within ArcGIS 9.2 (ESRI 2006, Figure 3). The information from the literature search guided the assigning of risk values to parameter levels. Risk values ranged from zero to ten, with zero being equivalent to no risk and ten equivalent to high risk. When possible, a typical curve shape, such as a normal or sigmoid curve, was applied using information on when risk begins, peaks, and ends for a particular parameter. For categorical data, such as vegetation, risk was determined by considering the suitability of each category to both invasion and survival. Assigned risks varied by the phase of invasion being considered. For categorical data, reclassification tables were created for every species and Park combination, which were entered into the model as a parameter. For example, vegetation data was grouped by cover type and had to be reclassified into the correct risk value for each category. The appropriate risk values for categorical data were determined using the information gained from the literature search, and were altered if needed during model adjustment. Each parameter in the model was assigned a rank and a confidence level. These values were weighted, with the rank having three times more importance than the confidence level, to calculate an influence value for each parameter. The influence was the percent weight calculated for the parameter, with the sum of all influences equal to 100%. Within the models the influence was entered into the weighted overlay table for each phase of invasion. During model development, the models were run using the data collected and compiled for each Park to find the optimal model structure.

Model Analysis and Adjustment

Once all of the models were completed, they were run for each of the 90 species and Park combinations. The output for each combination consisted of three 10 m x 10 m raster grids, one for each phase of invasion. The raster contained the risk values associated with each pixel of the Park and the surrounding area. The risks range from 0 to 10, with 10 representing highest risk. The invasive plant data provided by the National Parks were used to assess the initial accuracy of the models. For each species at each Park where invasion data were available (Table 3), the risk ratings for areas with a known invasion were extracted from the model output. The percent of pixels with a known invasion and a high risk rating (risk ≥ 7) was calculated. This was considered to be a measure of the accuracy of the model to correctly predict a high risk for areas both suitable to invasion and with a high probability of invasion. A model was considered to be sufficiently accurate if it correctly assigned a high risk rating to at least 70% of the known invaded pixels. This value was selected as a general rule to guide model adjustment and did not carry any statistical significance. An overall accuracy was calculated for all phases of invasion, all Parks, and all species by weighting the individual Park and species results by the number of invaded pixels and averaging this over the three phases of invasion. The results were further compared at various scales based on overall accuracy, accuracy by species, accuracy by Park, and accuracy by invasion phase. Models with insufficient accuracy were first examined to look for unnoticed errors in model structure and function. Then the presence data were compared to each of the model parameters to determine if incorrect or absent parameter data could be the cause of decreased model accuracy. When possible, missing or inaccurate data were replaced. Finally, the weights and risks assigned to the parameters were adjusted to obtain a consistent accuracy across all nine parks and invasion phases for each species model.

Field Sampling

During the summer 2006 field season initial ground-truthing took place within two National Parks, INDU and SLBE. This consisted of sampling randomly-generated points within these areas for the presence and abundance of the ten target invasive plants. The points were generated within the boundaries of each Park, excluding open water, using the Random Point Generator (Sawada 2002). Based on an estimate of required time and effort, an initial set of 75 points was generated for each Park. The points were spatially stratified, with the sample size based on the area of contiguous Park sections. The points were also assessed to ensure that each of the four common habitat types found in the Parks of the GLKN were represented. The GPS coordinates of each point were downloaded onto a Garmin GPS Map 76 unit. Using maps of the random points a field crew of three navigated to within 20 m of each point. In some rare cases points were inaccessible to within 20 m and sampling was done as close to the random point as possible. This was most often due to areas being restricted by fencing or open water. At each point, the exact coordinates were recorded as well as the accuracy of the GPS unit. A decrease in accuracy was noticeable under full cloud cover or dense canopy cover. The locations of the random points were later adjusted to the coordinates recorded in the field.

At each point a 40 m x 40 m plot was assessed for the presence of the ten invasive plants. The random point served as the center of the plot, and a compass was used to align the plot with the cardinal directions. The plot was divided into sixteen 10 m x 10 m blocks. Within these blocks, the presence as well as percent cover was recorded for each of the ten invasive plants. Percent cover was based on a visual assessment and was ranked on the following scale: rare (R, <1% cover), occasional (O, 1-10% cover), common (C, 10-25% cover), abundant (A, 25-50% cover), dominant (D, 50-100% cover). Additional notes were taken describing the site characteristics including presence of roads, railroads, and water, general vegetation type, presence of target invasive plants just outside of a plot or while traversing to a plot, and evidence of invasive plant treatment or removal. This information was used to evaluate incorrect predictions at single points to determine the cause of error.

During the summer 2007 field season, additional random points were sampled at SLBE to more intensively assess the accuracy of the final versions of the models. In order to do so, an appropriate sample size was determined using a standard sample size formula (Levy and Lemeshow 1999). Considering the available time and effort, a confidence level of 0.90 and a relative error of 0.20 were selected. This suggested a sample size of approximately 168. The points were generated using the same methods as used previously, and stratified spatially with equal amounts given to the three sections of the Park. The points were assessed to ensure that each of the four common habitat types and varying levels of risk for each species were represented. The field methods were similar to the previous sampling effort, although only one measure of presence and percent cover was taken for the entire 40 m x 40 m plot. The occurrence data were used to assess the accuracy of the models following adjustment. The cover data were used to assess the differences predicted for the three phases of invasion.

The data collected during the summer 2006 field season were assessed using the output from the adjusted models. From this the models were adjusted, if needed, and the final models were reassessed and validated using the summer 2007 field data. For each sample point, the predicted risk value was extracted from the output for each species. The value was interpolated so that it also represented the eight pixels surrounding that of the sampling point. The percentage of points with a high risk and occurrence of a known invasion was calculated for each species. The species accuracies were combined to create an average accuracy for each phase by weighting the species accuracies by the number of observations. These values were then averaged over the three phases to calculate an overall accuracy. Models that correctly predicted high risks for known invasions were considered to show a high degree of accuracy. To evaluate the differences in risk among phases, risk values were extracted using interpolation for each phase. The change in risk across phases for each species was assessed qualitatively. Points with a rare or occasional cover rating were considered to be in the introduction phase, points with a common cover rating were considered to be in the establishment phase, and points with an abundant or dominant rating were considered to be in the spread phase.

Risk Map Creation

The final outputs produced by each model were used to create risk maps by species and Park highlighting the areas at greatest risk for the three phases of invasion. The risk maps indicated the areas with risk ratings from seven to ten using a yellow, orange, red, and burgundy color scheme. An overall risk map was also created for each Park in order to determine if any areas of the Park were particularly more at risk for every species. These maps could also guide multi-species monitoring if it were not possible to monitor each species individually. The overall risk maps were created by averaging the establishment risk for all ten species. The areas with the greatest combined risk value were highlighted in the same color scheme as the species-specific risk maps.

Results

Multi-Criteria Risk Models

All ten multi-criteria risk models utilized the parameters of Park boundary, transportation (distance to roads, railroads, and trails), hydrology, vegetation type, disturbance features, soil drainage, and connectivity of suitable habitat. Invasive plant presence was also used in each of the models. Distance to the nearest known invasion was a parameter for the spread phase. The models for common reed, garlic mustard, glossy buckthorn, honeysuckle, and leafy spurge also utilized soil pH as a model parameter. Transportation and hydrology were usually combined to form one parameter, which represented potential pathways for dispersal. Each phase of invasion and species model made use of a different set of the parameters, and each parameter had a unique influence weight (Table 4). Introduction was based on Park boundary, dispersal, vegetation type, disturbance, soil drainage, and soil pH (when applicable). Establishment took into account the assigned introduction risk, as well as the parameters vegetation type, disturbance, soil drainage, and when important to a given species, soil pH, hydrology, and/or transportation. Spread included the assigned establishment risk, dispersal, connectivity of suitable habitat, and the locations of known invasions.

Park Data Analysis

There were 32 combinations of species and Parks with known invasion data. Preliminary assessment of model accuracy using the data provided by the Parks showed that 75.8% of pixels with the presence of a given invasive plant were correctly assigned a high risk rating (risk ≥ 7) by the models. This was an average of the three phases of invasion, for which the individual results were 60.1% for introduction, 80.3% for establishment, and 87.1% for spread. Breaking the data down by species and Park, there was a range of results from 0% to 100% (Table 5). Following final adjustments, the overall accuracy increased to 86.4%. This was based on a total of 130,718 invaded pixels across all ten species and eight Parks with known invasions. Broken down by phase, a high risk was correctly assigned for 77.5% of invaded pixels for introduction, 90.4% for establishment, and 91.2% for spread. Two individual accuracies were lower than the target accuracy of 70% (spotted knapweed introduction and establishment at VOYA) but this was based on a sample size of only two, so was not considered an error of the model. All other individual accuracies ranged between 70.2% and 100% (Table 6).

Field Sampling

During field work in summer 2006, 75 points were sampled at INDU, and 76 points were sampled at SLBE (Figures 4 and 5). Eight of the ten target invasive plants were identified at INDU and seven were identified at SLBE (Table 7). A range of abundance levels was encountered across both Parks. At INDU multiflora rose and garlic mustard were identified most frequently, found at 25 and 21 points respectively, and with the greatest abundances. At SLBE spotted knapweed was most frequent, being found at 14 points. During additional field work at SLBE in summer 2007, 162 points were sampled (Figure 6). Six of the ten invasive plants were identified during this sampling (Table 7). Spotted knapweed was again most frequent, found at 42 points. In 2007, honeysuckle and leafy spurge were found more often and with higher abundance than in 2006. Common reed and purple loosestrife were encountered in 2006 but were not found in 2007. Garlic mustard was absent from the random sample points in 2006, but occurred at three of the 2007 random sample points.

The initial overall model accuracy by Park for the 2006 field data was 64.1% for INDU and 78.4% for SLBE. Following model correction and adjustments, the model accuracy increased for both Parks. INDU had an overall model accuracy of 86.8%, while SLBE had an overall model accuracy of 87.0%. By phase the results were 89.7% for introduction, 82.8% for establishment, and 87.9% for spread at INDU, and 91.7% for introduction, 91.7% for establishment, and 77.8% for spread at SLBE. This was based on 58 observations of target invasive plants at INDU and 36 at SLBE. For the 2007 SLBE data the overall model accuracy was 83.3%. Based on phase, the results were 88.8% for introduction, 85.0% for establishment, and 76.3% for spread. This was based on 80 observations of target invasive plants.

Model Predictions and Risk Maps

The risk maps created from the model output (Appendix 4) offered the best glimpse at the level of invasion risk predicted across Parks. The risk maps for APIS showed garlic mustard and multiflora rose as having the greatest risk across the Park. At GRPO, common buckthorn, garlic mustard, honeysuckle, multiflora rose, and spotted knapweed risk maps all showed large portions of the Park at risk. However, GRPO is the smallest Park, covering only 287 hectares. At INDU the species showing the most extensive areas at risk were garlic mustard, honeysuckle, multiflora rose, and spotted knapweed. Common reed and honeysuckle had the greatest areas of risk at ISRO. At MISS, garlic mustard, honeysuckle, and spotted knapweed risk maps all showed the majority of the Park at risk. Multiflora rose and honeysuckle risk maps had the greatest areas at risk at PIRO. At SACN common buckthorn, garlic mustard, honeysuckle, and spotted knapweed showed the greatest areas at risk. The variety of habitat types at SLBE created a range of risks across the park for each species. The SLBE risk maps for baby's breath, glossy buckthorn, garlic mustard, honeysuckle, and spotted knapweed all showed large areas at risk. At VOYA the species showing the greatest areas of risk were common buckthorn, honeysuckle, multiflora rose, and spotted knapweed. In addition, purple loosestrife showed high risk around all bodies of water at VOYA.

The overall risk maps, which reflected all ten species, showed small areas at high risk and the majority of most Parks at low or moderate risk (Appendix 5). Overall risk maps for GRPO, INDU, MISS, and PIRO showed distinct areas of high risk, despite these making up only a small portion of the Parks. Overall maps for SACN and SLBE showed a few small areas of high risk, and overall maps for APIS, ISRO, and VOYA showed no noticeable areas of high risk.

Discussion

The multi-criteria risk models correctly identified invaded areas as high risk for introduction, establishment, and spread for each of the ten target invasive plants. The models also predicted high risks for areas identified as invaded through field sampling. The success of the models throughout initial testing, adjustments, and final assessment indicates that they will be of use to assist with monitoring invasive plant presence and managing known populations of these ten invasive plants. The accuracy results were similar to those obtained by Underwood *et al*. (2004), who correctly predicted 76% of invasive plant species presence at Yosemite National Park. Underwood *et al*. (2004) also noted that areas with a high level of human activity and disturbance, such as campgrounds, roads, and trails, were predicted to have higher probability of invasive plant occurrence, despite not being included as a factor in their model. The success of the models created in this project is due in part to the addition of these anthropogenic parameters.

The parameters selected for the multi-criteria risk models differed notably from other similar predictive models (Peterson *et al.* 2003, Dark 2004, Underwood *et al.* 2004). Human activity was stressed through the addition of transportation and disturbance features. Elevation was not included, but has been a common factor in other predictive models (Dark 2004, Underwood *et al*. 2004). It was not included here since it offered no limitation to plant invasion by the target invasive plants. This was also encountered by Peterson *et al*. (2003), who eliminated elevation from a predictive model for garlic mustard in California. Additionally, elevation may not have had an effect due to the lack of variation across the Great Lakes Region and within Parks. Other common factors were also excluded, such as climate, which had little variation across the region, and slope and aspect, which were eliminated due to insufficient evidence of association with the target invasive plants.

The accuracy for introduction was lower than that of establishment and spread during analysis of the known invasion data. This could be due to the stochastic nature of invasive plant introductions, making them more difficult to model. Propagule pressure is particularly difficult to measure and represent spatially as GIS data (Lockwood *et al*. 2005). In the analysis of the field data, the spread accuracy was generally lower than the spread accuracy for the Park data. This was expected, since distance to the nearest invasion was included as a parameter for the spread phase, and was based on the same dataset being used to analyze the model. This parameter is important, however, since it will create higher spread risks surrounding known invasions, where propagule pressure is expected to be highest, and where management is most needed.

Parks with a high density of roads generally showed a greater extent of area at risk than Parks with few roads. This was expected since roads provide pathways of dispersal, areas of disturbance, and in general, a habitat with ample light and water (Forman and Alexander 1998). Dark (2004) also found that roads played an important role in the locations of plant invasions. ISRO, which has only a few trails and no roads, had few areas with a high risk of invasion. INDU, MISS, and SACN are located in more urban locations and had grids of roads visible in the risk maps since the roads ranked higher in risk relative to the surrounding areas. It is likely correct to assume an overall lower risk for ISRO since there are fewer opportunities for dispersal and less disturbance, while it is accurate for INDU, MISS, and SACN to have higher risks due to high levels of disturbance and many opportunities for dispersal by humans.

The methods used for this project helped to avoid one common obstacle to predictive modeling: the amount of data points required to both create a model and validate its accuracy. By using a literature review to determine the environmental preferences of each invasive species, fewer data points were needed overall. This permitted an initial accuracy assessment of the models, model adjustment, and a final assessment of accuracy. During the literature review, information was collected from many sources, across diverse sites. As a result, the models were useful within different areas and not limited to the extent of known invasions. This allowed for risk assessment in National Parks where the invasive plant is not yet present.

The setup of the multi-criteria risk models allows for the parameter influences to be easily adjusted within the ArcGIS ModelBuilder program (ESRI 2006). It is also possible to manipulate the lower and upper limits of a parameter, as well as each level within this range. This will be useful for tailoring the model to the specific site where it is being applied. For example, at INDU, multiflora rose was planted along railroads and roads and is continuing to spread further along these routes. In the model, the influence of dispersal (roads, railroads, and trails) could be increased since it is strongly related to the presence of multiflora rose. The ease of tweaking the models also makes them easy to update as new research and management information becomes available.

The risk maps created from the model output will be useful for invasive plant management. However, predictive models should not be used as the only resource when making management decisions. The results are only predictions. The invasion of exotic species is difficult to predict accurately because of the complex relationship between predictors and the chance occurrences of dispersal and disturbance that facilitate introduction and establishment. More detailed information on ecosystem properties, such as species present, population sizes, and resource availability, are necessary to determine the probability of invasion in a specific area (Stohlgren and Schnase 2006). This information is not readily available as GIS data, and would be difficult to obtain over a large area. Interspecies interactions, such as competition, herbivory, and predation, are also important factors that are difficult to quantify and, therefore, are not included (Stohlgren and Schnase 2006). A clearer understanding of dispersal is also needed.

Lockwood *et al*. (2005) stressed the need for propagule pressure to be incorporated into the predictive modeling of invasive species. Unfortunately, propagule pressure is difficult to quantify, as pathways and rates of introduction are complex and poorly understood (Lockwood *et al*. 2005). What is more, the degree of propagule pressure required to establish populations may have an association with disturbance and ecosystem properties, further confounding the ability to measure and utilize these aspects as predictors (Lockwood *et al*. 2005).

The process of predictive modeling inevitably has flaws and limitations. Error can occur during parameter risk estimation, or can arise from limitations in the input data. The predictions must be balanced between over- and under-fitting, which, when modeling invasive plants, is difficult to assess since their full invasion potential is unknown. The standard strategy for assessing actual accuracy is to test for statistical significance of randomly selected subsets of the data (Stockwell and Peters 1999). With invasive plants this is not possible since they have not reached their full potential distribution, and therefore sample points with the absence of an invasive plant may in reality be an optimal invasion site.

Predictive models are also limited by the quality of the data (e.g. GIS layers) being used. Data may not be complete due to missing information or from estimates and generalizations made during surveying, such as listing the vegetation type as the dominant species when a variety of other species exist at the site (Stohlgren and Schnase 2006). In this project, missing or incomplete data affected the ability of the models to correctly predict risk. For example, 4.68%, nearly 40 hectares, of SLBE had undefined soil pH. The precision and scale of the data should also be considered. In the case of some Parks, coarse scale STATSGO data had to be used rather than more detailed SSURGO data, since these data were not available for every area. The Parks affected by this were ISRO, GRPO, PIRO and the southern portion of SLBE. In the case of ISRO, the soil data only provided one soil type and therefore only one soil pH and soil drainage for the entire Park. This may have affected the ability of the model to predict areas of risk precisely, resulting in similar risk predictions across the Park. Once available, running the models with completed SSURGO data may provide more accurate and precise results.

Although the models can be adjusted to a specific Park, a multi-criteria risk model created for one specific Park may be more efficient and result in higher accuracy. Risk assessment focused on a particular species in a specific area has had more success than larger-scale, multiple species predictions (Stohlgren and Schnase 2006). Working with smaller areas allows for a more in-depth assessment of ecosystem properties and processes that affect the probability of invasion. The overall risk maps indicated that there is likely an association between the occurrences of these invasive plants. Areas along roads were clearly at high risk for all species at Parks such as INDU and MISS, and more than likely at risk at other Parks as well. However, this may not be the case for wetland species, which would rely more on the locations of wet habitat along lakes and streams. Grouping the target invasive plants by common habitat type and creating combined risk maps should provide improved results for monitoring and managing multiple species.

At many of the National Parks, the target invasive plants have already invaded, established populations, and begun to spread. Once established, control can be used to remove the plants or to stop the spread of the population either in size or to new locations. Time, effort, and funding are not always available to implement control methods for every population. In this case the best management option is to treat populations with the highest potential for spread and the greatest threat to ecosystems. By consulting the models and risk maps, known populations can be prioritized for treatment based on their risk of establishment and spread. When multiple species of invasive plants are present, combined species risk maps should be utilized.

Conclusion

Despite some limitations, the multi-criteria risk models developed in this project will be useful tools. Assessment of the models should be continued as they are utilized and adjusted. Three tasks were suggested by Underwood *et al*. (2004) to continue improving their predictive model for plant invasions at Yosemite National Park: rapid ground validation, known occurrence mapping, and experimental studies to understand the relationship between species occurrence and predictors. The models created for the Great Lakes Network were field tested at only two of the nine National Parks, Indiana Dunes National Lakeshore and Sleeping Bear Dunes National Lakeshore. Further ground-truthing would be beneficial toward understanding the function and accuracy of the predictive models. Mapping new locations of the target invasive plants will also be important for assessing the accuracy of the model predictions over time. This includes identifying locations where invasive plants have been removed or treated, as this will affect the accuracy when ground-truthing the models (Higgins *et al.* 1999).

Monitoring National Parks for ecosystem change is a difficult but important task, since these areas protect threatened and endangered plants, animals, and ecosystems, show historical conditions, and provide recreation and enjoyment to visitors. The models will be able to assist the NPS with the management of these ten invasive species and can be easily adapted to fit other areas. Used as one of many resources, the risk maps created from the model output can assist with management of invasive plants through improved monitoring and control efforts by focusing on areas at highest risk for individual invasive plants or for groups of similar species.

Literature Cited

Dark, S. J. 2004. The biogeography of invasive alien plants in California: an application of GIS and spatial regression analysis. Diversity Distrib 10:1-9.

Devine, B. 1999. Clear-cut mission. Communities unite to free native landscapes from the grip of invasive species. Nature Conservancy 49:12-17.

ESRI. 2006. ArcGIS 9.2. Badlands, CA.

Forman, R. T. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Ann Rev Ecol Syst 29:207-231.

Higgins, S. I. and D. M. Richardson. 1996. A review of models of alien plant spread. Ecol Model 87:249-265.

Higgins, S. I., D. M. Richardson, R. M. Cowling, and T. H. Trinder-Smith. 1999. Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. Conserv Biol 13:303-313.

Kolar, C. S. and D. M. Lodge. 2001. Progress in invasion biology: predicting invaders. Trends Ecol Evol 16:199-204.

Levy, P. S. and S. Lemeshow. 1999. *Sampling of Populations*. Wiley-Interscience; New York.

Lockwood, J. L., P. Cassey, and T. Blackburn. 2005. The role of propagule pressure in explaining species invasions. Trends Ecol Evol 20:223-228.

Peterson, A. T., M. Papes, and D. A. Kluza. 2003. Predicting the potential invasive distributions of four alien plant species in North America. Weed Sci 51:863-868.

Petrella, S., N. Shutt, and R. McNeill. 1999. A Survey of Invasive Exotic Plants in Seney National Wildlife Refuge. U. S. Fish and Wildlife Service Puplication.

Richardson, D. M., P. Pysek, M. Rejmanek, M. G. Barbour, F. D. Panetta, and C. J. West. 2000. Naturalization and invasion of alien plants: concepts and definitions. Diver Distrib 6:93-107.

Route, B. and J. Elias (eds). 2007. Long-term Ecological Monitoring Plan: Great Lakes Inventory and Monitoring Network. National Park Service, Fort Collins, Colorado.

Sakai, A. K., F. W. Allendorf, J. S. Holt, D. M. Lodge, J. Molofsky, K. A. With, S. Baughman, R. J. Cabin, J. E. Cohen, N. C. Ellstrand, D. E. McCauley, P. O'Neil, I. M. Parker, J. N. Thompson, and S. G. Weller. 2001. The population biology of invasive species. Ann Rev Ecol Syst 32:305-332.

Sawada, M. 2004. Random Point Generator. Available from ESRI ArcScripts (http://arcscripts.esri.com/details.asp?dbid=12098).

Stockwell, D., and D. Peters. 1999. The GARP modeling system: problems and solutions to automated spatial prediction. Int J Geogr Inf Sci 13:143–158.

Stohlgren, T. J. and J. L. Schnase. 2006. Risk analysis for biological hazards: what we need to know about invasive species. Risk Anal 26:163-173.

Underwood, E. C., R. Klinger, and P. E. Moore. 2004. Predicting patterns of non-native plant invasions in Yosemite National Park, California, USA. Diver Distrib 10:447-459.

United States Forest Service, Hiawatha National Forest. 1998. Michigan's Upper Peninsula Weeds. Collaborative brochure.

Van Driesche, R. 2002. Introduction. *In* Van Driesche, R. Biological Control of Invasive Plants in the Eastern United States. USDA Forest Service Publication FHTET-2002-04.

Voss, E.G. 1985. Michigan Flora. A Guide to the Identification and Occurrence of the Native and Naturalized Seed Plants of the State. Part II. Dicots. Kingsport Press; Bloomfield Hills, Michigan.

Voss, E.G. 1996. Michigan Flora. A Guide to the Identification and Occurrence of the Native and Naturalized Seed Plants of the State. Part III. Dicots Concluded. Kingsport Press; Bloomfield Hills, Michigan.

Williamson, M. 1996. *Biological Invasions*. Chapman & Hall, London.

Table 1. The National Parks within the Great Lakes Network and their associated codes.

National Park	Code
Apostle Islands National Lakeshore	APIS
Grand Portage National Monument	GRPO
Indiana Dunes National Lakeshore	INDU
Isle Royale National Park	ISRO
Mississippi National River and Recreation Area	MISS
Pictured Rocks National Lakeshore	PIRO
Saint Croix National Scenic River	SACN
Sleeping Bear Dunes National Lakeshore	SLBE
Voyageurs National Park	VOYA

Table 2. The ten target invasive plants within nine National Parks in the Great Lakes Network and the code used to identify each.

Common Name	**Scientific Name**	**Code**
Baby's Breath	*Gypsophila paniculata*	BB
Common Buckthorn	*Rhamnus cathartica*	CB
Common Reed	*Phragmites* spp.	CR
Garlic Mustard	*Alliaria petiolata*	GN
Glossy Buckthorn	*Frangula alnus*	GB
Honeysuckle	*Lonicera* spp.	HS
Leafy Spurge	*Euphorbia esula*	LS
Multiflora Rose	*Rosa multiflora*	MR
Purple Loosestrife	*Lythrum salicaria*	PL
Spotted Knapweed	*Centaurea biebersteinii*	SK

Table 3. Known occurrence of the ten target invasive plants within the National Parks of the Great Lakes Network (X = present, - = not yet detected). See Tables 1 and 2 for Park and species codes.

National Park	BB	CB	CR	GM	GB	HS	LS	MR	PL	SK
APIS	-	-	-	-	-	-	-	-	-	X
GRPO	-	-	-	-	-	-	-	-	-	-
INDU	-	-	X	X	X	X	X	X	X	X
ISRO	-	-	-	X	-	-	-	-	-	X
MISS	-	X	-	X	-	X	X	-	-	X
PIRO	-	-	-	-	-	-	-	-	-	X
SACN	-	-	-	-	-	-	X	-	X	X
SLBE	X	-	X	X	-	X	X	X	X	X
VOYA	-	-	-	-	-	-	-	-	X	-

Table 4. Parameters and influence weights used in the multi-criteria models. (PB = Park Boundary, DP = Dispersal, SD = Soil Drainage, SP = Soil pH, DT = Disturbance, HD = Hydrology, TR = Transportation, VT = Vegetation Type, I = Introduction, E = Establishment, CN = Connectivity of Suitable Habitat, IS = Invasive Species Presence)

Species	Weighted Overlay Formula
Introduction	
Baby's Breath	0.28*DP + 0.28*SD + 0.27*VT + 0.09*DT + 0.08*PB
Common Buckthorn	0.26*DP + 0.25*SD + 0.25*VT + 0.14*DT + 0.10*PB
Common Reed	0.26*DP + 0.26*SD + 0.24*VT + 0.09*SP + 0.08*DT + 0.07*PB
Garlic Mustard	0.27*DP + 0.26*VT + 0.16*SD + 0.16*SP + 0.09*DT + 0.06*PB
Glossy Buckthorn	0.24*SD + 0.23*VT + 0.14*TR + 0.14*HD + 0.10*VT + 0.10*SP + 0.07*PB
Honeysuckle	0.26*DP + 0.26*VT + 0.16*SD + 0.16*SP + 0.08*DT + 0.08*PB
Leafy Spurge	0.25*DP + 0.24*VT + 0.24*SD + 0.11*SP + 0.09*DT + 0.07*PB
Multiflora Rose	0.33*DP + 0.31*VT + 0.19*SD + 0.09*PB + 0.08*DT
Purple Loosestrife	0.33*DP + 0.31*VT + 0.19*SD + 0.09*PB + 0.08*DT
Spotted Knapweed	0.32*DP + 0.31*VT + 0.19*SD + 0.09*DT + 0.09*PB
Establishment	
Baby's Breath	0.34*SD + 0.32*VT + 0.19*DT + 0.15*I
Common Buckthorn	0.34*VT + 0.34*SD + 0.16*DT + 0.16*I
Common Reed	0.33*SD + 0.32*VT + 0.14*SP + 0.11*DT + 0.10*I
Garlic Mustard	0.27*SD + 0.26*SP + 0.25*VT + 0.12*DT + 0.10*I
Glossy Buckthorn	0.26*SD + 0.26*VT + 0.26*SP + 0.13*HD + 0.09*I
Honeysuckle	0.25*SD + 0.25*SP + 0.24*VT + 0.14*DT + 0.12*I
Leafy Spurge	0.31*VT + 0.31*SD + 0.18*SP + 0.10*DT + 0.10*I
Multiflora Rose	0.34*VT + 0.32*SD + 0.19*DT + 0.15*I
Purple Loosestrife	0.37*VT + 0.22*SD + 0.17*HD + 0.13*DT + 0.11*I
Spotted Knapweed	0.35*SD + 0.34*VT + 0.16*I + 0.15*DT
Spread	
Baby's Breath	0.37*E + 0.22*DP + 0.19*CN + 0.12*DT + 0.10*IS
Common Buckthorn	0.36*E + 0.22*DP + 0.20*CN + 0.12*DT + 0.10*IS
Common Reed	0.37*E + 0.22*DP + 0.20*CN + 0.12*DT + 0.09*IS
Garlic Mustard	0.38*E + 0.21*CN + 0.17*DP + 0.15*DT + 0.09*IS
Glossy Buckthorn	0.36*E + 0.22*DP + 0.20*CN + 0.12*DT + 0.10*IS
Honeysuckle	0.37*E + 0.22*DP + 0.19*CN + 0.12*DT + 0.10*IS
Leafy Spurge	0.37*E + 0.22*DP + 0.19*CN + 0.12*DT + 0.10*IS
Multiflora Rose	0.37*E + 0.22*DP + 0.21*CN + 0.11*DT + 0.09*IS
Purple Loosestrife	0.37*E + 0.22*DP + 0.19*CN + 0.12*DT + 0.10*IS
Spotted Knapweed	0.36*E + 0.22*DP + 0.20*CN + 0.12*DT + 0.10*IS

Table 5. The percent of invaded pixels correctly assigned a high risk for each Park, species, and invasion phase for the initial version of the models.

Species	Introduction	Establishment	Spread	Pixels
Apostle Islands National Lakeshore (APIS)				
Spotted Knapweed	4.56%	27.72%	4.91%	285
Indiana Dunes National Lakeshore (INDU)				
Common Reed	87.03%	84.50%	88.65%	987
Garlic Mustard	76.46%	87.24%	99.29%	19414
Honeysuckle	89.02%	89.95%	90.05%	965
Multiflora Rose	19.72%	46.48%	46.48%	71
Purple Loosestrife	64.67%	48.15%	69.31%	8145
Spotted Knapweed	100.00%	100.00%	100.00%	51
Isle Royale National Park (ISRO)				
Common Reed	84.19%	83.33%	83.33%	54
Spotted Knapweed	20.47%	20.47%	51.97%	197
Mississippi National River and Recreation Area (MISS)				
Common Buckthorn	23.60%	73.09%	76.79%	30101
Garlic Mustard	25.70%	59.66%	90.65%	1284
Honeysuckle	17.65%	41.18%	52.94%	17
Leafy Spurge	49.25%	62.75%	65.50%	400
Spotted Knapweed	39.14%	78.62%	92.26%	1188
Pictured Rocks National Lakeshore (PIRO)				
Spotted Knapweed	77.37%	99.42%	99.88%	3424
Saint Croix National Scenic River (SACN)				
Common Buckthorn	0.00%	100.00%	100.00%	5
Garlic Mustard	35.76%	37.75%	88.74%	151
Glossy Buckthorn	45.00%	60.00%	95.00%	20
Honeysuckle	27.27%	27.27%	27.27%	11
Leafy Spurge	100.00%	100.00%	100.00%	1
Purple Loosestrife	5.45%	10.91%	10.91%	55
Spotted Knapweed	0.00%	33.33%	66.67%	3
Sleeping Bear Dunes National Lakeshore (SLBE)				
Baby's Breath	81.27%	81.40%	81.15%	785
Common Reed	71.65%	71.65%	70.87%	127
Garlic Mustard	37.04%	15.62%	80.84%	621
Honeysuckle	94.69%	94.91%	96.15%	9258
Leafy Spurge	83.14%	92.49%	92.82%	14358
Multiflora Rose	72.31%	90.88%	94.65%	6688
Purple Loosestrife	100.00%	90.91%	100.00%	11
Spotted Knapweed	71.60%	71.60%	71.60%	81
Voyageurs National Park (VOYA)				
Purple Loosestrife	33.33%	33.33%	33.33%	3
Spotted Knapweed	50.00%	100.00%	100.00%	2
Average	**60.12%**	**80.29%**	**87.08%**	

Table 6. The percent of invaded pixels correctly assigned a high risk for each Park, species, and invasion phase for the final version of the models.

Species	Introduction	Establishment	Spread	Pixels
Apostle Islands National Lakeshore (APIS)				
Spotted Knapweed	73.33%	72.28%	81.05%	285
Indiana Dunes National Lakeshore (INDU)				
Common Reed	83.01%	70.17%	92.49%	865
Garlic Mustard	73.37%	74.99%	81.00%	28803
Honeysuckle	99.42%	99.33%	100.00%	1041
Multiflora Rose	92.49%	90.06%	93.71%	493
Purple Loosestrife	96.42%	92.15%	96.37%	8205
Spotted Knapweed	100.00%	100.00%	100.00%	51
Isle Royale National Park (ISRO)				
Common Reed	84.19%	83.33%	83.33%	54
Spotted Knapweed	83.61%	83.61%	95.38%	238
Mississippi National River and Recreation Area (MISS)				
Common Buckthorn	72.04%	96.36%	89.98%	47311
Garlic Mustard	90.63%	96.68%	99.88%	3223
Honeysuckle	100.00%	100.00%	100.00%	134
Leafy Spurge	70.81%	95.32%	95.00%	620
Spotted Knapweed	72.08%	83.18%	97.33%	4001
Pictured Rocks National Lakeshore (PIRO)				
Spotted Knapweed	78.18%	99.42%	99.56%	3424
Saint Croix National Scenic River (SACN)				
Common Buckthorn	100.00%	100.00%	100.00%	5
Garlic Mustard	88.74%	86.09%	99.34%	151
Glossy Buckthorn	90.00%	100.00%	95.00%	20
Honeysuckle	100.00%	88.89%	100.00%	9
Leafy Spurge	100.00%	100.00%	100.00%	1
Purple Loosestrife	98.18%	98.18%	100.00%	55
Spotted Knapweed	87.10%	83.87%	100.00%	31
Sleeping Bear Dunes National Lakeshore (SLBE)				
Baby's Breath	97.20%	97.20%	98.22%	785
Common Reed	99.21%	100.00%	100.00%	127
Garlic Mustard	93.60%	98.85%	100.00%	609
Honeysuckle	97.39%	94.97%	99.39%	9288
Leafy Spurge	77.21%	96.98%	97.56%	14061
Multiflora Rose	70.50%	88.18%	96.98%	6698
Purple Loosestrife	100.00%	100.00%	100.00%	11
Spotted Knapweed	72.62%	85.71%	88.10%	84
Voyageurs National Park (VOYA)				
Purple Loosestrife	100.00%	96.97%	100.00%	33
Spotted Knapweed	50.00%	50.00%	100.00%	2
Average	**77.50%**	**90.29%**	**91.21%**	

Table 7. Number of points with presence of the target invasive plant at random points sampled within Indiana Dunes (n=75) and Sleeping Bear Dunes (n=76) during summer 2006 and within Sleeping Bear Dunes (n=162) during summer 2007.

Species	**INDU**	**SLBE (2006)**	**SLBE (2007)**
Baby's Breath	1	7	7
Common Buckthorn	-	-	-
Common Reed	4	1	-
Garlic Mustard	21	-	3
Glossy Buckthorn	2	-	-
Honeysuckle	2	6	15
Leafy Spurge	-	2	11
Multiflora Rose	25	4	2
Purple Loosestrife	2	3	-
Spotted Knapweed	3	14	42

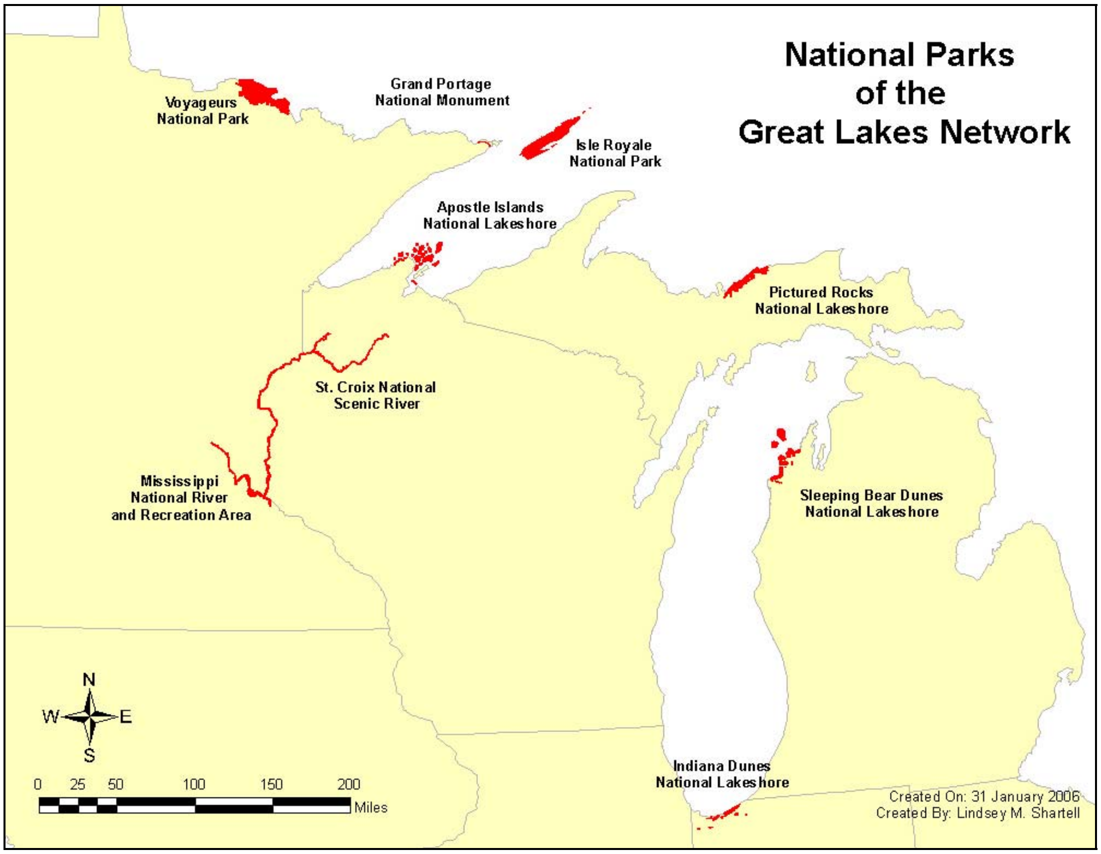


Figure 1. The locations of the nine National Parks of the Great Lakes Network.

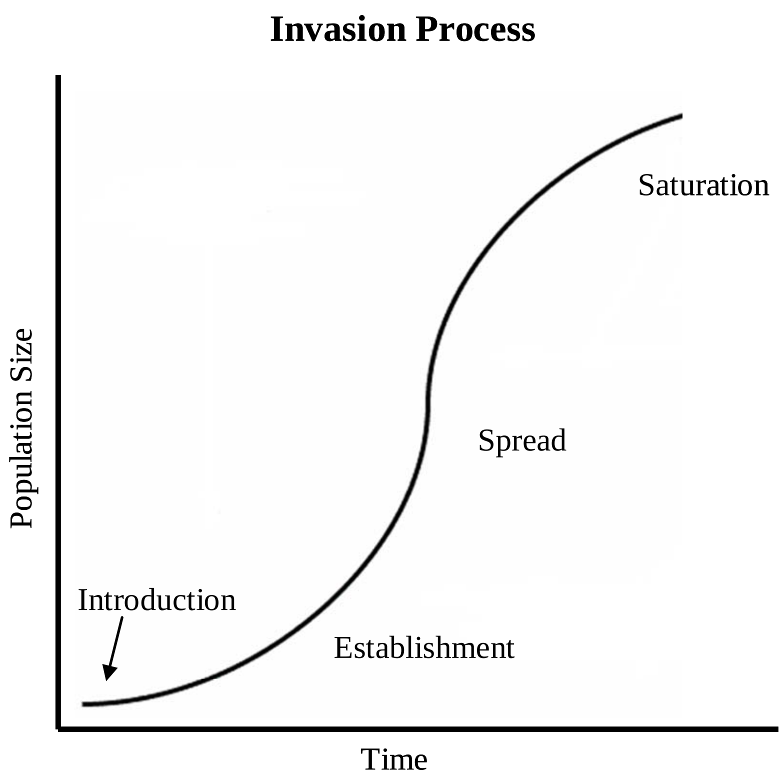


Figure 2. The phases of invasion follow an S-shaped population growth curve.

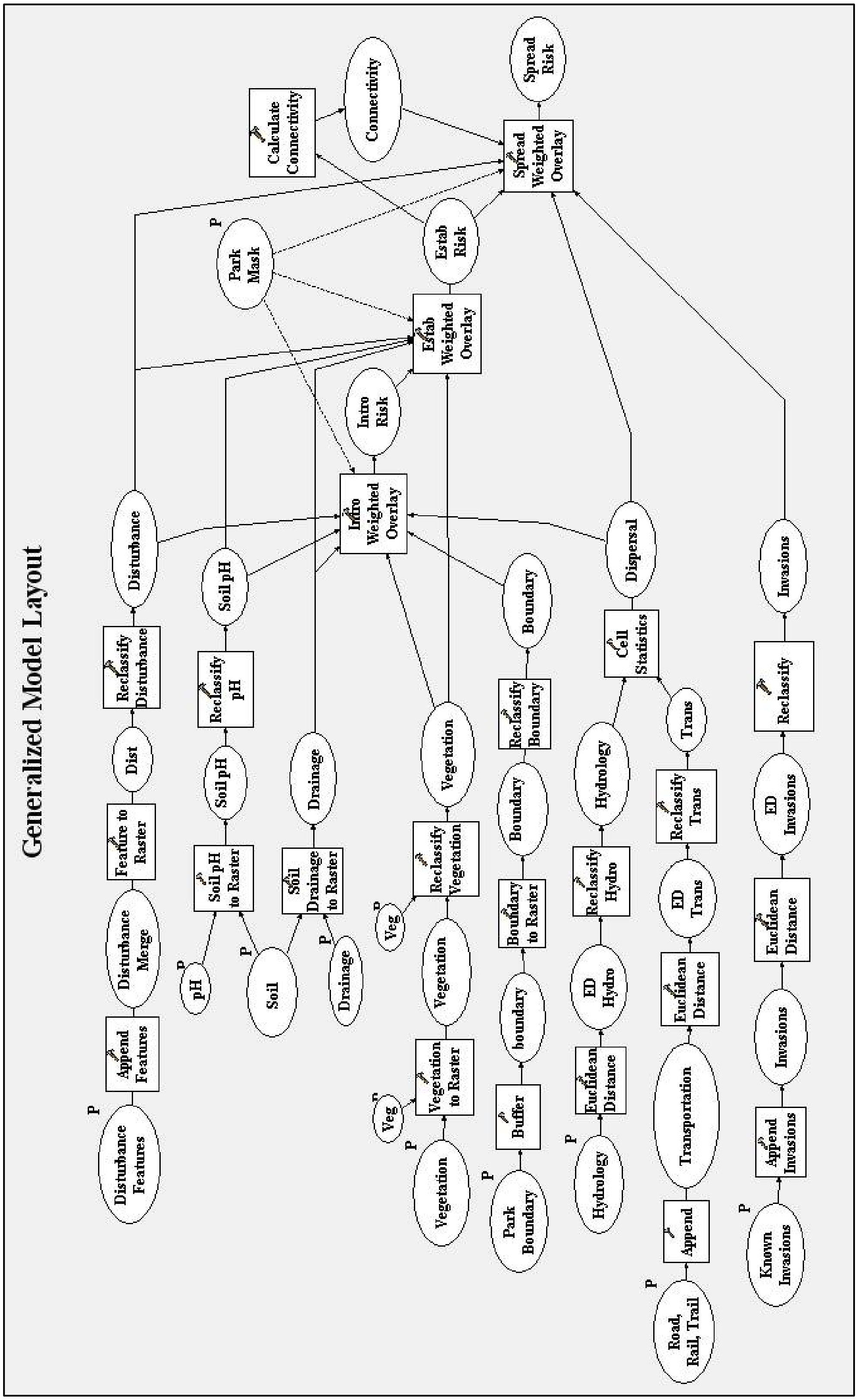


Figure 3. Generalized model layout for the multi-criteria risk models.

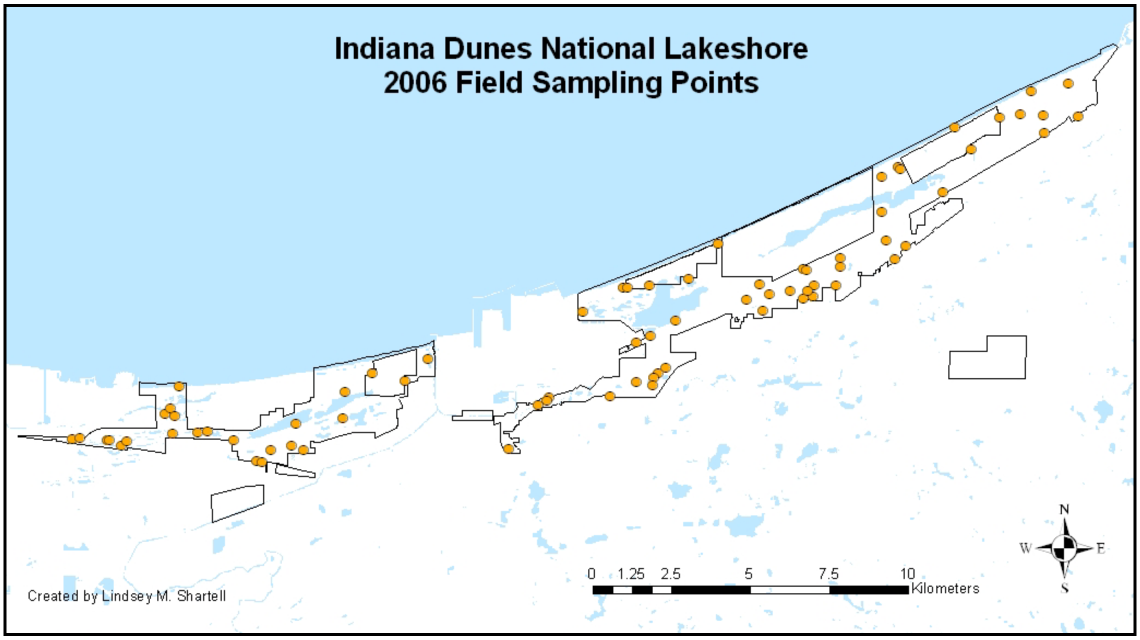


Figure 4. The location of field sampling points at Indiana Dunes National Lakeshore during summer 2006.

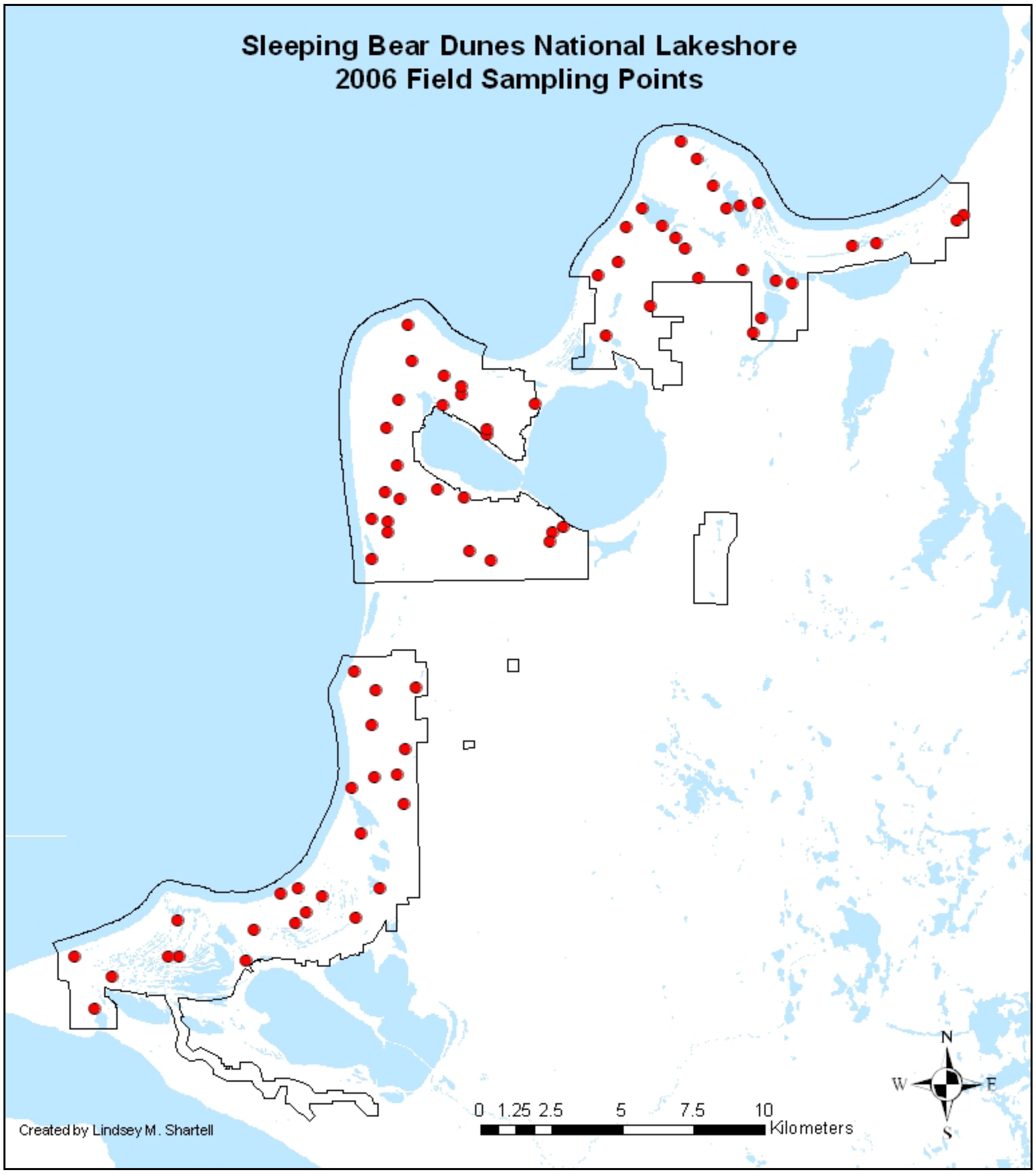


Figure 5. The location of field sampling points at Sleeping Bear Dunes National Lakeshore during summer 2006.

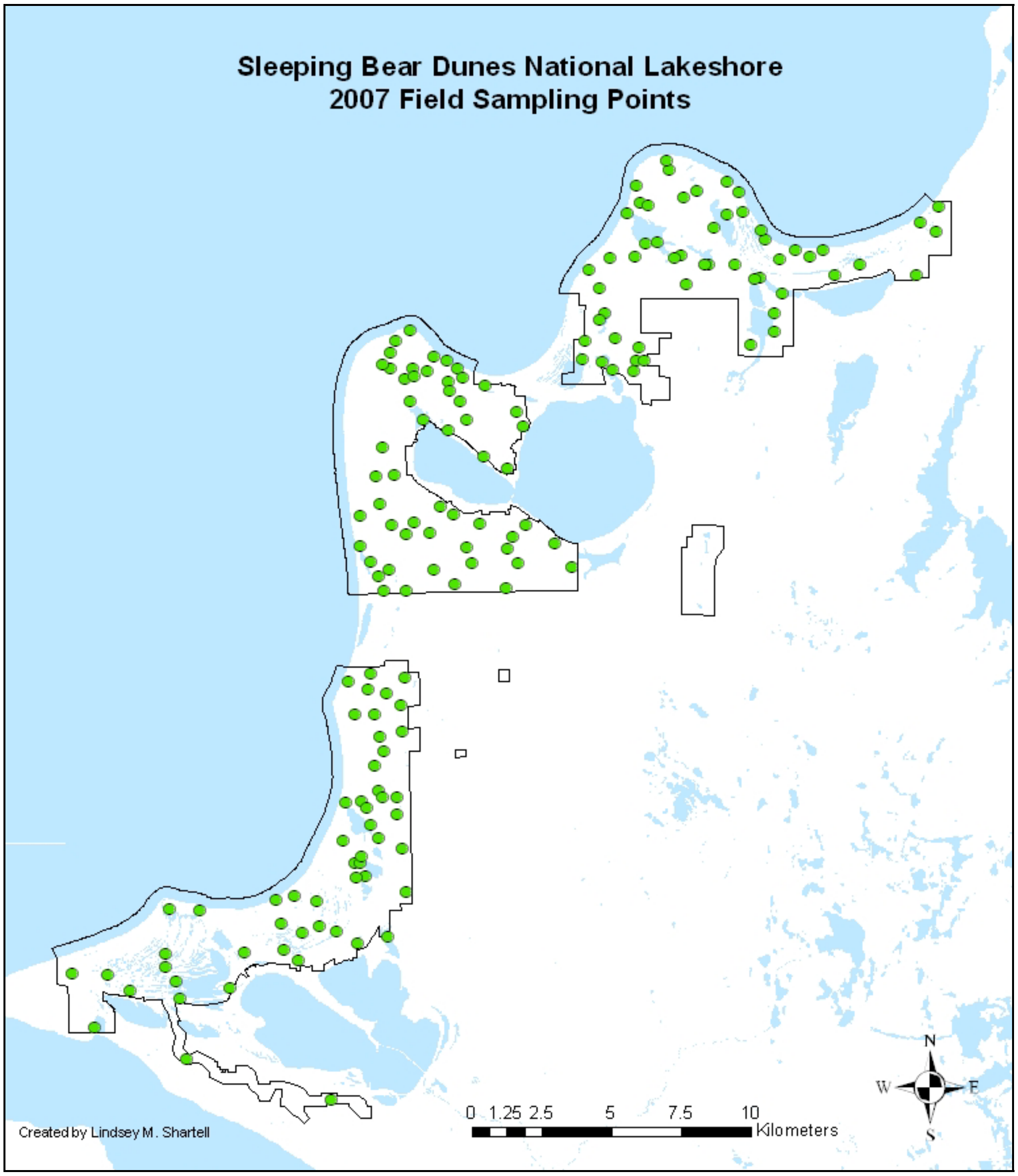


Figure 6. The location of field sampling points at Sleeping Bear Dunes National Lakeshore during summer 2007.

Appendix 1. Characteristics of the ten target invasive plants.

Species	**Description**
Baby's Breath *Gypsophila paniculata*	A perennial herb that invades shores, dunes, fields, sandy roadsides, railroad embankments, and ditches. Often found in disturbed areas. The plants are dome-shaped, branched and bushy, with tiny flowers (Voss 1985). Native to Europe and Asia, it likely became invasive once it escaped from horticulture.
Common Buckthorn *Rhamnus cathartica*	A shrub or small tree that invades woodlands and open areas. Native to Europe and Asia, it was introduced for use as an ornamental shrub, fencerows, and wildlife habitat. It has prolific seed production and can form dense stands that prevent native tree and shrub regeneration. It can survive in a broad range of soil and light conditions.
Common Reed *Phragmites* spp.	A tall wetland grass. Some genotypes are native, but the aggressive invaders are thought to be exotic. It is found in wetlands as well as along the edges of ponds, lakes, and streams, and along roadsides in drainage ditches. It is a strong competitor and often crowds out other plants. The rapid expansion of populations may be associated with disturbance or environmental stress.
Garlic Mustard *Alliaria petiolata*	A biennial herb with heart-shaped, coarsely toothed leaves, white flowers, and seeds in slender pods. It was first documented in the United States in 1868. It is native to northern Europe, but is now widely distributed across the eastern and central United States. It invades woodlands, roadsides, and urban areas, and is promoted by disturbances. Where established, it can eliminate native vegetation and impact ecosystem function.
Glossy Buckthorn *Frangula alnus*	A small tree or tall shrub. It is identified by its glossy dark green leaves and gray bark. Native to Europe and Asia (Voss 1985). It is highly invasive, invades natural habitats, and replaces native species (USFS 1998). Glossy buckthorn invades wetlands and, when cut, resprouts vigorously from the stump. It also consumes a large amount of water, which can lower the water table significantly (Devine 1999).

Species	**Description**
Honeysuckle *Lonicera* spp.	A deciduous shrub identified by its egg shaped leaves, white to pink flowers, and presence of a hollow stem. Native to Eurasia, it was first collected in the Midwest in the 1890s (Voss 1996). It can be found in woodlands, open areas, and roadsides. Some species can also be found in wetland habitats. Honeysuckle competes with native plants by decreasing light, moisture, and nutrient availability. It can release a toxic chemical that prevents other plant growth.
Leafy Spurge *Euphorbia esula*	A perennial herb with small greenish-yellow flowers. Native to Europe and Asia, it was brought to the United States in the late 1890s in impure seed. It is most aggressive in dry soils but can survive in moist soils as well. It invades fields, grasslands, roadsides, and woodlands. It displaces native vegetation and can produce plant toxins that prevent the growth of other plants. The stems and leaves contain a latex that is toxic to most grazing mammals and can irritate the skin of animals and humans if touched.
Multiflora Rose *Rosa multiflora*	A perennial shrub from the rose family. Native to eastern Asia, it introduced for use as a living fence and for food and cover for wildlife. It is identified by its arching canes and clusters of flowers ranging in color from white to pink. It can invade woodlands, fields, roadsides, and some wetland habitats. As it grows, it crowds out native plants and can create an impenetrable wall. Due to its tolerance for a variety of conditions, as well as its production of up to a million seeds per year, it spreads easily (Petrella et al. 1999). It is associated with disturbed areas.
Purple Loosestrife *Lythrum salicaria*	An herbaceous perennial with showy purple flowers. It was first introduced in the early 1800s. It invades wetland habitats and moist roadsides. Invaded wetlands often lose 50% of native plant biomass, and in extreme cases can be completely covered (Van Driesche 2002). It often outcompetes endangered, threatened, or declining plant species, and can also alter food and cover, resulting in the reduction of vertebrate and invertebrate populations (Van Driesche 2002). It is associated with disturbance and can be transported by water, wind, animals, and humans.

Species	Description
Spotted Knapweed *Centaurea biebersteinii*	A perennial herb native to Eastern Europe. It invades grasslands, woodlands, roadsides, and open sites. It is most competitive in dry sunny sites. It produces an allelopathic compound that reduces the growth of other plants. It can crowd out native plants and cover entire areas. Grazing animals will not eat it, but will instead feed on the native plants reducing their presence further. It has also been found to degrade soil over time by removing much of the moisture and nutrients.

Appendix 2. Bibliography used for literature review during model development.

Baby's Breath (*Gypsophila paniculata*)

California Department of Food and Agriculture. Weed Management - Baby's Breath [*Gypsophila paniculata* L. var. paniculata]. http://www.cdfa.ca.gov/phpps/ipc/weedinfo/gypsophi la-paniculata.htm

Darwent, A. L. 1975. The Biology of Canadian Weeds. *Gypsophila paniculata* L. Can J Plant Sci 55:1049-58.

Darwent, A.L. and R.T. Coupland. 1966. Life History of *Gypsophila paniculata*. Weeds 14:313-18.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Common Buckthorn (*Rhamnus cathartica*)

Archibold, O. W., D. Brooks, and L. Delanoy. 1997. An investigation of the invasive shrub European buckthorn, *Rhamnus cathartica* L., near Saskatoon, Saskatchewan. Can Field Nat 111:617-621.

Boudreau, D. 1992. Buckthorn research and control at Pipestone National Monument (Minnesota). Restor Manage Notes 10:94-95.

Converse, C. K. 1984. Element stewardship abstract for *Rhamnus cathartica, Rhamnus frangula* (syn. *Frangula alnus*). The Nature Conservancy.

Gourley, L. C. and E. Howell. 1984. Factors in buckthorn invasion documented; control measure checked (Wisconsin). Restoration and Management Notes 2:87.

Heidorn, R. 1991. Vegetation management guideline: exotic buskthorns – common buckthorn (*Rhamnus cathartica*), glossy buckthorn (*Rhamnus frangula* L.), dahurian buckthorn (*Rhamnus davurica* Pall.). Nat Areas J 11:216-217.

Hiawatha National Forest. 2005. Non-native invasive plants species invasiveness rank form. Common and Glossy Buckthorn (*Rhamnus cathartica* and *Frangula alnus*). US Department of Agriculture.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Common Reed (*Phragmites* spp.)

Blossey, B., M. Schwarzlander, P. Hafliger, R. Casagrande, and L. Tewksbury. 2002. Common Reed *In:* Driesche, R. V., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 131-138. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Coops, H. and G. Van Der Velde. 1995. Seed dispersal, germination and seedling growth of six helophyte species in relation to water-level zonation. Freshwater Biol 34:13-20.

Haslam, S. M. 1971. The development and establishment of young plants of *Phragmites communis* Trin. Ann Bot 35:1059-1072.

Havens, K. J., W. I. Priest, and H. Berquist. 1997. Investigation and long-term monitoring of *Phragmites australis* within Virginia's constructed wetland sites. Environ Manage 21:599-605.

Appendix 2. Cont.

Common Reed Cont.

Hellings, S. E. and J. L. Gallagher. 1992. The effects of salinity and flooding on *Phragmites australis*. J App Ecol 29:41-49.

Marks, M., B. Lapin, and J. Randall. 1994. Phragmites australis (P. communis): Threats, management, and monitoring. Nat Areas J 14:285-294.

McNabb, C. D. and T. R. Batterson. 1991. Occurrence of the common reed, *Phragmites australis*, along roadsides in Lower Michigan. Mich Academician 23:211-220.

Rickey, M. A. and R. C. Anderson. 2004. Effects of nitrogen addition on the invasive grass *Phragmites australis* and a native competitor *Spartina pectinata*. J App Ecol 41:888-896.

Silliman, B. R. and M. D. Bertness. 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of plant diversity on New England salt marshes. Conserv Biol 18:1424-1434.

US Department of Agriculture. 2007. Plants Database. http://plants.usda.gov.

Weisner, S. E. B., W. Graneli, and B. Ekstam. 1993. Influence of submergence on growth of seedlings of *Scirpus lacustris* and *Phragmites australis*. Freshwater Biol 29:371-375.

Garlic Mustard (*Alliaria petiolata*)

Anderson, R. C., S. S. Dhillion, and T. M. Kelley. 1996. Aspects of the ecology of an invasive plant, garlic mustard (*Alliaria petiolata*), in central Illinois. Restor Ecol 4:181-191.

Baskin, J. M. and C. C. Baskin. 1992. Seed germination biology of the weedy biennial *Alliaria petiolata*. Nat Areas J 12:191-197.

Blossey, B., V. Nuzzo, H. Hinz, and E. Gerber. 2001. Developing biological control of *Alliaria petiolata* (M. Bieb.) Cavara and Grande (garlic mustard). Nat Areas J 21:357-367.

---------- 2002. Garlic Mustard *In:* Van Driesche, R., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 365-372. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Byers, D. L. and J. A. Quinn. 1998. Demographic variation in *Alliaria petiolata* (Brassicaceae) in four contrasting habitats. J Torrey Bot Soc 125:138-149.

Cavers, P. B., M. I. Heagy, and R. F. Kokron. 1979. The biology of Canadian weeds. 35. *Alliaria petiolata* (M. Bieb.) Cavara and Grande. Can J Plant Sci 59:217-229.

Cruden, R. W., A. M. McClain, and G. P. Shrivastava. 1996. Pollination and breeding system of *Alliaria petiolata* (Brassicaceae). Bull Torrey Bot Club 123:273-280.

Drayton, B. and R. B. Primack. 1999. Experimental extinction of garlic mustard (*Alliaria petiolata*) populations: implications for weed science and conservation biology. Biol Invasions 1:159-167.

Luken, J. O. and M. Shea. 2000. Repeated prescribed burning at Dinsmore Woods State Nature Preserve (Kentucky, USA): Responses of the understory community. Nat Areas J 20:150-158.

Appendix 2. Cont.

Garlic Mustard Cont.

Lund, J. M. 2005. Garlic mustard (*Alliaria petiolata*) germination at different pH levels and detection and control in the Upper Peninsula of Michigan. M.S. Thesis, Michigan Technological University.

Mackenzie, S. J. B. 1995. Response of garlic mustard (*Alliaria petiolata* (M. Bieb.) Cavara and Grande) seeds and first year plants to cold, heat, and drought. M.S. Thesis, Wright State University.

Meekins, J. F. and B. C. McCarthy. 1999. Competitive ability of *Alliaria petiolata* (garlic mustard, Brassicaceae), an invasive, nonindigenous forest herb. Int J Plant Sci 160:743-752.

---------- 2000. Responses of the biennial forest herb *Alliaria petiolata* to variation in population density, nutrient additions, and light availability. J Ecol 88:447-463.

---------- 2002. Effect of population density on the demography of an invasive plant (*Alliaria petiolata*, Brassicaceae) population in a southeastern Ohio forest. Am Midl Nat 147:256-278.

Myers, C. V. and R. C. Anderson. 2003. Seasonal variation in photosynthetic rates influences success of an invasive plant, garlic mustard (*Alliaria petiolata*). Am Midl Nat 150:231-245.

Myers, C. V., R. C. Anderson, and D. L. Byers. 2005. Influence of shading on the growth and leaf photosynthesis of the invasive non-indigenous plant garlic mustard [*Alliaria petiolata* (M. Bieb) Cavara and Grande] grown under simulated late-winter to mid-spring conditions. J Torrey Bot Soc 132:1-10.

Nuzzo, V. A. 1991. Experimental control of garlic mustard [*Alliaria petiolata* (Bieb.) Cavara and Grande] in northern Illinois using fire, herbicide, and cutting. Nat Areas J 11:158-167.

---------- 1993. Distribution and spread of the invasive biennial *Alliaria petiolata* (garlic mustard) in North America *In:* McKnight, B. N. (ed.). *Biological Pollution: The Control and Impact of Invasive Exotic Species*. pp 137-146. Indiana Academy of Science, Indianapolis.

---------- 1999. Invasion pattern of the herb garlic mustard (*Alliaria petiolata*) in high quality forests. Biol Invasions 1:169-179.

Peterson, A. T., M. Papes, and D. A. Kluza. 2003. Predicting the potential invasive distribution of four alien plant species in North America. Weed Sci 51:863-868.

Roberts, H. A. and J. E. Boddrell. 1983. Seed survival and periodicity of seedling emergence in eight species of Cruciferae. Ann Appl Biol 103:301-304.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Welk, E., K. Schubert, and M. H. Hoffmann. 2002. Present and potential distribution of invasive garlic mustard (*Alliaria petiolata*) in North America. Divers Distrib 8:219-233.

Appendix 2. Cont.

Glossy Buckthorn (*Frangula alnus*)

Catling, P. M. and Z. S. Porebski. 1994. The history of invasion and current status of glossy buckthorn, *Rhamnus frangula*, in southern Ontario. Can Field Nat 108:305-310.

Converse, C. K. 1984. Element stewardship abstract for *Rhamnus cathartica, Rhamnus frangula* (syn. *Frangula alnus*). The Nature Conservancy.

Frappier, B. and R. T. Eckert. 2003. Utilizing the USDA PLANTS database to predict exotic woody plant invasiveness in New Hampshire. Forest Ecol Manag 185:207-215.

Frappier, B., R. T. Eckert, and T. D. Lee. 2003. Potential impacts of the invasive exotic shrub *Rhamnus frangula* L. (glossy buckthorn) on forests of southern New Hampshire. Northeast Nat 10:277-296.

Frappier, B., T. D. Lee, K. F. Olson, and R. T. Eckert. 2003. Small-scale invasion pattern, spread rate, and lag-phase behavior of *Rhamnus frangula* L. Forest Ecol Manag 186:1-6.

Heidorn, R. 1991. Vegetation management guideline: exotic buskthorns – common buckthorn (*Rhamnus cathartica*), glossy buckthorn (*Rhamnus frangula* L.), dahurian buckthorn (*Rhamnus davurica* Pall.). Nat Areas J 11:216-217.

Hiawatha National Forest. 2005. Non-native invasive plants species invasiveness rank form. Common and Glossy Buckthorn (*Rhamnus cathartica* and *Frangula alnus*). US Department of Agriculture.

Howell, J. A. and W. H. Blackwell, Jr. 1977. The history of *Rhamnus frangula* (glossy buckthorn) in the Ohio flora. Castanea 42:111-115.

Possessky, S. L., C. E. Williams, and W. J. Moriarity. 2000. Glossy buckthorn *Rhamnus frangula* L.: a threat to riparian plant communities of the northern Allegheny Plateau (USA). Nat Areas J 20:290-292.

Sanford, N. L., R. A. Harrington, and J. H. Fownes. 2003. Survival and growth of native and alien woody seedlings in open and understory environments. Forest Ecol Manag 183:377-385.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Honeysuckle *(Lonicera* spp.)

Hutchinson, T. F. and J. L. Vankat. 1998. Landscape structure and spread of the exotic shrub *Lonicera maackii* (amur honeysuckle) in Southwestern Ohio forests. Am Midl Nat 139:383-390.

US Department of Agriculture. 2007. Plants Database. http://plants.usda.gov.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Yates, E. D., D. F. Levia Jr., and C. L. Williams. 2004. Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. Forest Ecol Manag 190:119-130.

Appendix 2. Cont.

Leafy Spurge (*Euphorbia esula*)

Best, K. F., G. G. Bowes, A. G. Thomas, and M. G. Maw. 1980. The biology of Canadian weeds 39. *Euphorbia esula* L. Can J Plant Sci 60:651-663.

Lacey, J. R., R. Wallander, and K. Olson-Rutz. 1992. Recovery, germinability, and viability of leafy spurge (*Euphorbia esula*) seeds ingested by sheep and goats. Weed Technol 6:599-602.

Lym, R. G. 1998. The biology and integrated management of leafy spurge (*Euphorbia esula*) on North Dakota Rangeland. Weed Technol 12:367-373.

Maxwell, B. D., M. V. Wilson, and S. R. Radosevich. 1988. Population modeling approach for evaluating leafy spurge (*Euphorbia esula*) development and control. Weed Technol 2:132-183.

Nowierski, R. M. and R. W. Pemberton. 2002. Leafy Spurge *In:* Driesche, R. V., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 181-194. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Selleck, G. W., R. T. Coupland, and C. Frankton. 1962. Leafy spurge in Saskatchewan. Ecol Monographs 32:1-29.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Multiflora Rose (*Rosa multiflora*)

Amrine, J. W. 2002. Multiflora Rose *In:* Driesche, R. V., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 265-292. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Fawcett, R. S. 1980. Today's weed: multiflora rose. Weeds Today 11:22-23.

Schery, R. 1977. The curious double life of *Rosa multiflora*. Horticulture (6):56-61.

Steavenson, H. A. 1946. Multiflora rose for farm hedges. J Wildlife Manage 10:227-234.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Yates, E. D., D. F. Levia Jr., and C. L. Williams. 2004. Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. Forest Ecol. Manag. 190:119-130.

Purple Loosestrife (*Lythrum salicaria*)

Blossey, B. 2002. Purple Loosestrife *In:* Driesche, R. V., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 149-157. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Forman, R. T. T. and L. E. Alexander. 1998. Roads and their major ecological effects. Annu Rev Ecol Syst 29:207-231.

Hager, H. A. 2004. Differential effects of *Typha* litter and plants on invasive *Lythrum salicaria* seedling survival and growth. Biol Invasions 6:433-444.

Haworth-Brockman, M. J., H. R. Murkin, and R. T. Clay. 1993. Effects of shallow flooding on newly established purple loosestrife seedlings. Wetlands 13:224-227.

Appendix 2. Cont.

Purple Loosestrife Cont.

Hiawatha National Forest. 2005. Non-native invasive plants species invasiveness rank form. *Lythrum salicaria* (purple loosestrife). US Department of Agriculture.

Keddy, P. A. and T. H. Ellis. 1985. Seedling recruitment of 11 wetland plant species along a water level gradient: shared or distinct responses? Can J Bot 63:1876-1879.

Mullin, B. H. 1998. The biology and management of purple loosestrife (*Lythrum salicaria*). Weed Technol 12:397-401.

Nagel, J. M. and K. L. Griffin. 2001. Construction cost and invasive potential: comparing *Lythrum salicaria* (Lythraceae) with co-occurring native species along pond banks. Am J Bot 88:2252-2258.

Rawinski, T. J. and R. A. Malecki. 1984. Ecological relationships among purple loosestrife, cattail, and wildlife at the Montezuma National Wildlife Refuge. New York Fish and Game Journal 31:81-87.

Shadel, W. P. and J. Molofsky. 2003. Habitat and population effects on the germination and early survival of the invasive weed, *Lythrum salicaria* L. (purple loosestrife). Biol Invasions 4:413-423.

Shamsi, S. R. A. and F. H. Whitehead. 1974a. Comparative eco-physiology of *Epilobium hirsutum* L. and *Lythrum salicaria* L. I. General biology, distribution, and germination. J Ecol 62:279-290.

---------- 1974b. Comparative eco-physiology of *Epilobium hirsutum* L. and *Lythrum salicaria* L. II. Growth and development in relation to light. J Ecol 62:631-645.

---------- 1977. Comparative eco-physiology of *Epilobium hirsutum* L. and *Lythrum salicaria* L. IV. Effects of temperature and inter-specific competition and concluding discussion. J Ecol 65:71-84.

Stuckey, R. L. 1980. Distributional history of *Lythrum salicaria* (purple loosestrife) in North America. Bartonia 47:3-20.

Thompson, D. Q., R. L. Stuckey, and E. B. Thompson. 1987. Spread, impact, and control of purple loosestrife (*Lythrum salicaria*) in North American wetlands. U. S. Fish and Wildlife Service. 55 pages.

U. S. Department of Agriculture. Plant Guide: Purple Loosestrife. Plants Database. http://plants.usda.gov.

Uveges, J. L., A. L. Corbett, and T. K. Mal. 2002. Effects of lead contamination on the growth of *Lythrum salicaria* (purple loosestrife). Environ Pollut 120:319-323.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Weihe, P. E. and R. K. Neely. 1997. The effects of shading on competition between purple loosestrife and broad-leaved cattail. Aquat. Bot. 59:127-138.

Weiher, E., I. C. Wisheu, P. A. Keddy, and D. R. J. Moore. 1996. Establishment, persistence, and management implications of experimental wetland plant communities. Wetlands 16:208-218.

Appendix 2. Cont.

Purple Loosestrife Cont.

Welk, E. 2004. Constraints in range predictions of invasive plant species due to non-equilibrium distribution patterns: Purple loosestrife (*Lythrum salicaria*) in North America. Ecol Model 179:551-567.

Welling, C. H. and R. L. Becker. 1990. Seed bank dynamics of *Lythrum salicaria* L.: implications for control of this species in North America. Aquat Bot 38:303-309.

---------- 1993. Reduction of purple loosestrife establishment in Minnesota wetlands. Wildl Soc Bull 21:56-64.

Wilcox, D. A. 1989. Migration and control of purple loosestrife (*Lythrum salicaria*) along highway corridors. Environ Manage 13:365-370.

Yakimowski, S. B., H. A. Hager, and C. G. Eckert. 2005. Limits and effects of invasion by the nonindigenous wetland plant *Lythrum salicaria* (purple loosestrife): a seed bank analysis. Biol Invasions 7:687-698.

Young, J. A. and C. D. Clements. 2001. Purple loosestrife (*Lythrum salicaria*) seed germination. Weed Technol 15:337-342.

Spotted Knapweed (*Centaurea biebersteinii*)

Chicoine, T. K., P. K. Fay, and G. A. Nielsen. 1985. Predicting weed migration from soil and climate maps. Weed Sci 34:57-61.

Eddleman, L. E. and J. T. Romo. 1988. Spotted knapweed germination response to stratification, temperature, and water stress. Can J Bot 66:653-657.

Jacobs, J. S. and R. L. Sheley. 1998. Observation: life history of spotted knapweed. J Range Manage 51:665-673.

Kennett, G. A., J. R. Lacey, C. A. Butt, K. M. Olson-Rutz, and M. R. Haferkamp. 1992. Effects of defoliation, shading, and competition on spotted knapweed and bluebunch wheatgrass. J Range Manage 45:363-369.

Lacey, C. A., J. R. Lacey, P. K. Fay, J. M. Story, and D. L. Zamora. 1995. Controlling knapweed in Montana rangeland. Montana State University Cooperative Extension Service Circ 311. 17 p.

Pearson, D. E. and Y. K. Ortega. 2001. Evidence of an indirect dispersal pathway for spotted knapweed, *Centaurea maculosa*, seeds, via deer mice, *Peromyscus maniculatus*, and great horned owls, *Bubo virginianus*. Can Field-Nat 115:354.

Roze, L. D., B. D. Frazer, and A. Mclean. 1984. Establishment of diffuse and spotted knapweed from seed on disturbed ground in British Columbia, Canada. J Range Manage 37:501-502.

Schirman, R. 1981. Seed production and spring seedling establishment of diffuse and spotted knapweed. J Range Manage 34:45-47.

Sheley, R. L., J. S. Jacobs, and M. F. Carpinelli. 1998. Distribution, biology, and management of diffuse knapweed (*Centaurea diffusa*) and spotted knapweed (*Centaurea maculosa*). Weed Technol 12:353-362.

Sperber, T. D., J. M. Wraith, and B. E. Olson. 2003. Soil physical properties associated with the invasive spotted knapweed and native grasses are similar. Plant Soil 252:241-249.

Appendix 2. Cont.

Spotted Knapweed Cont.

Story, J. 2002. Spotted Knapweed *In:* Driesche, R. V., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 149-157. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

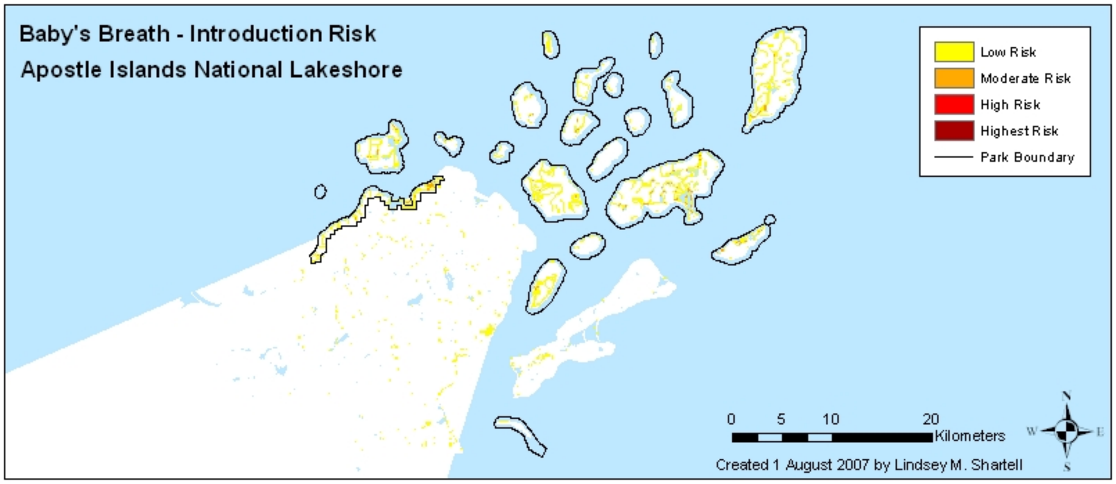
Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

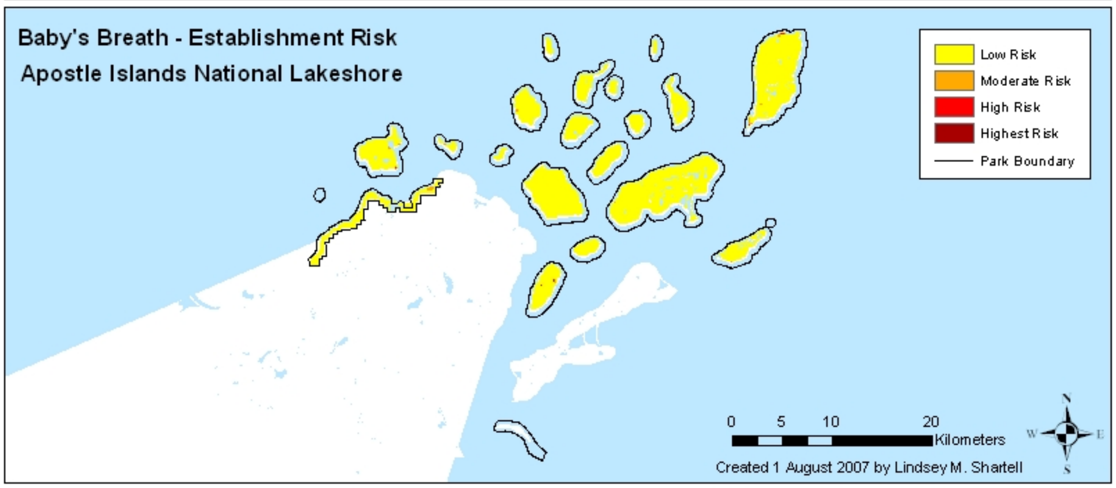
Wallander, R. T., B. E. Olson, and J. R. Lacey. 1995. Spotted knapweed seed viability after passing through sheep and mule deer. J Range Manage 48:145-149.

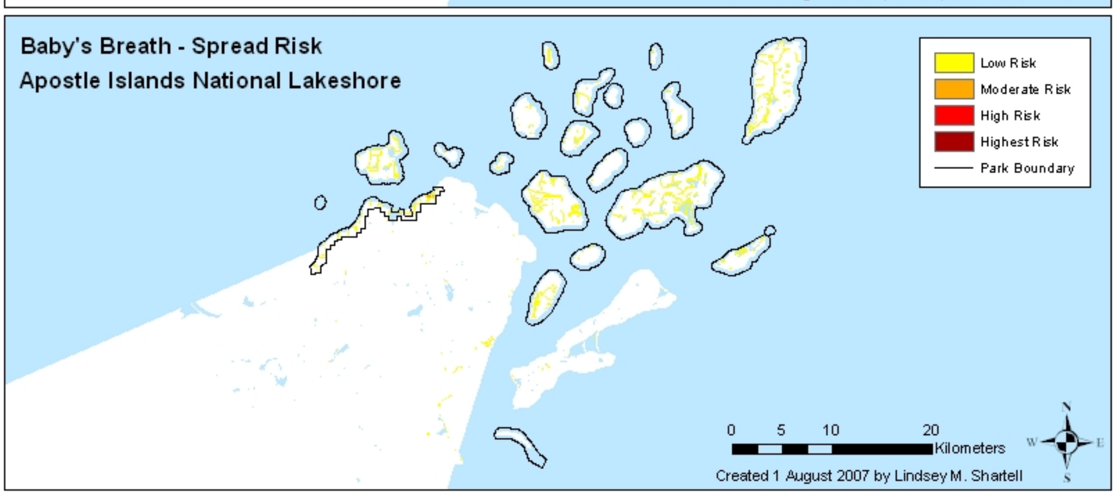
Watson, A. K. and A. J. Renney. 1974. The biology of Canadian weeds 6. *Centaurea diffusa and C. maculosa.* Can J Plant Sci 54:687-701.

Zouhar, Kris. 2001. Centaurea maculosa. *In:* Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory. http://www.fs.fed.us/database/feis/

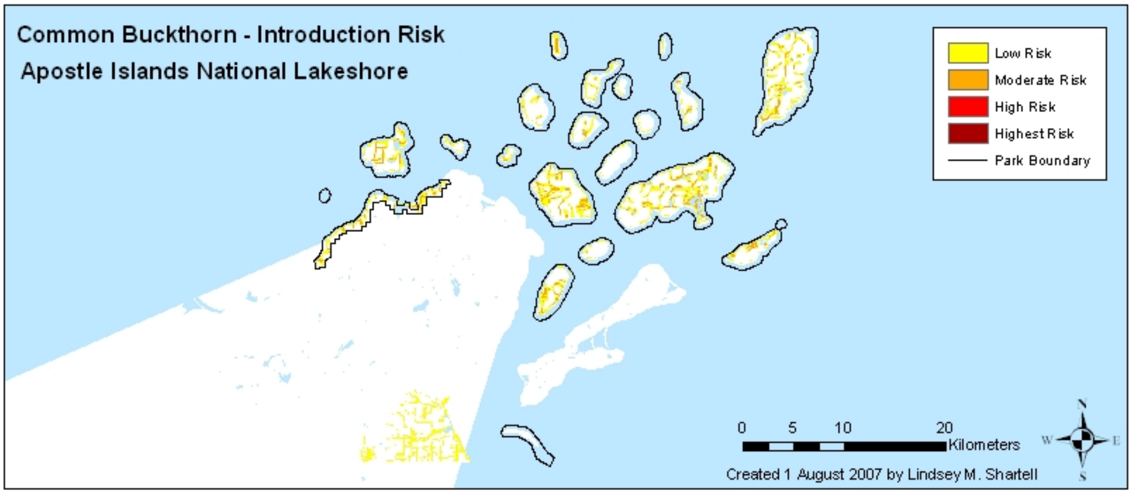
Appendix 3. Risks maps for the nine National Parks within the Great Lakes Network showing the predicted areas at risk, at three phases of invasion, for the ten target invasive plants.

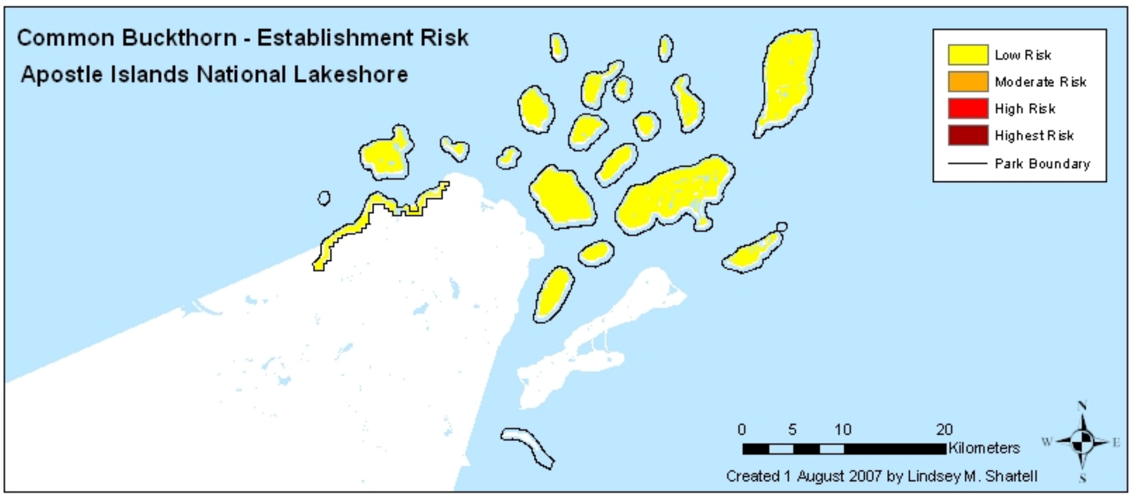


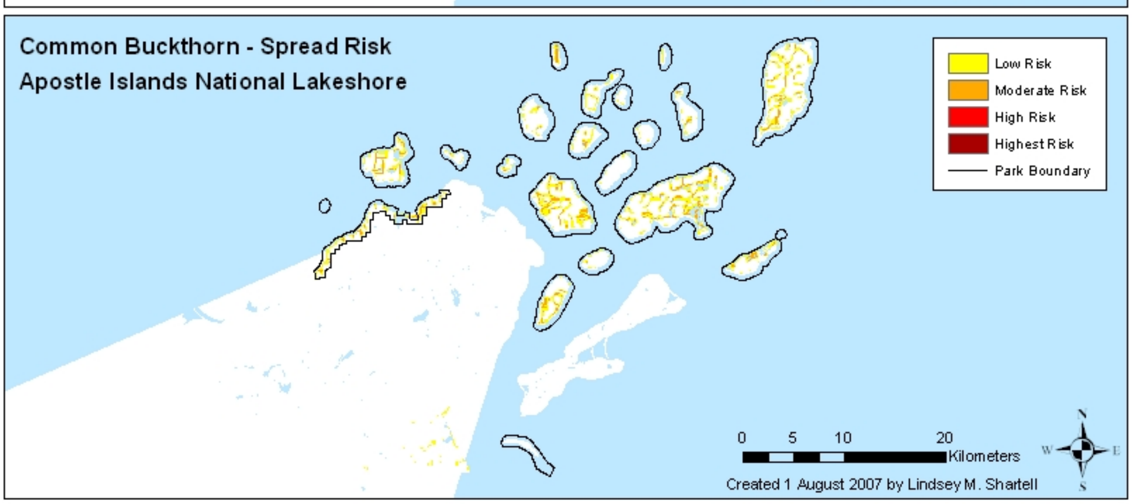




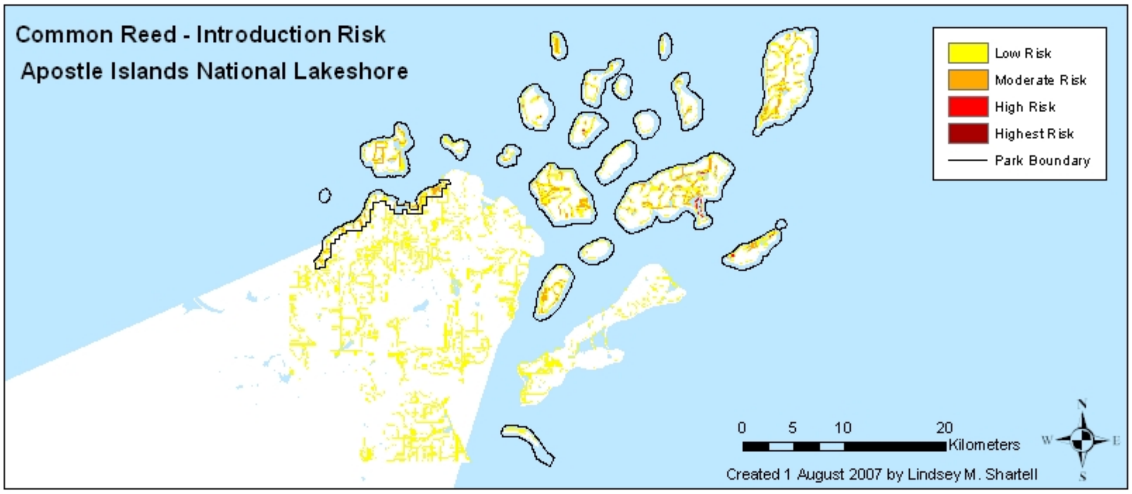
Appendix 3. Cont.

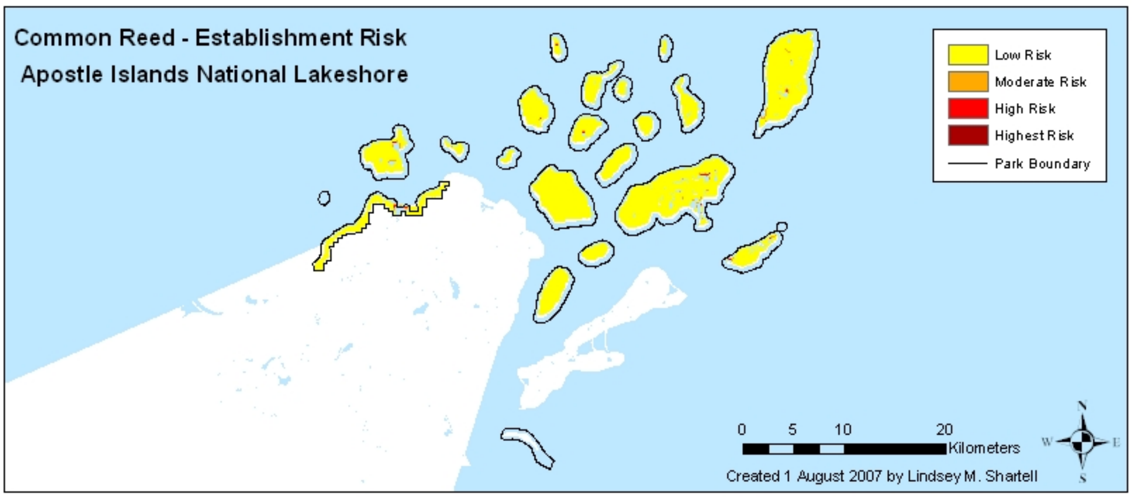


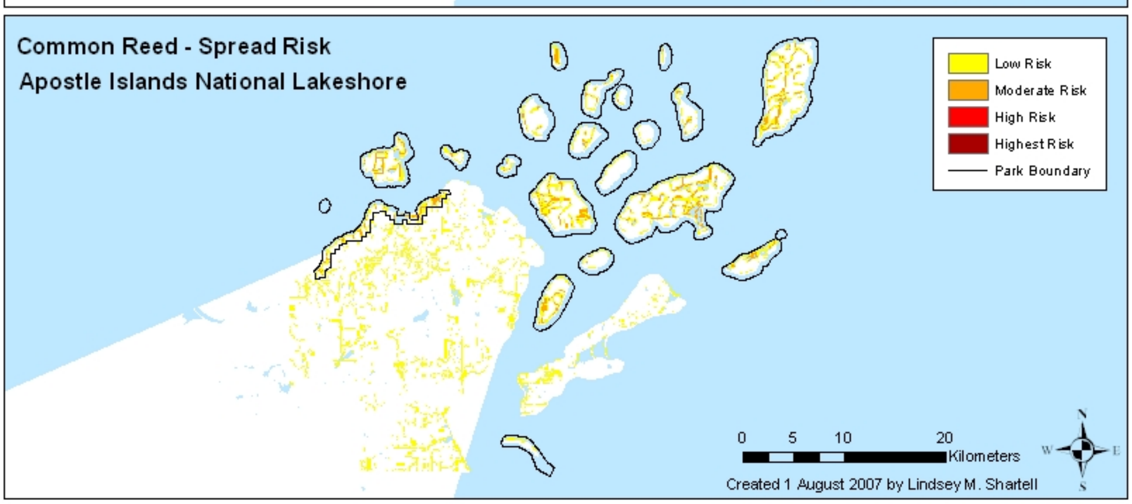




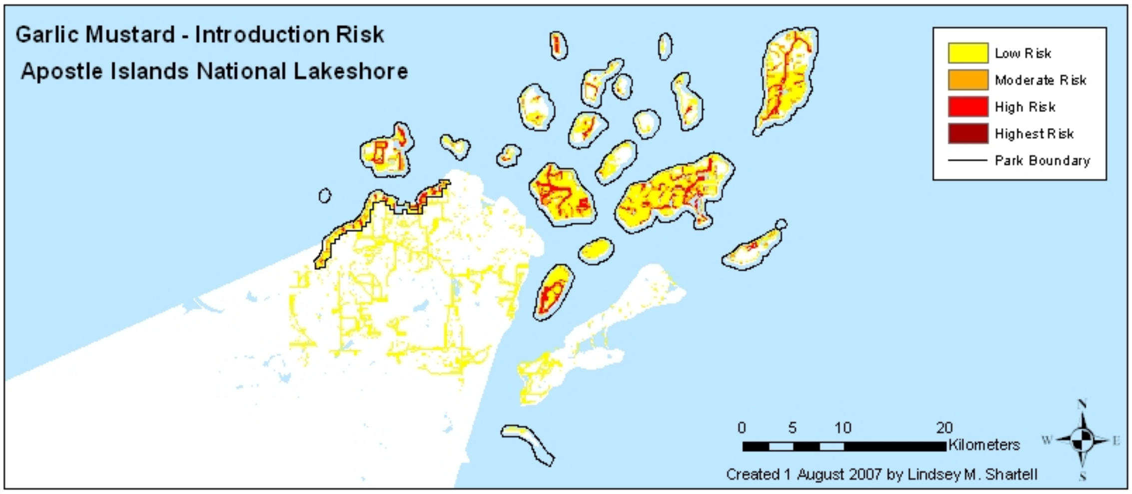
Appendix 3. Cont.

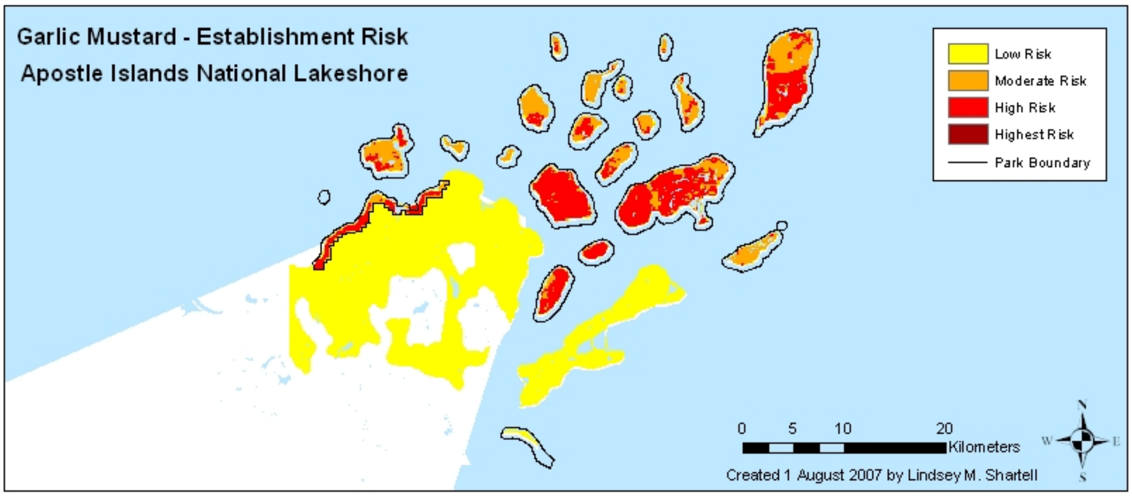


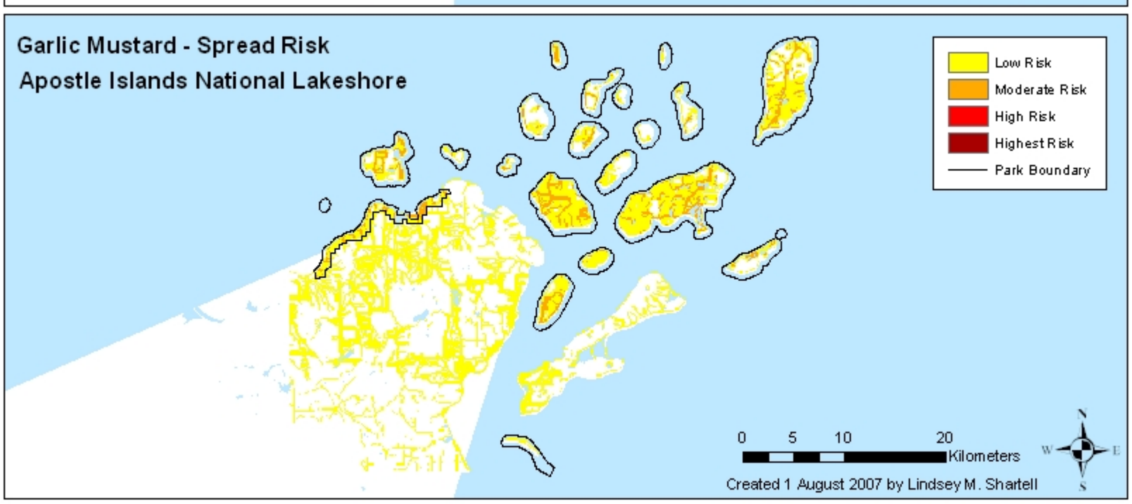




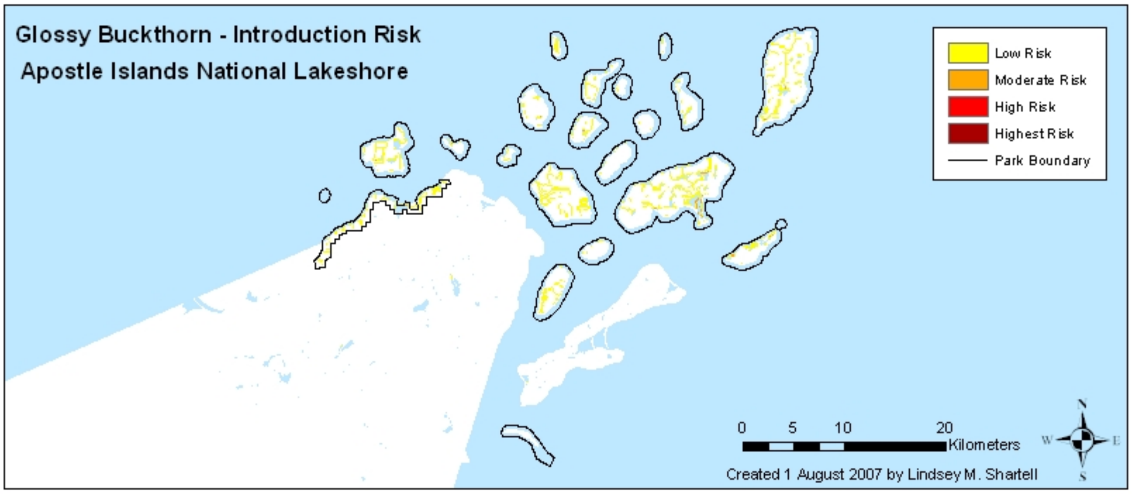
Appendix 3. Cont.

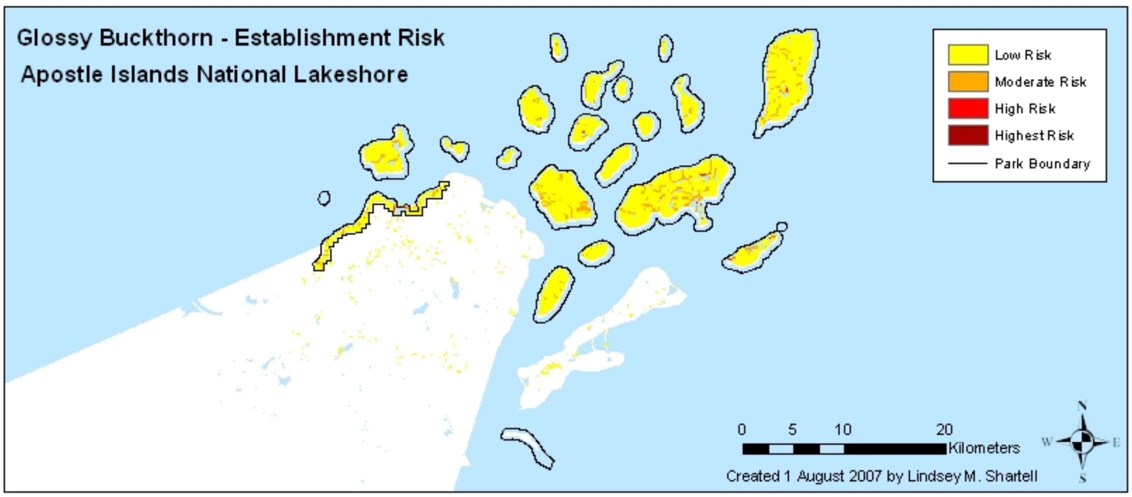


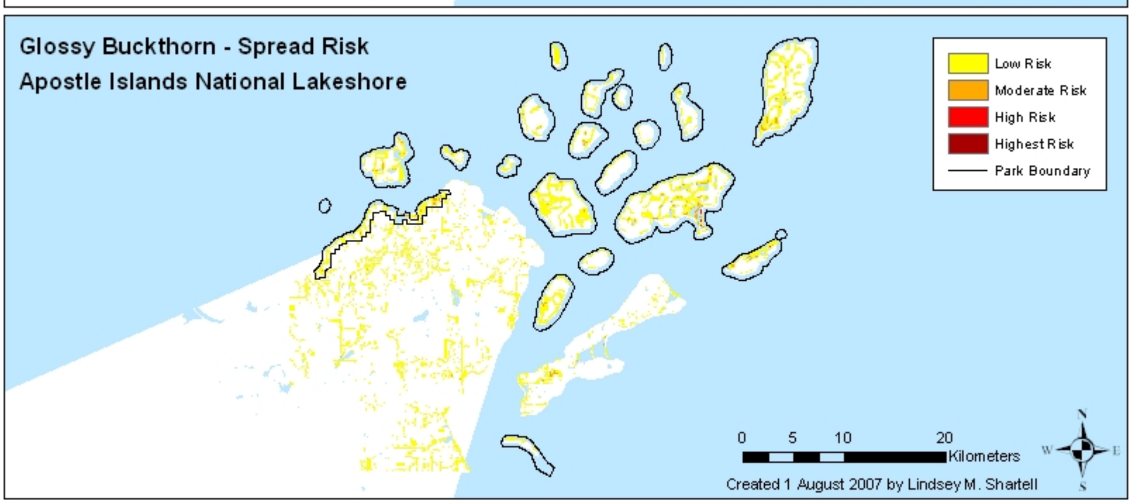




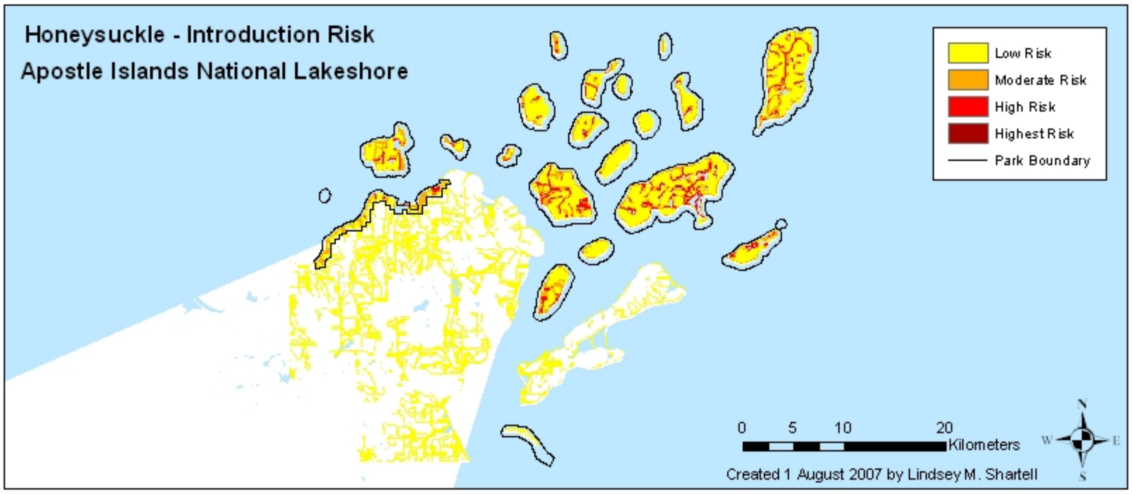
Appendix 3. Cont.

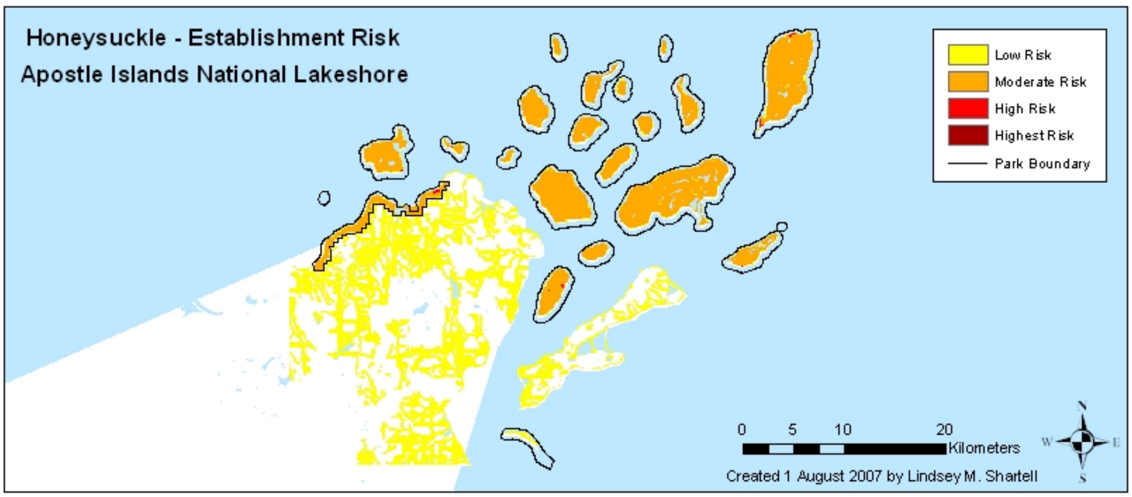


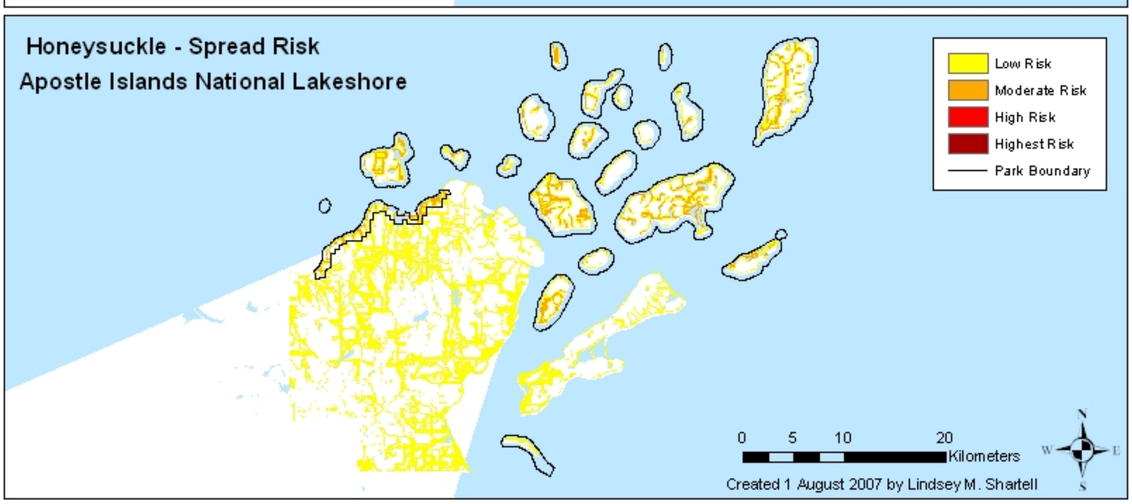




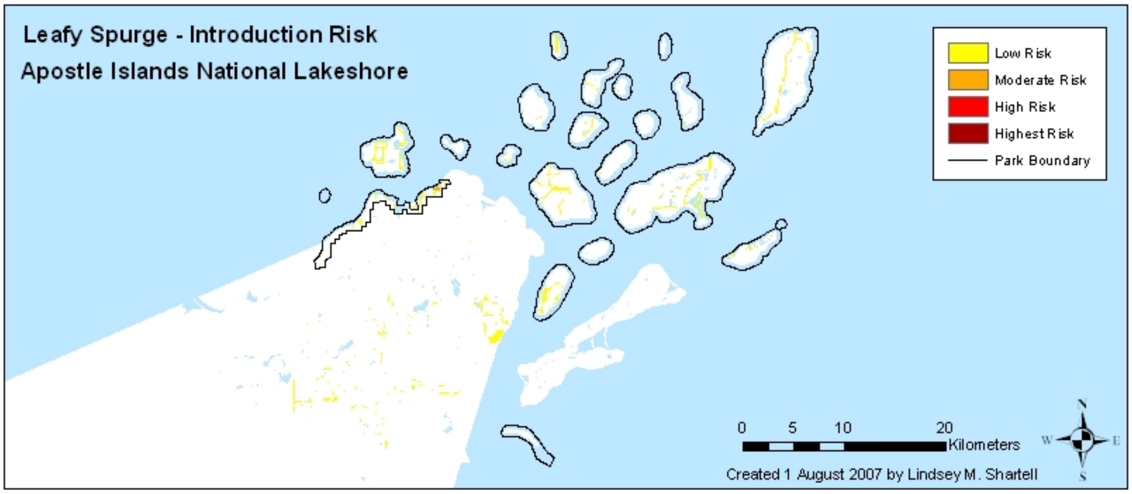
Appendix 3. Cont.

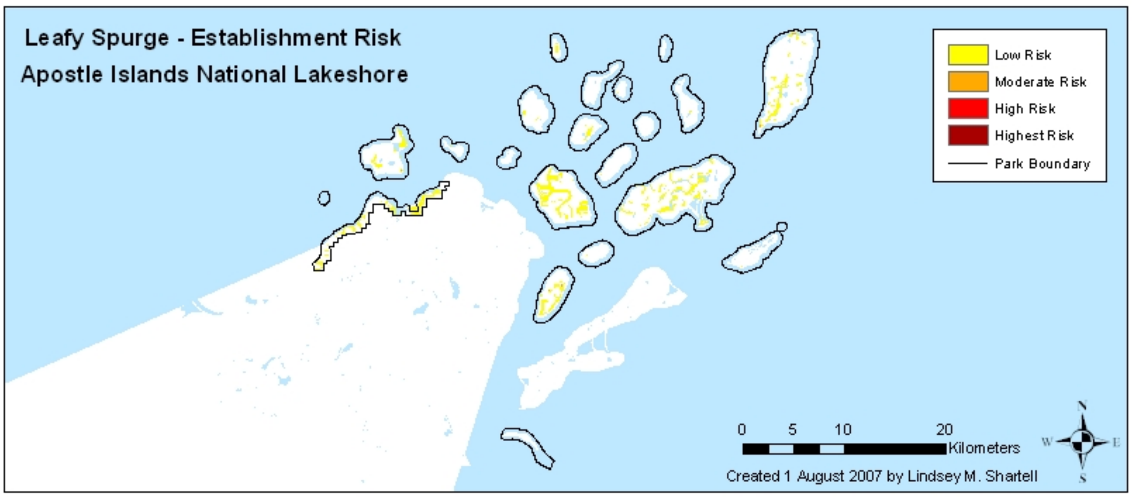


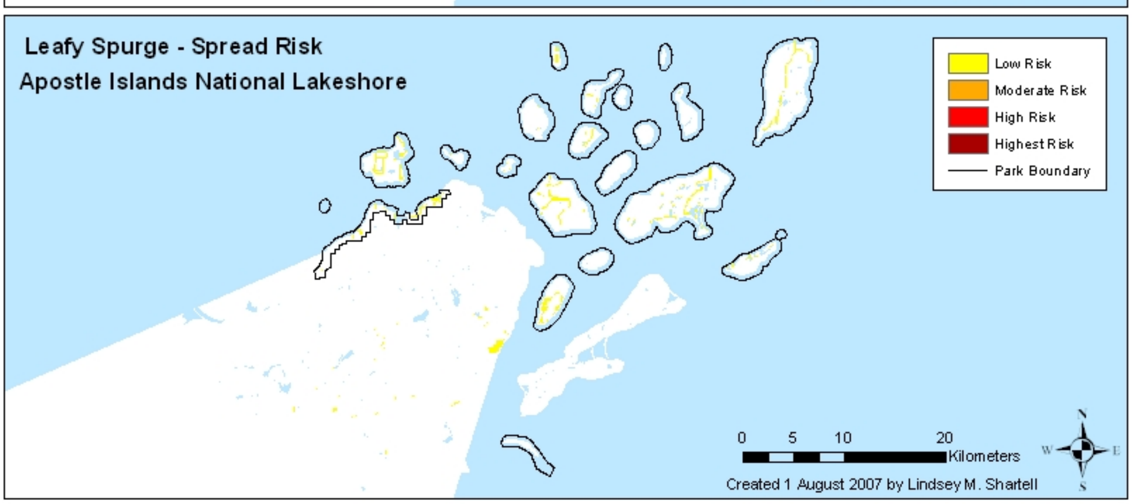




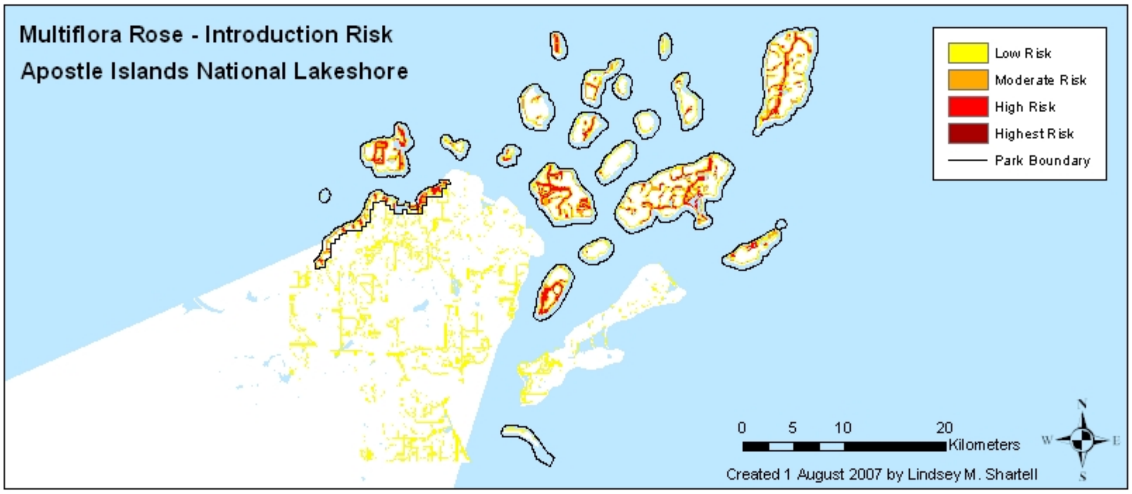
Appendix 3. Cont.

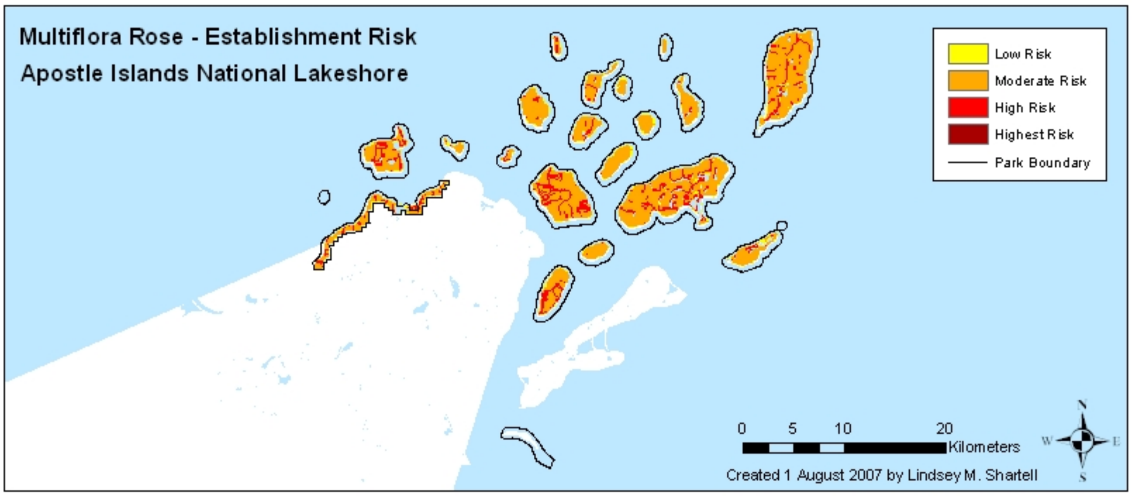


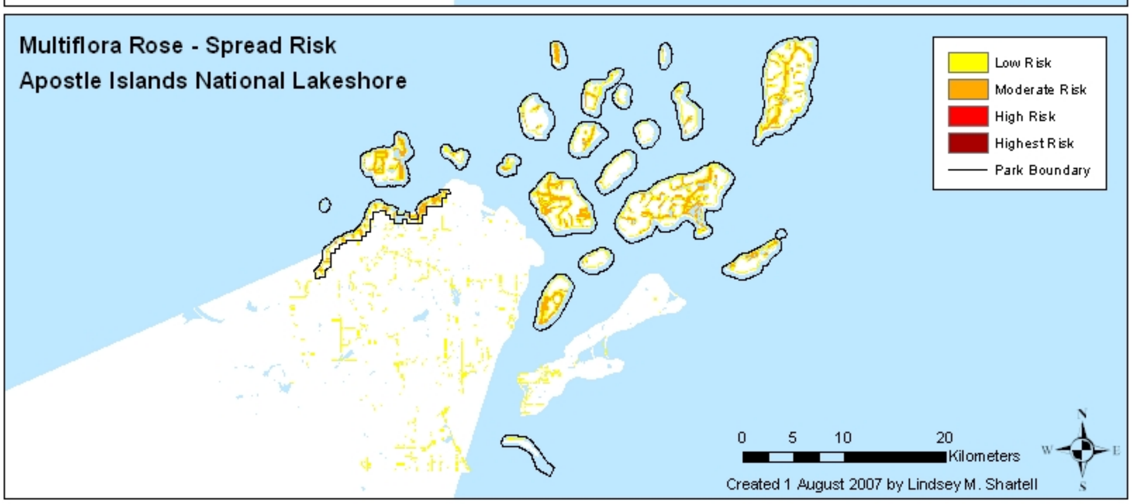




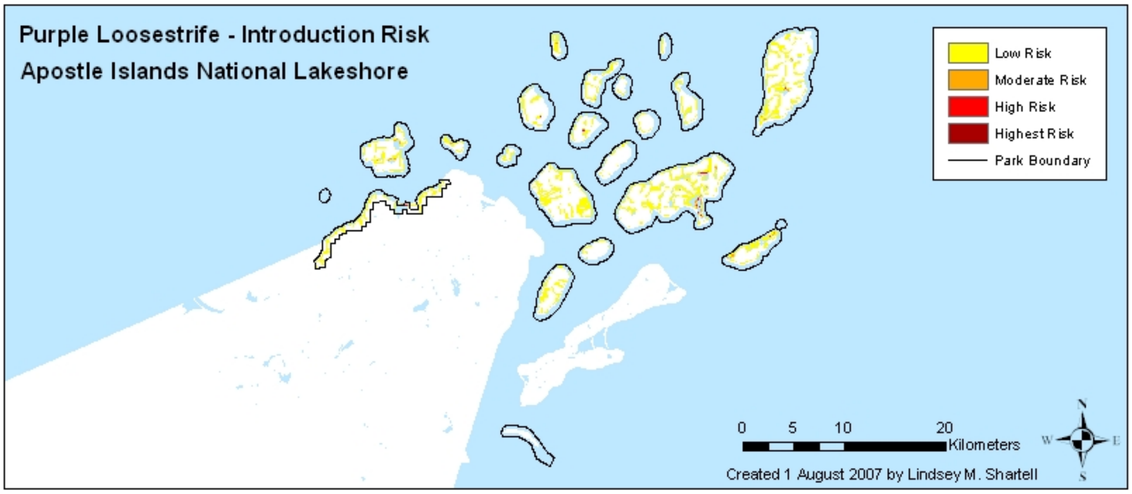
Appendix 3. Cont.

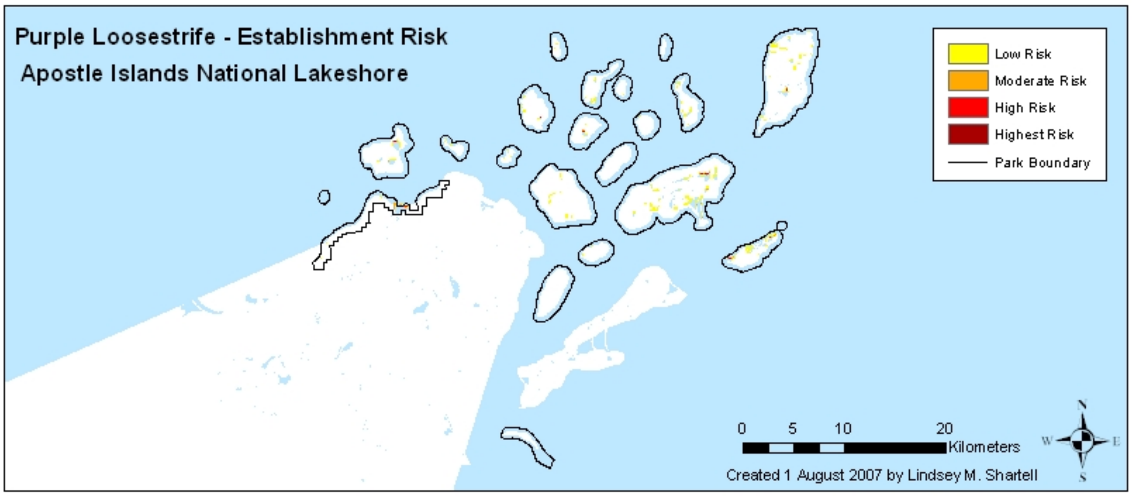


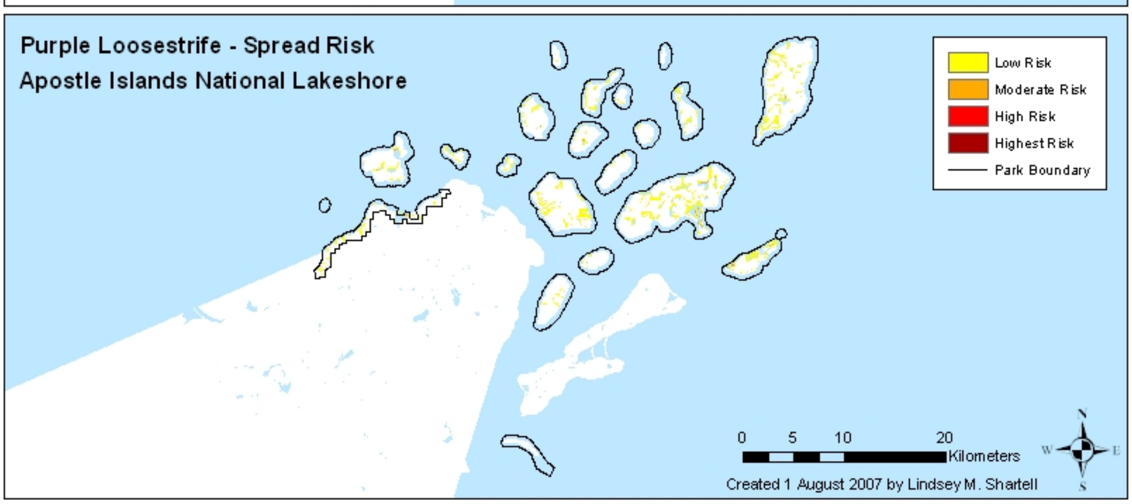




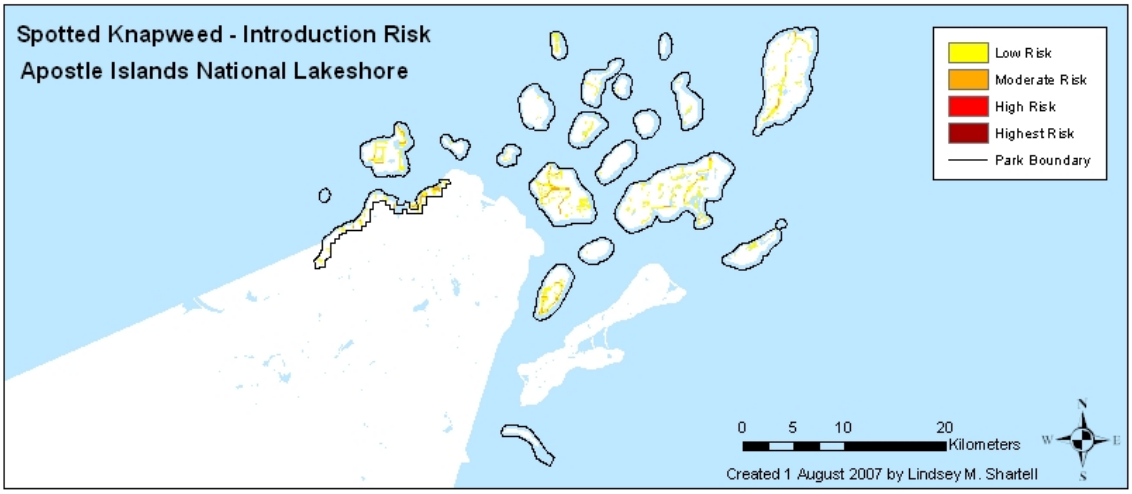
Appendix 3. Cont.

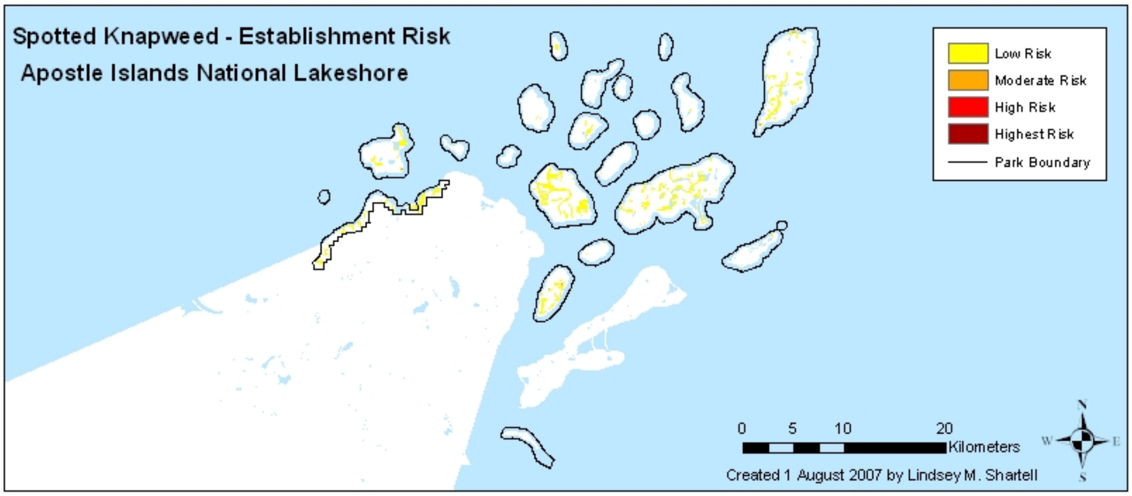


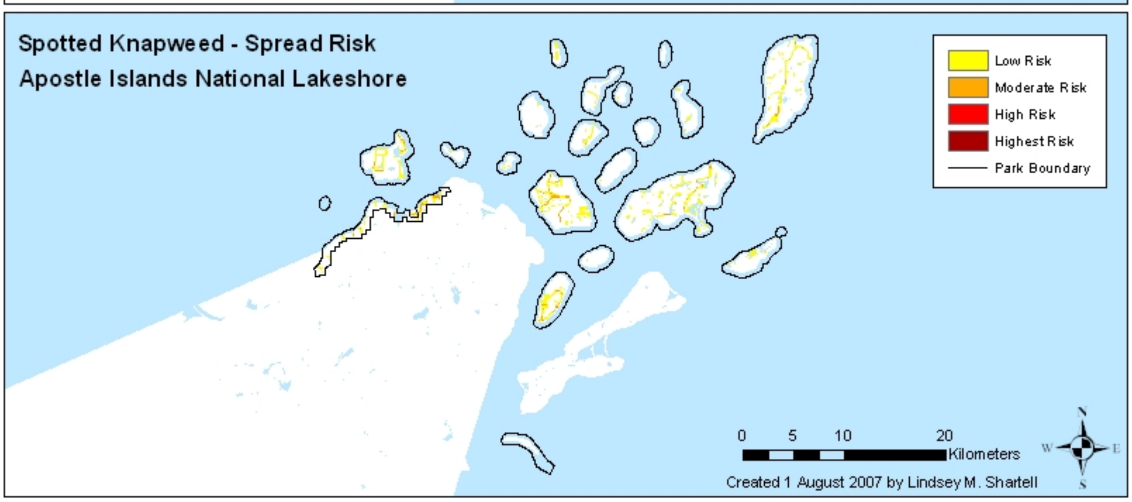




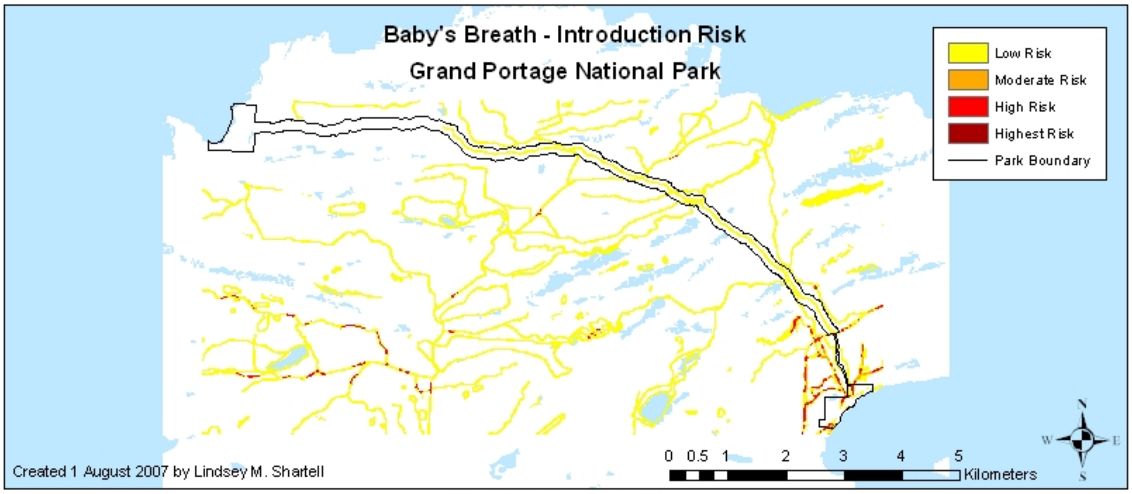
Appendix 3. Cont.

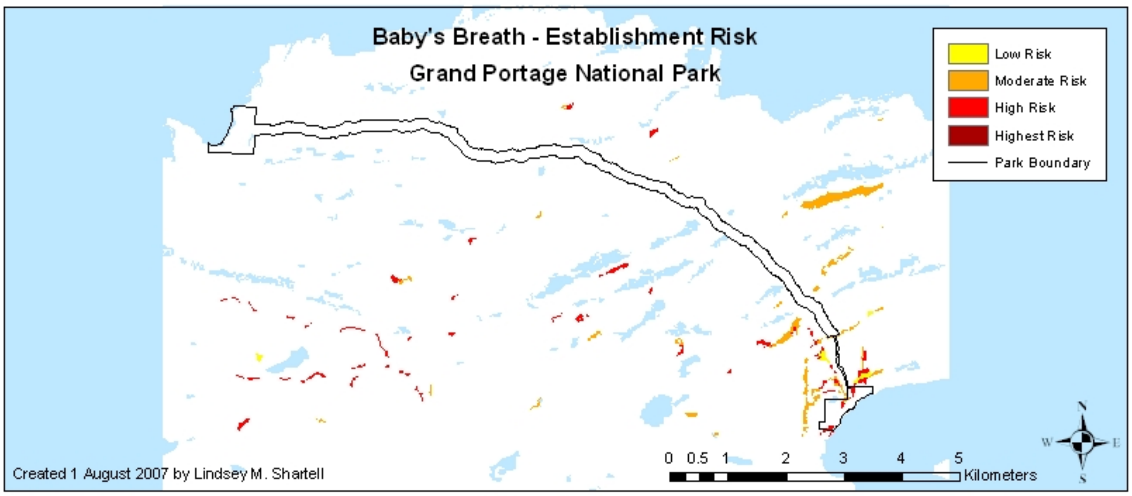


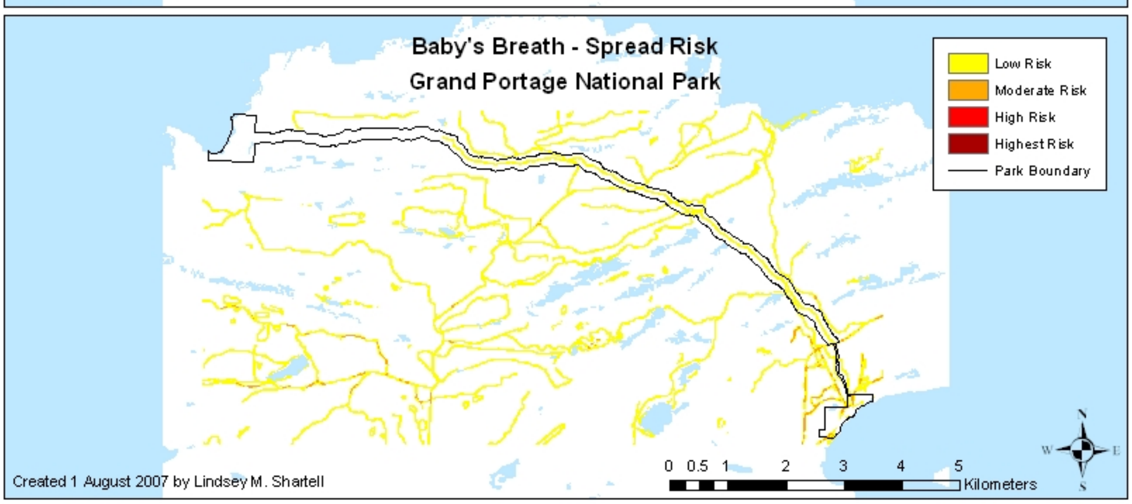




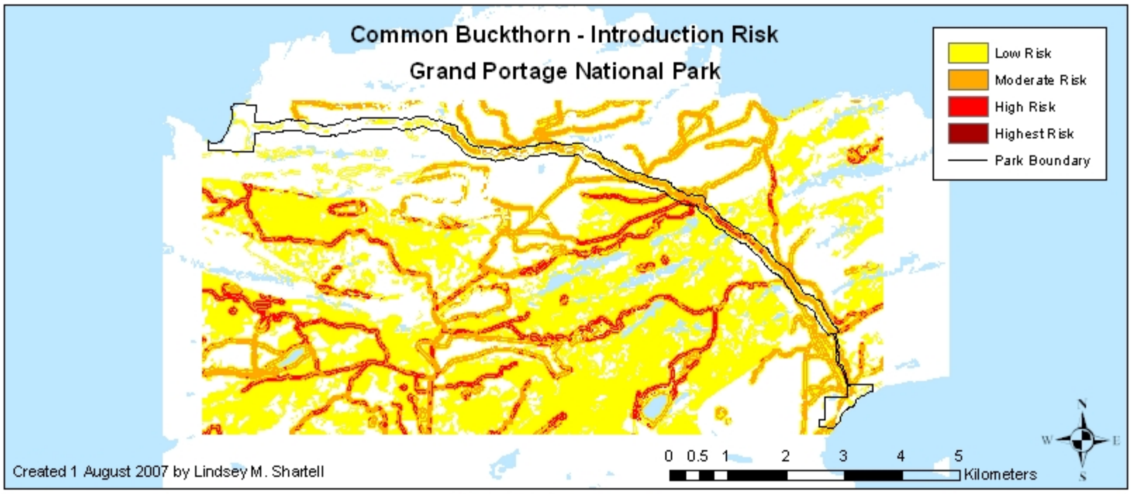
Appendix 3. Cont.

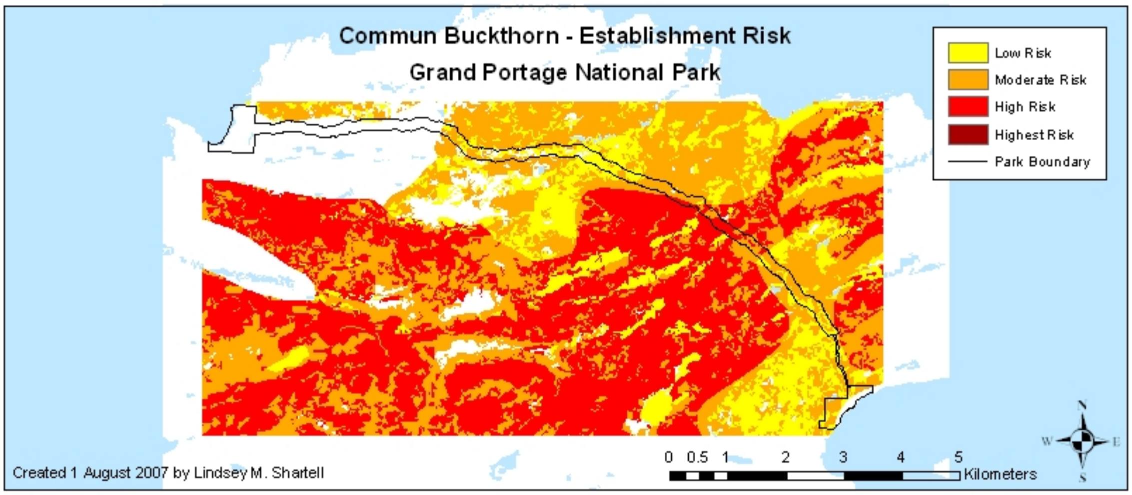


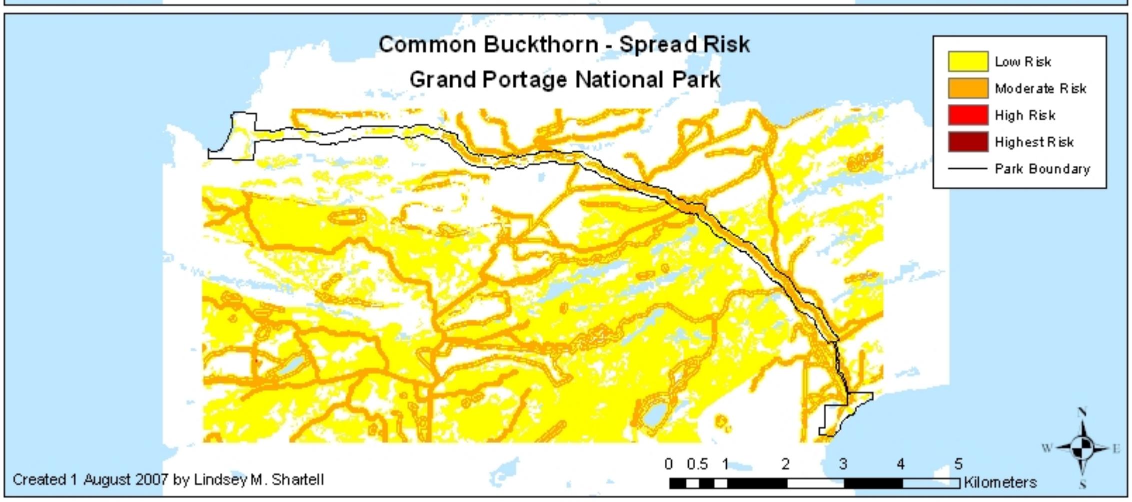




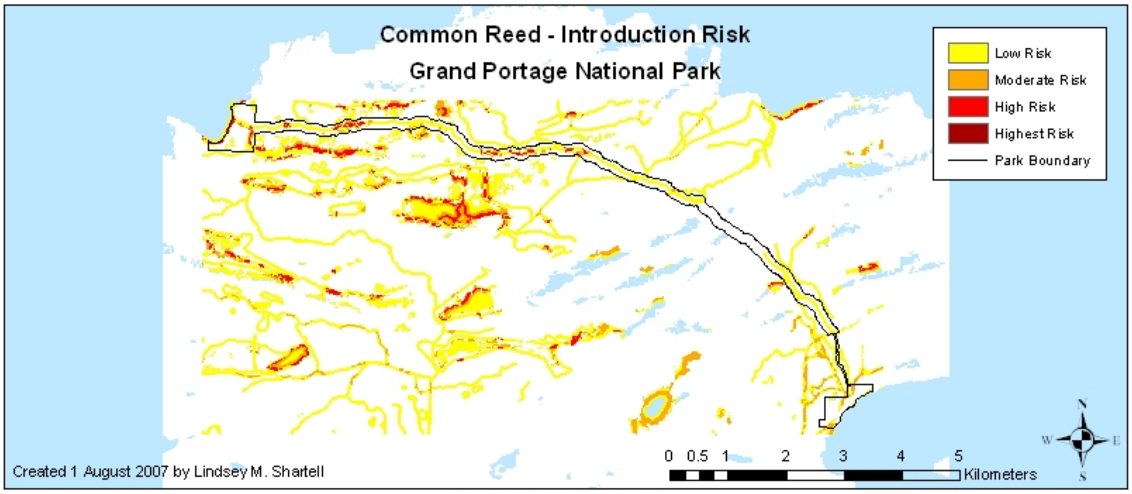
Appendix 3. Cont.

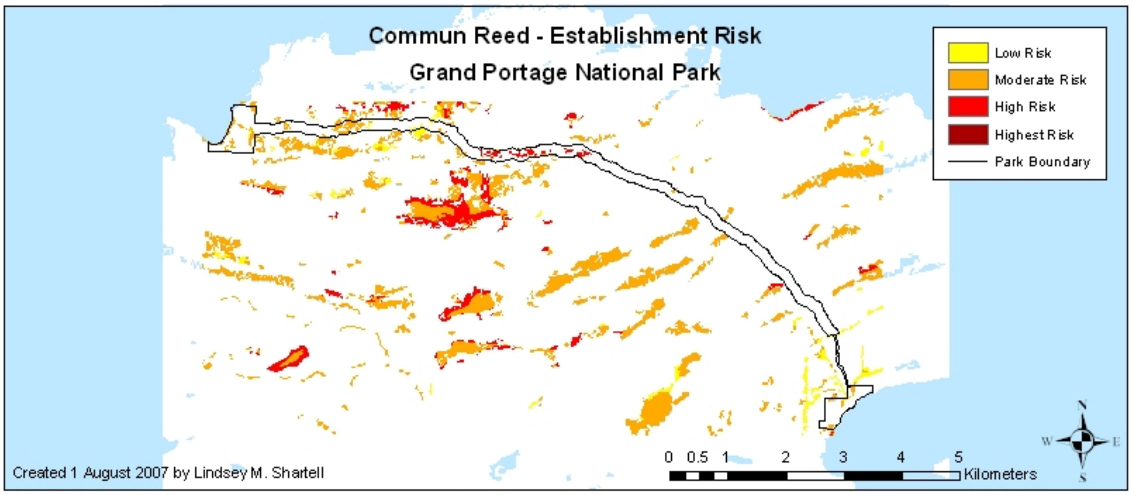


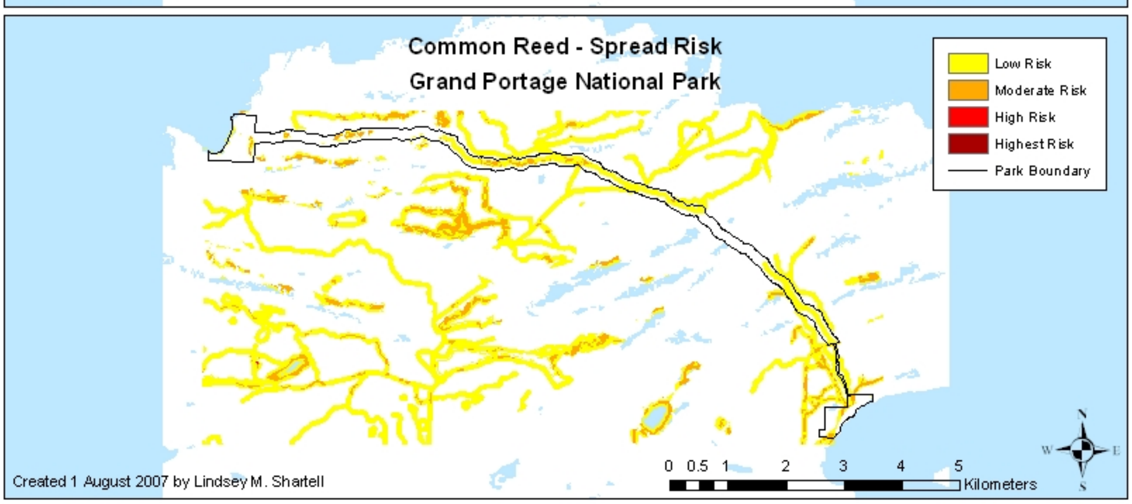




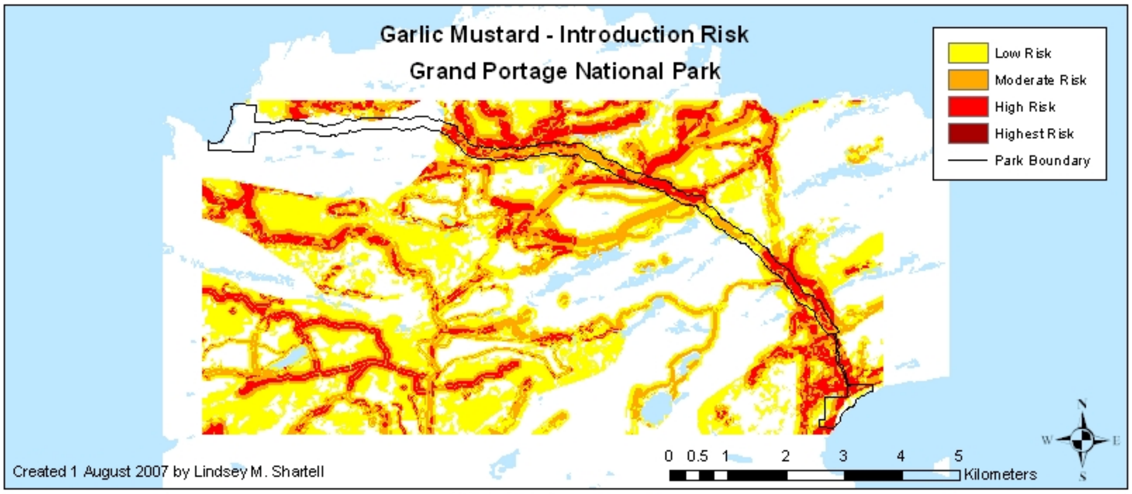
Appendix 3. Cont.

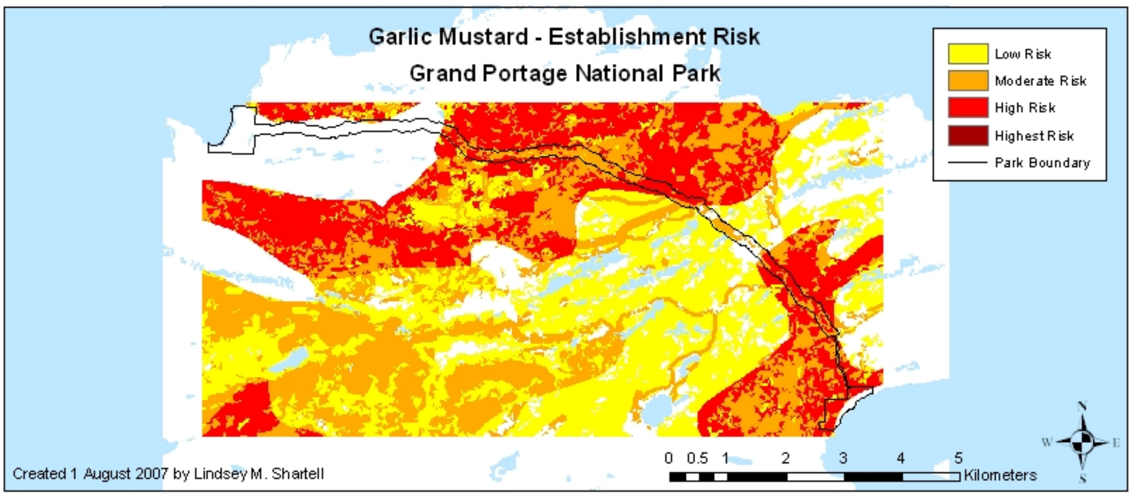


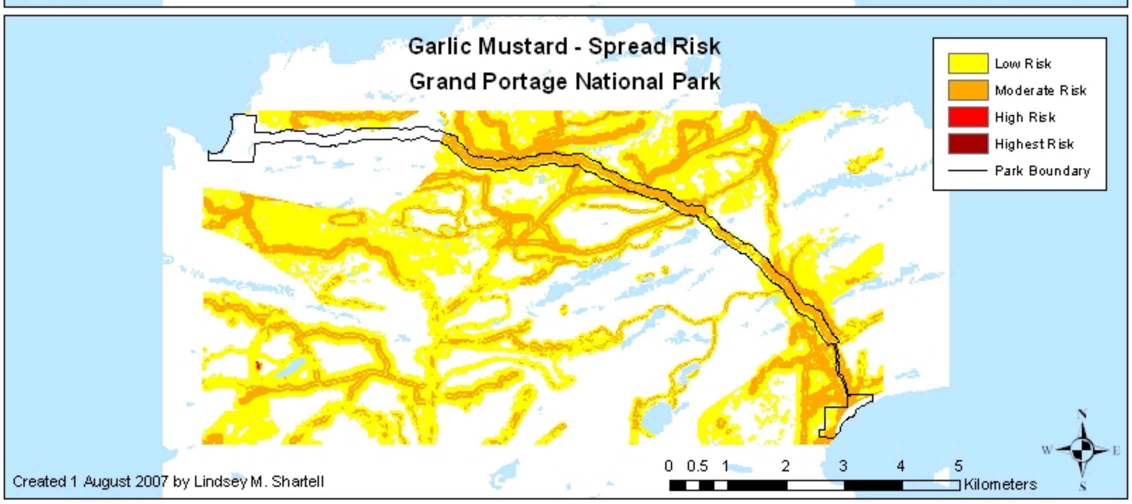




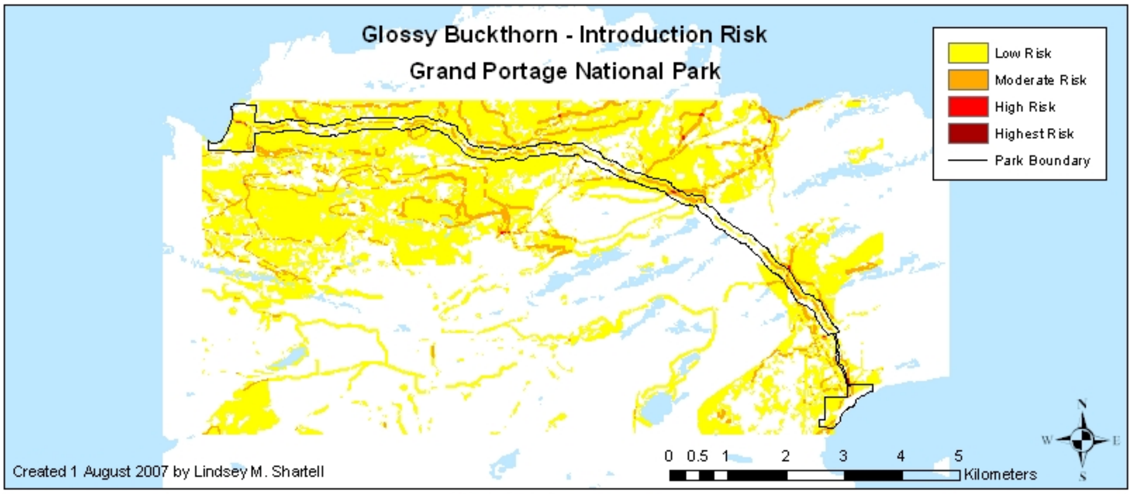
Appendix 3. Cont.

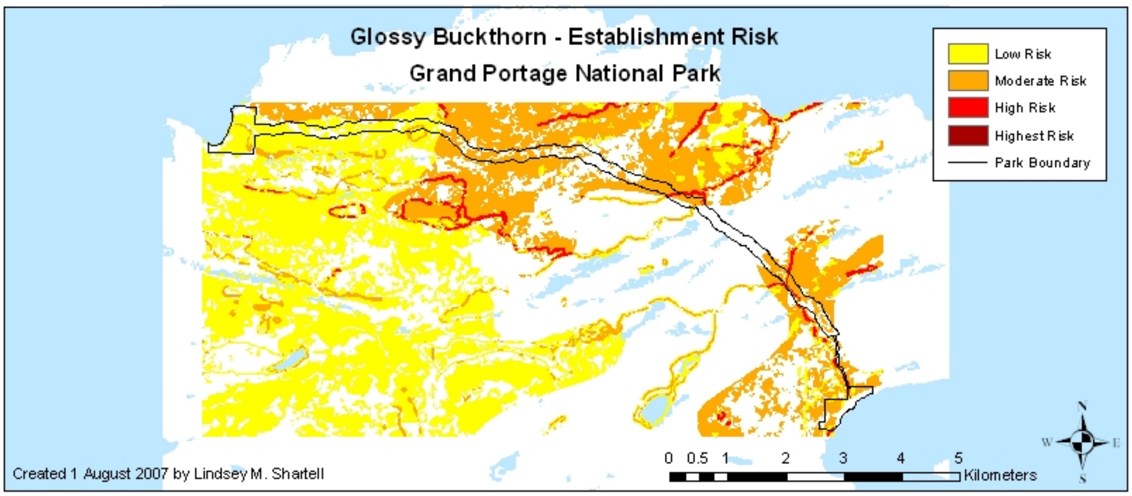


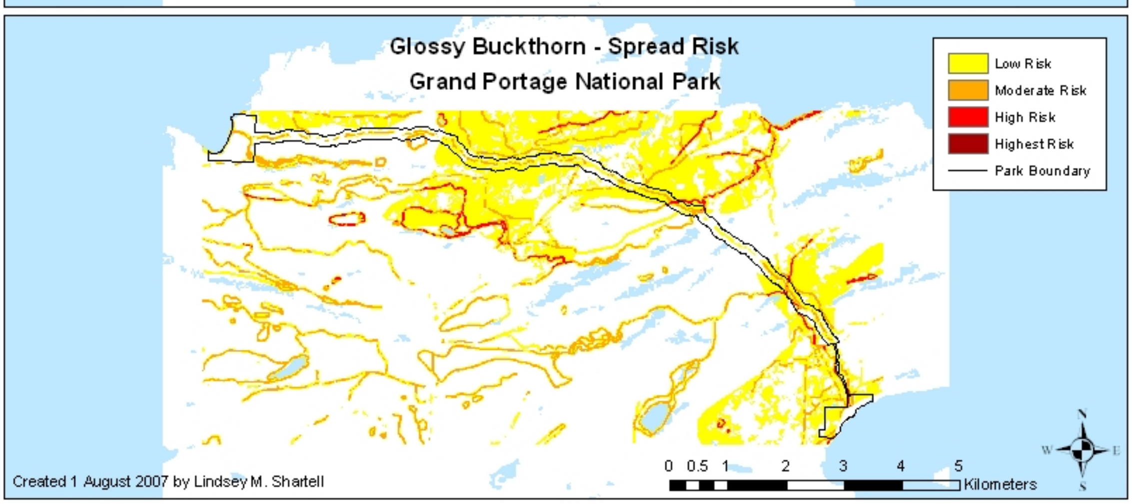




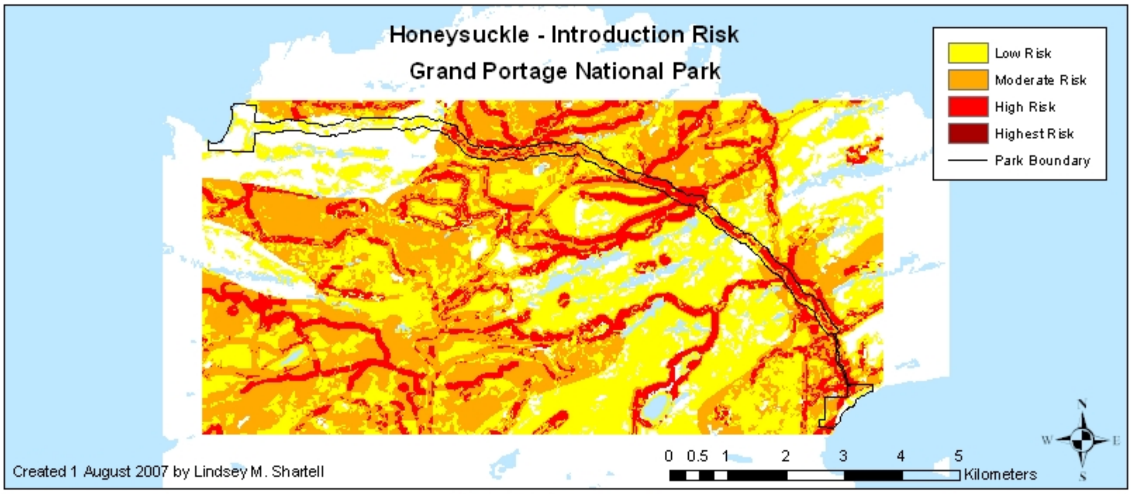
Appendix 3. Cont.

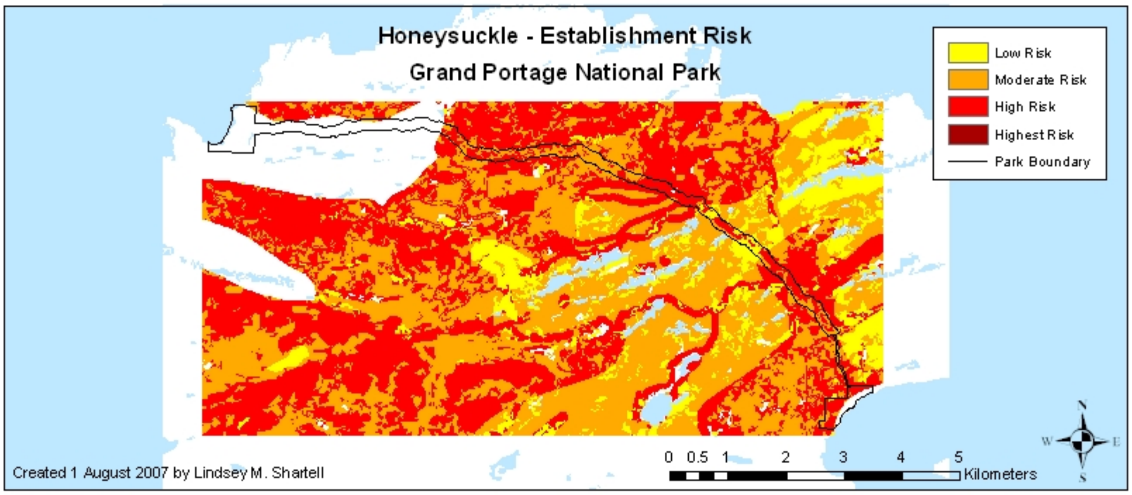


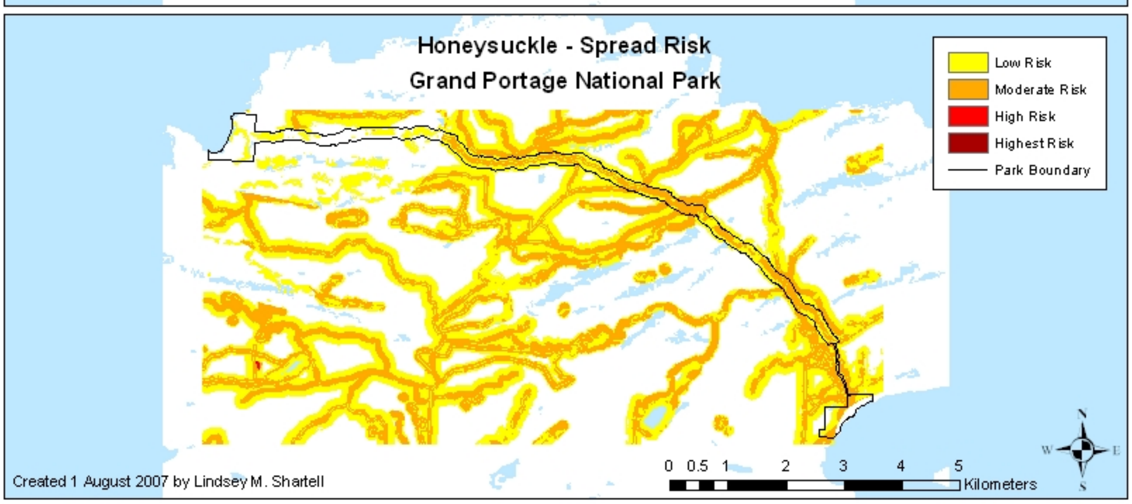




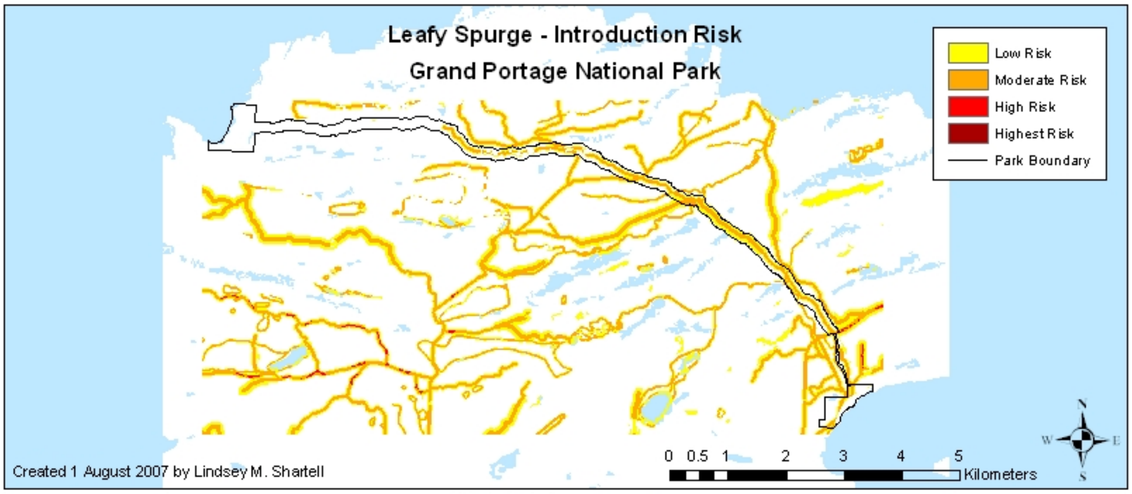
Appendix 3. Cont.

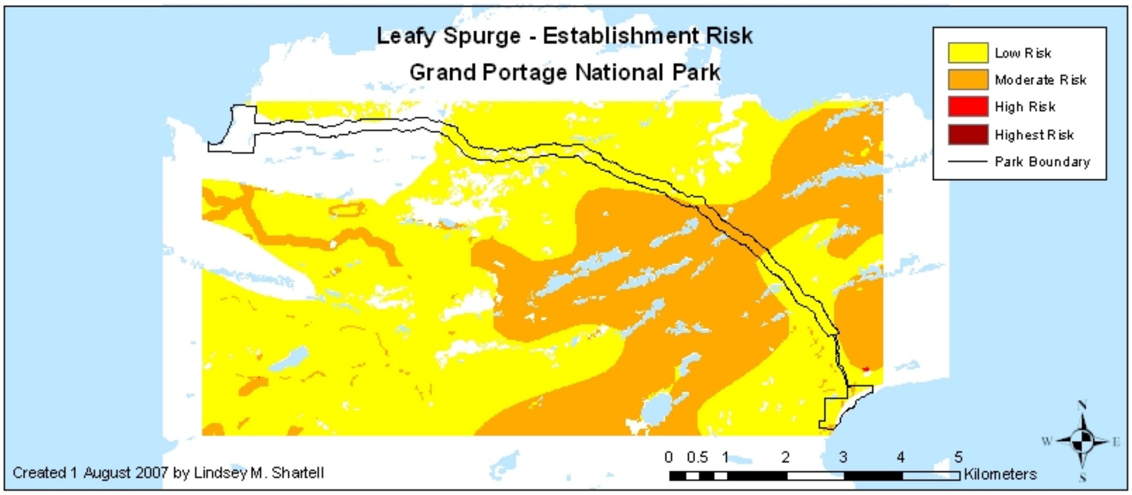


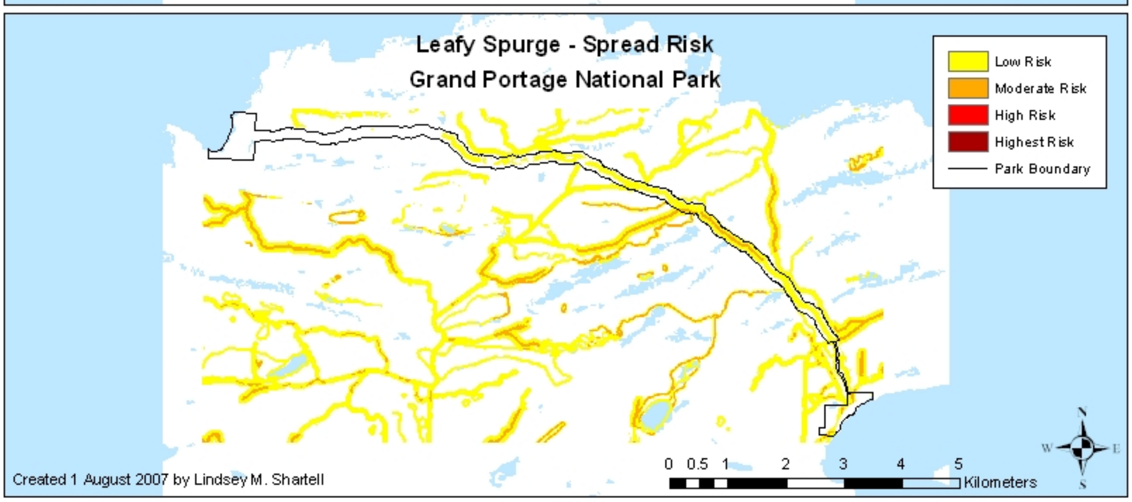




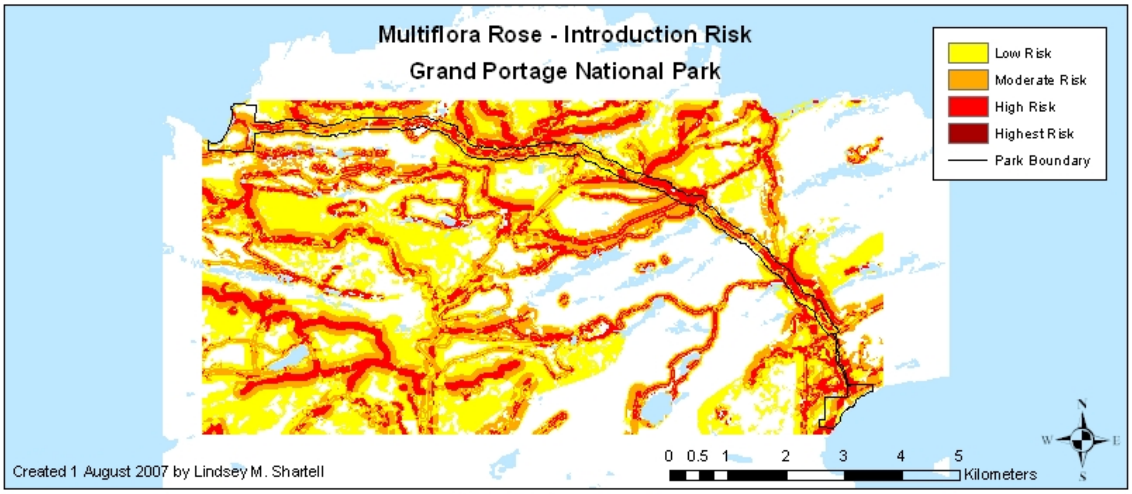
Appendix 3. Cont.

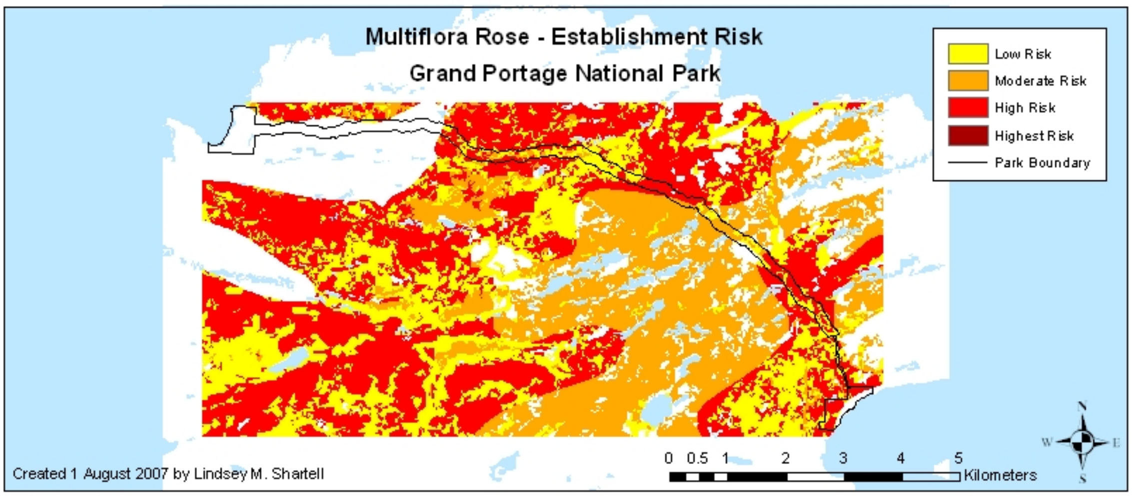


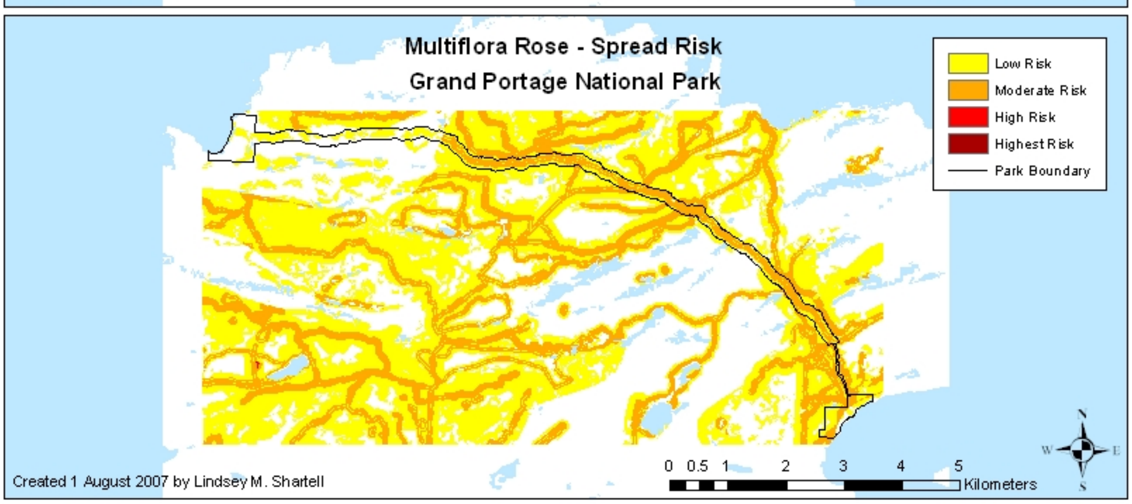




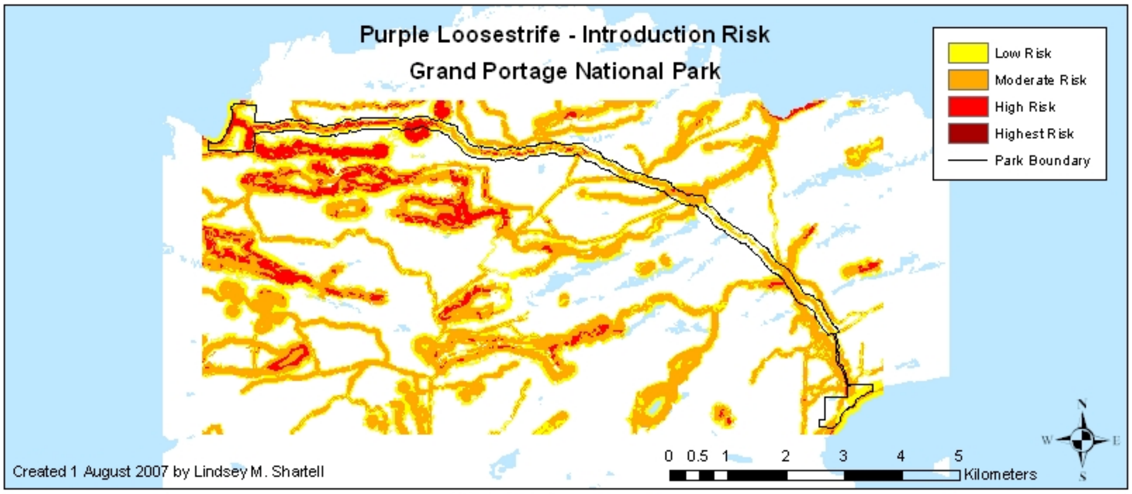
Appendix 3. Cont.

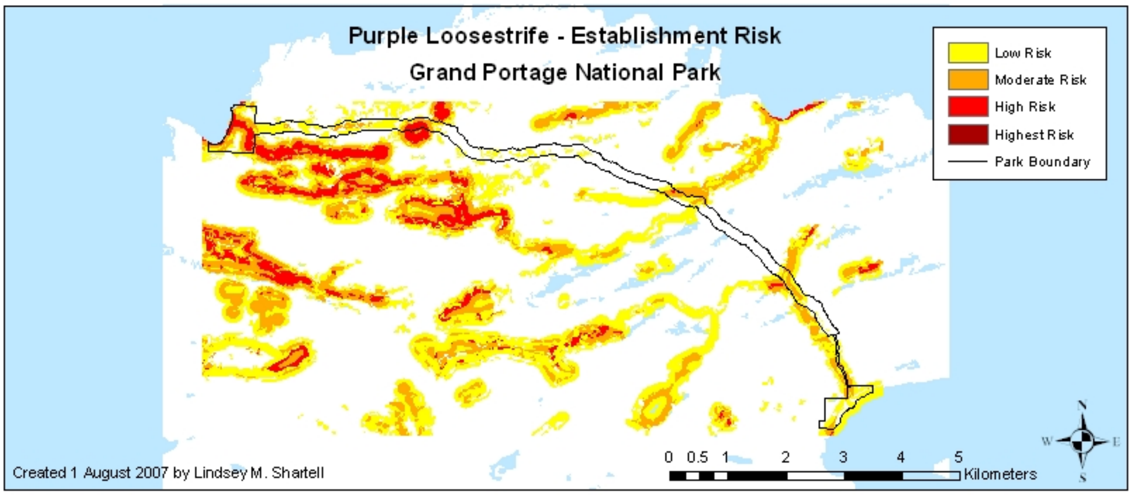


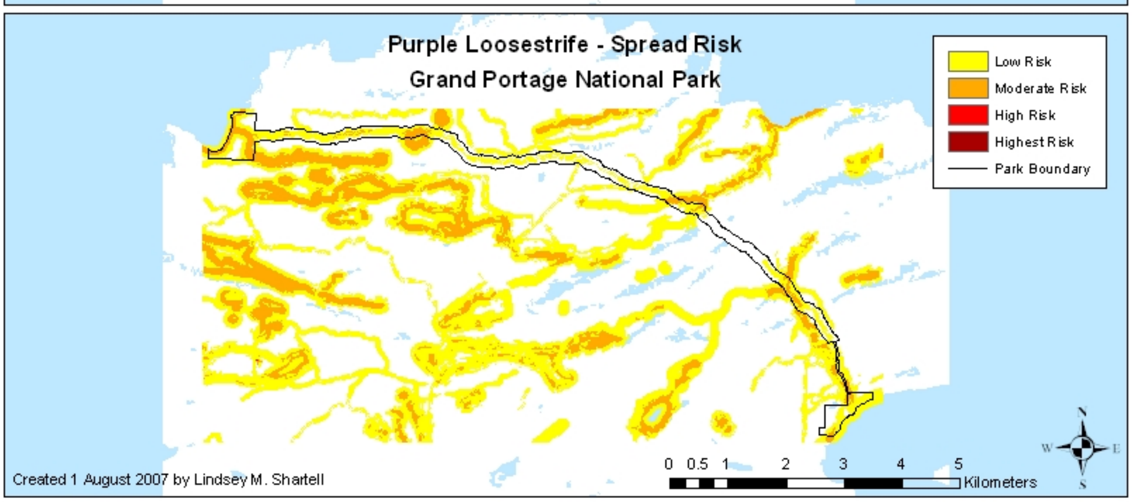




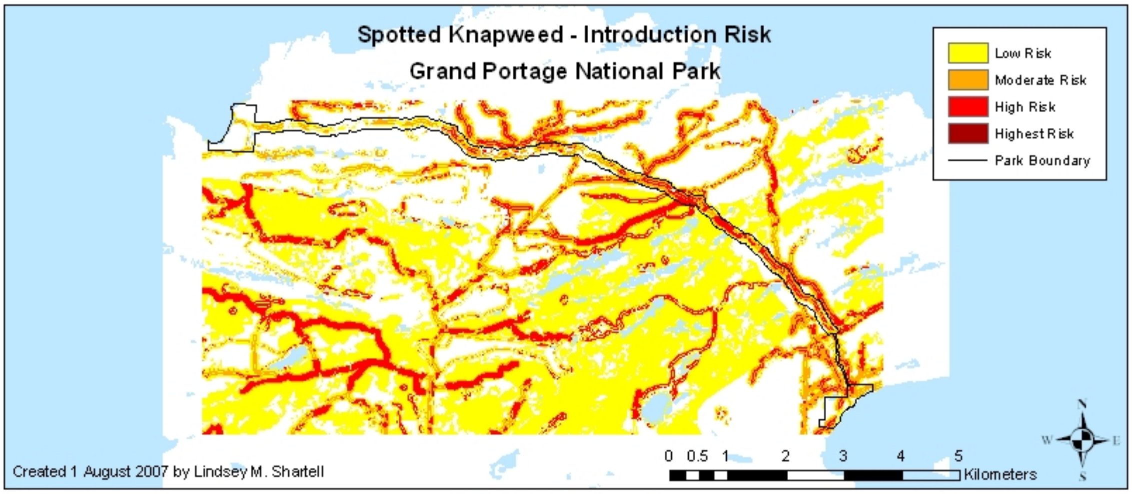
Appendix 3. Cont.

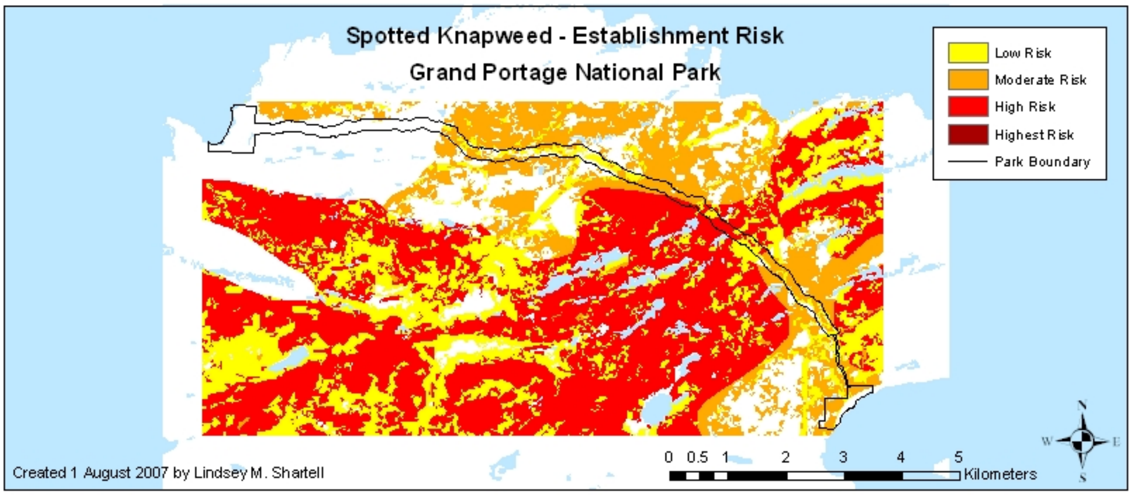


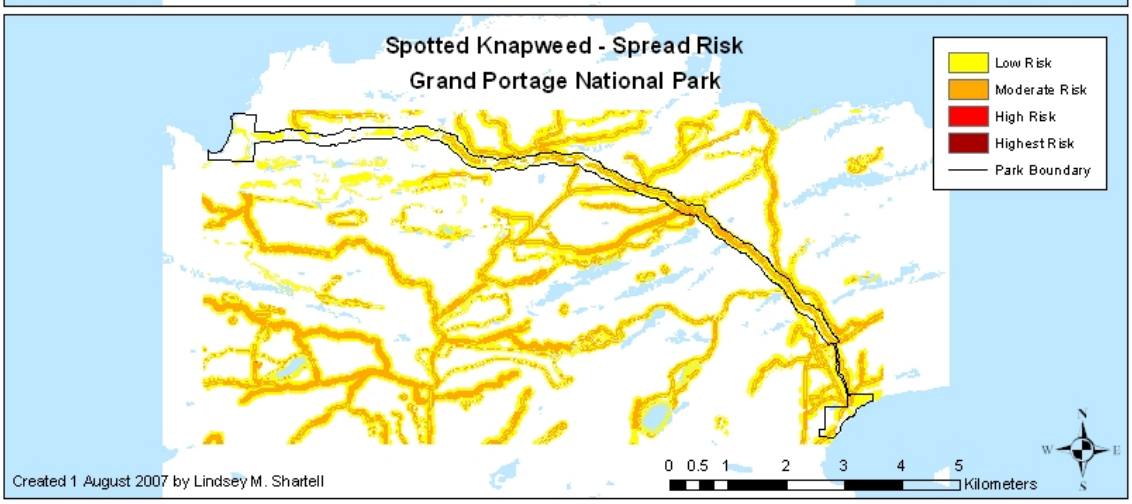




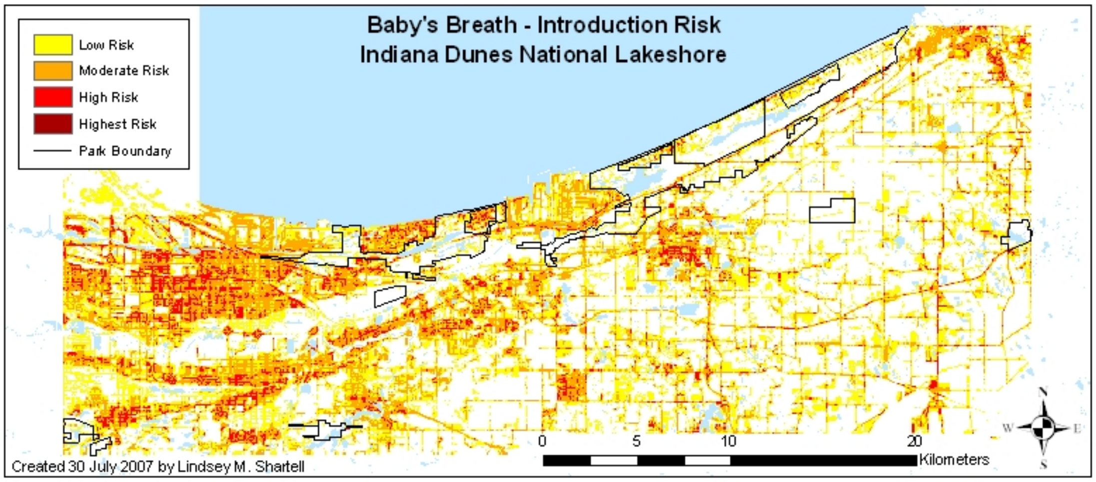
Appendix 3. Cont.

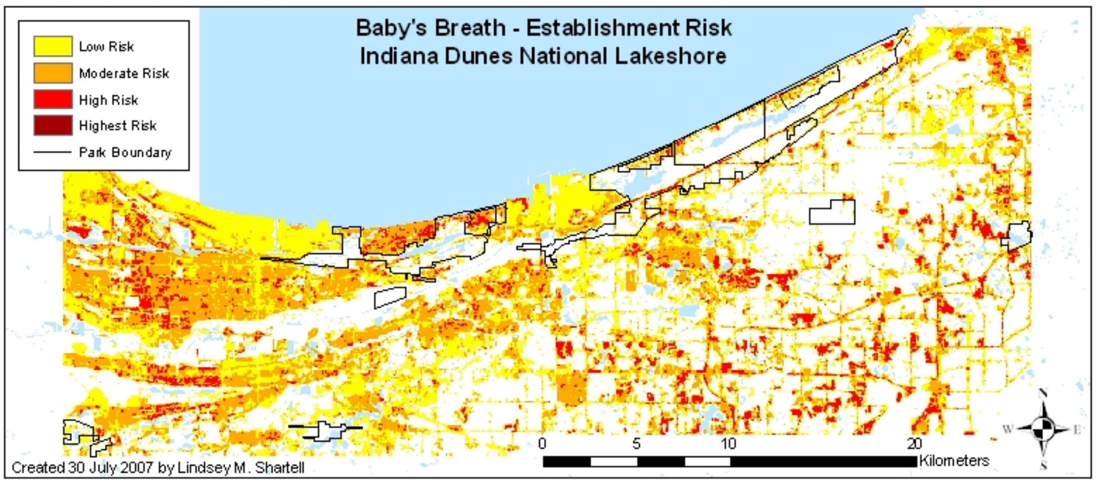


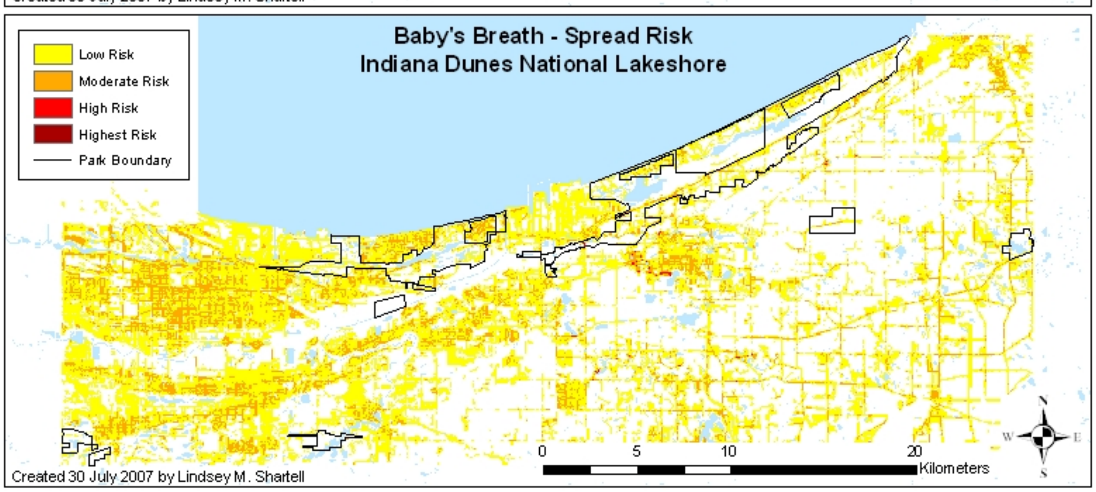




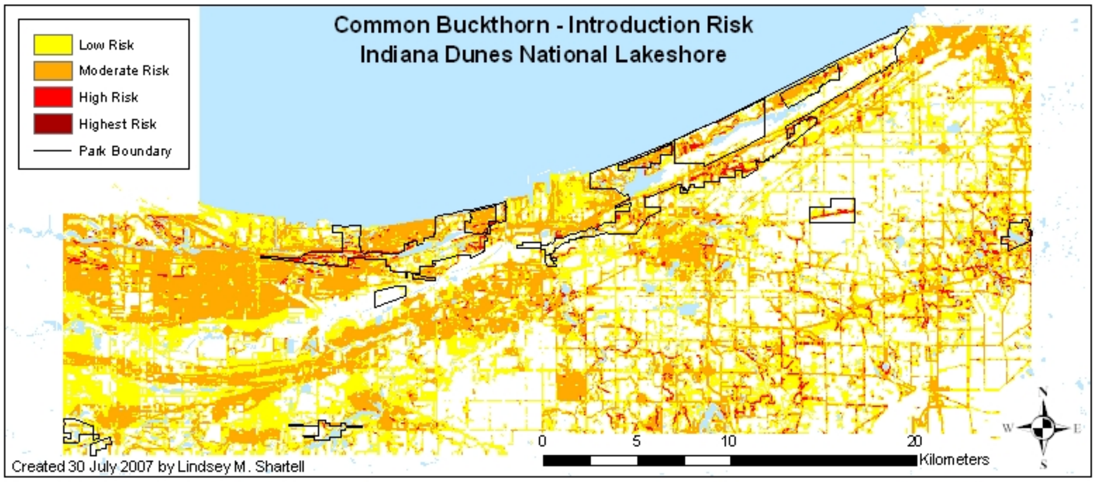
Appendix 3. Cont.

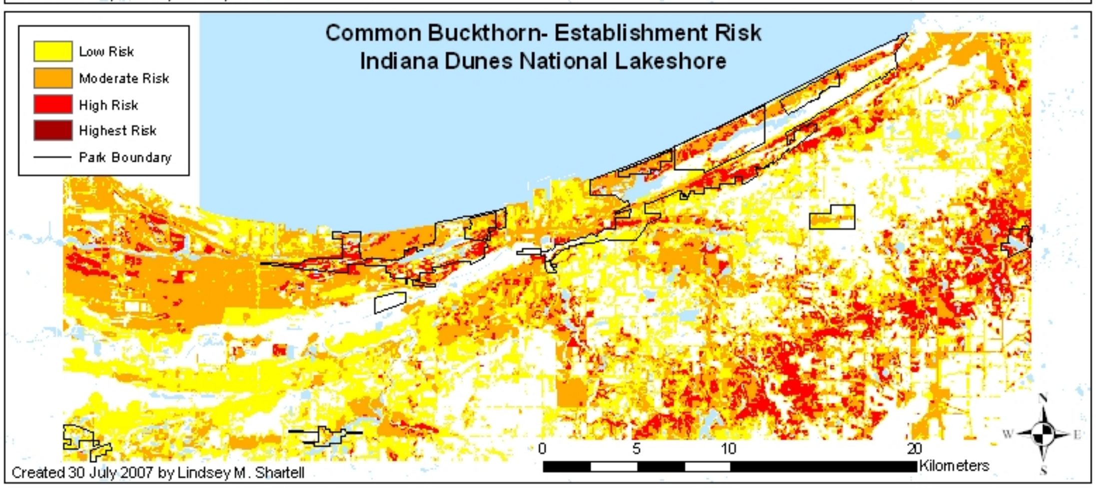


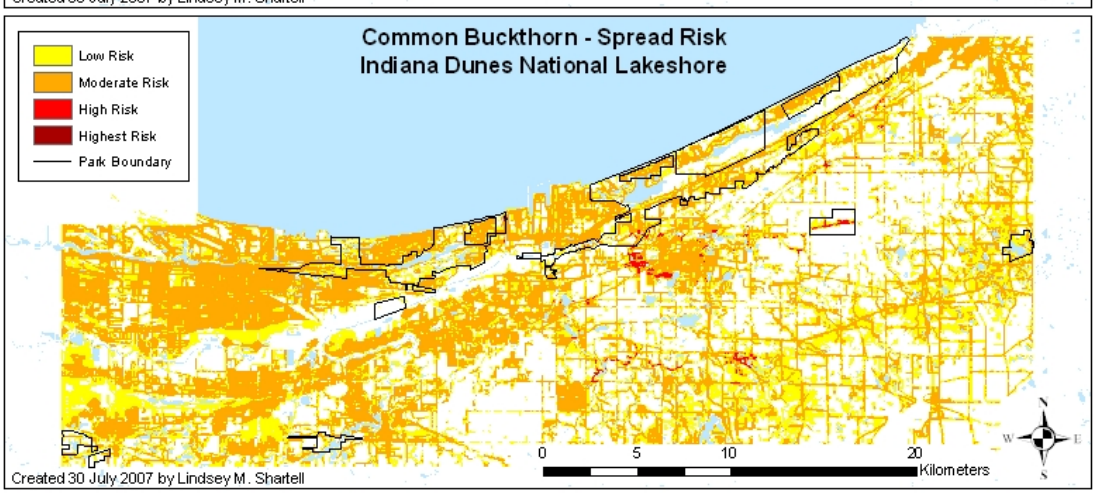




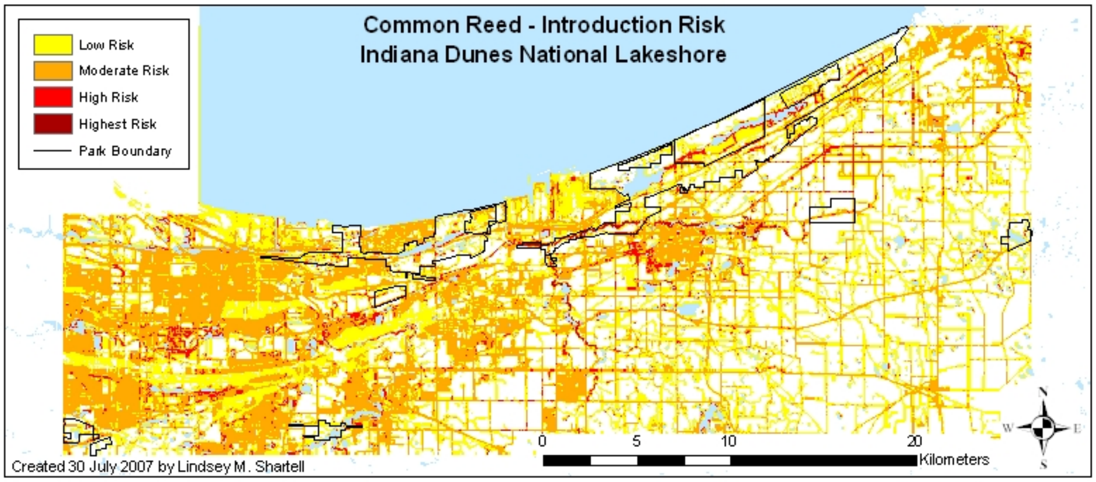
Appendix 3. Cont.

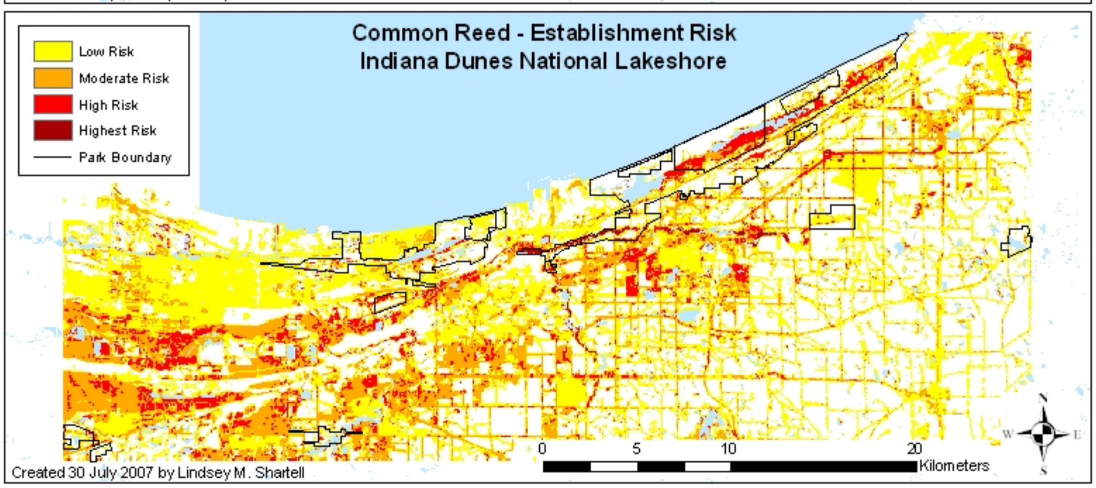


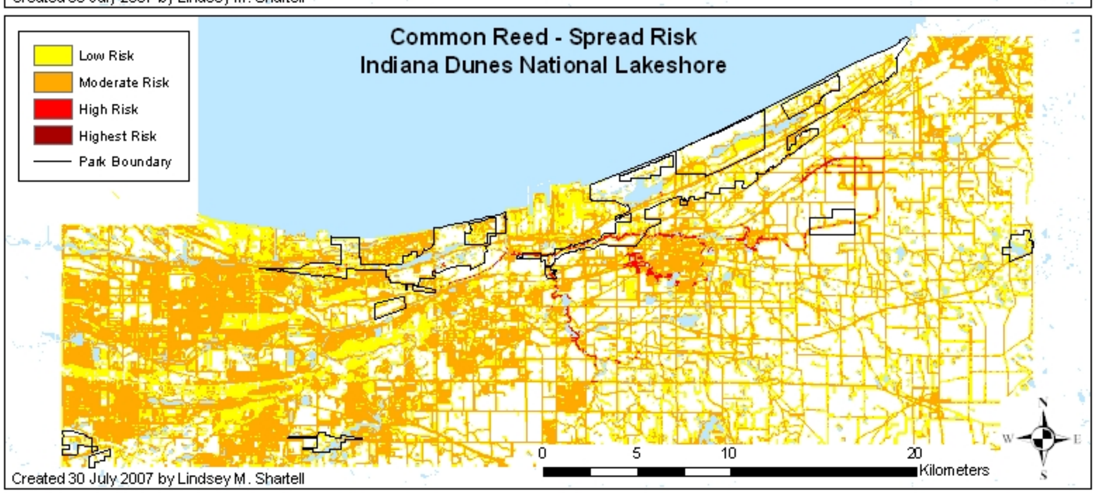




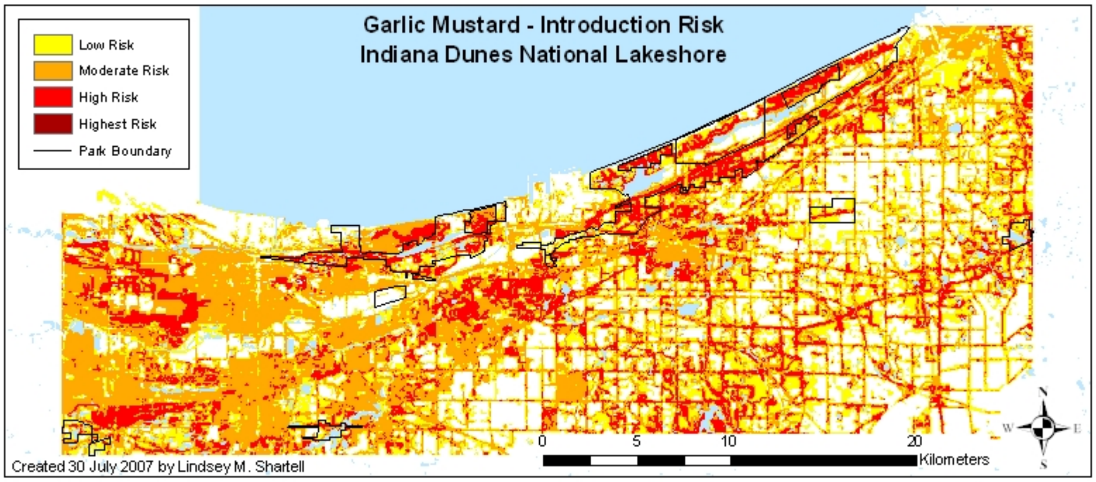
Appendix 3. Cont.

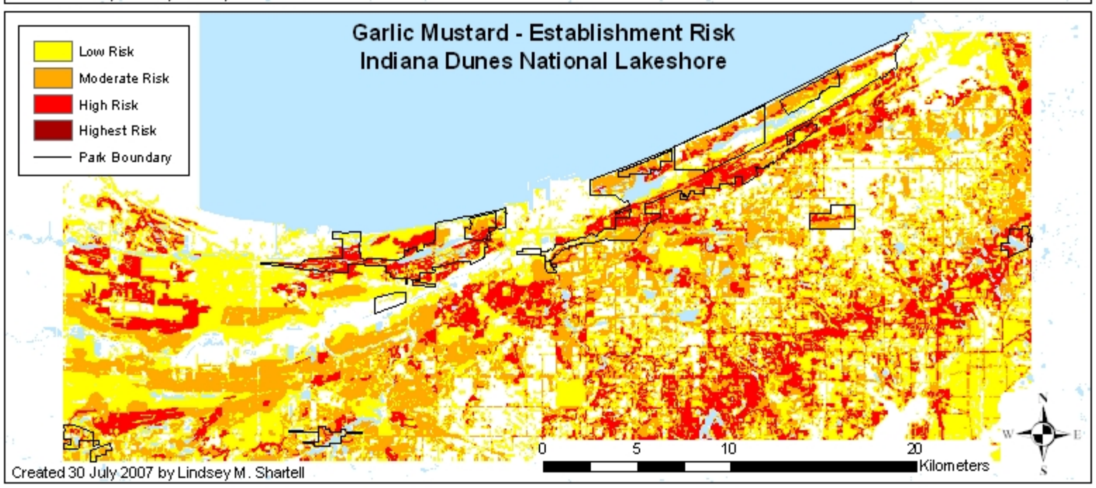


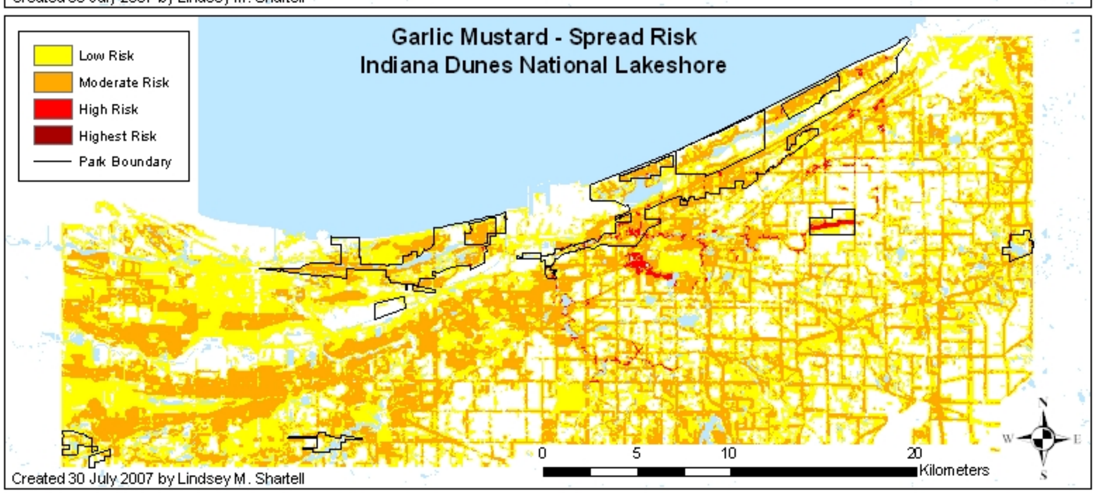




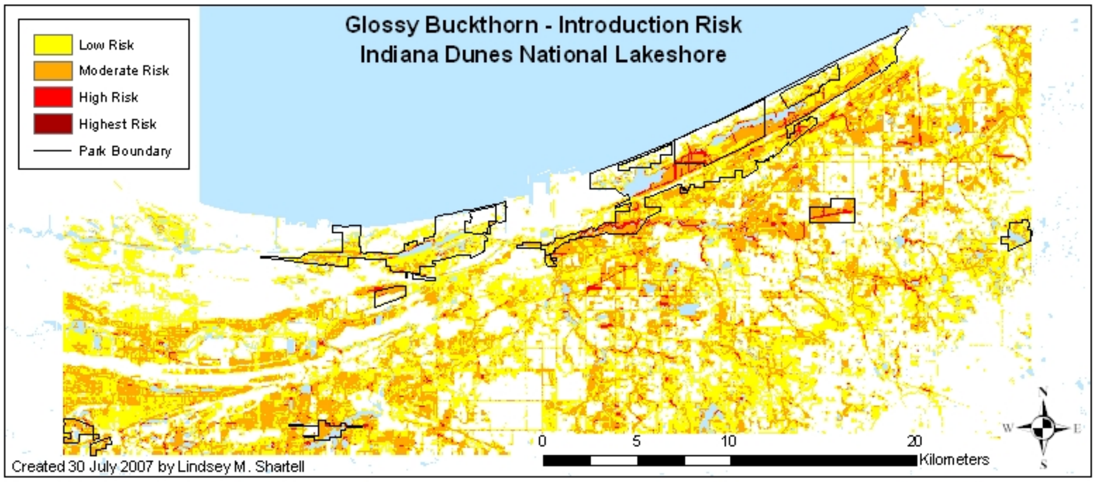
Appendix 3. Cont.

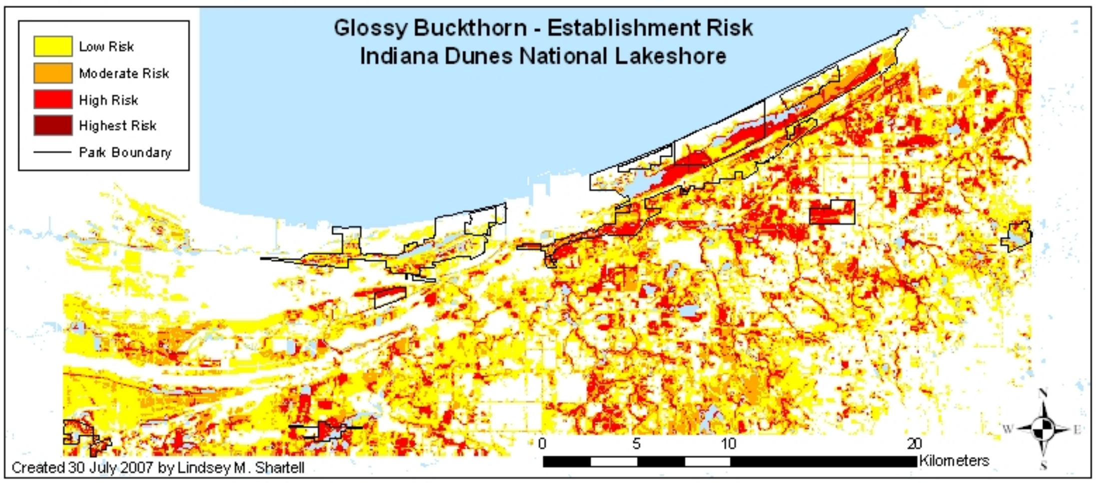


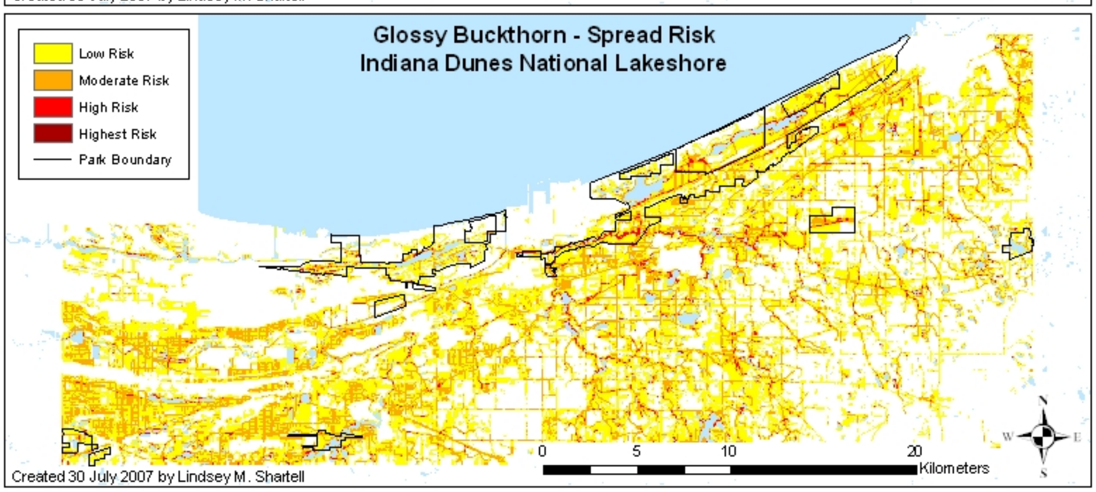




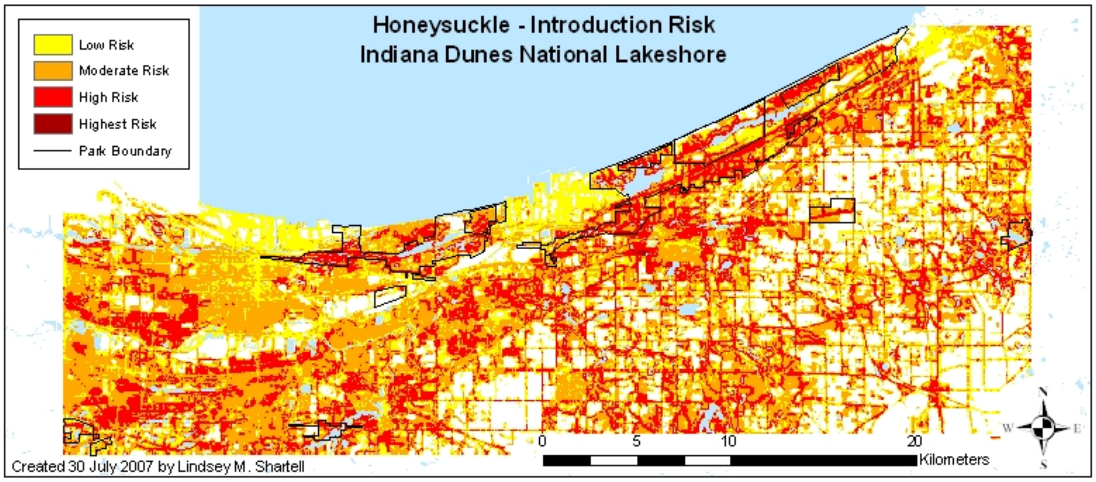
Appendix 3. Cont.

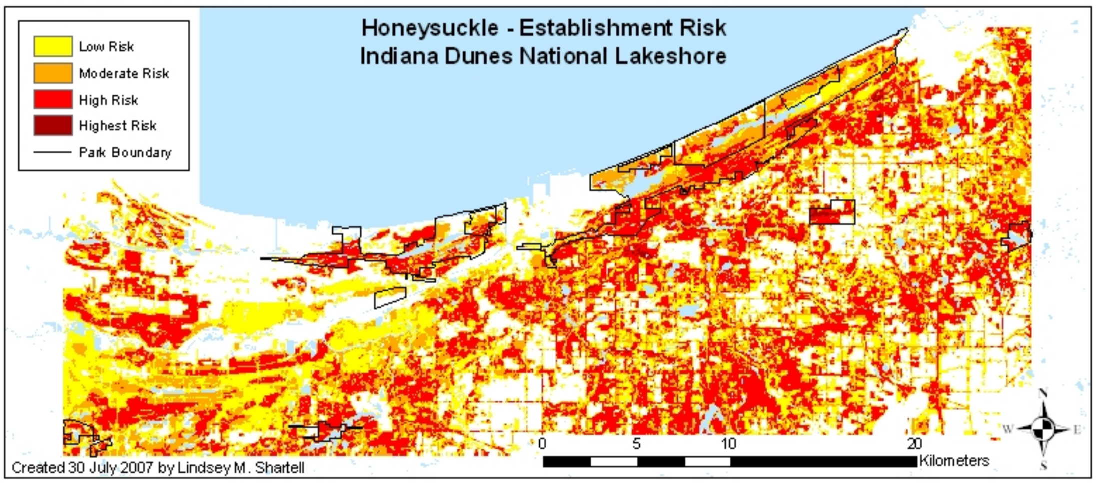


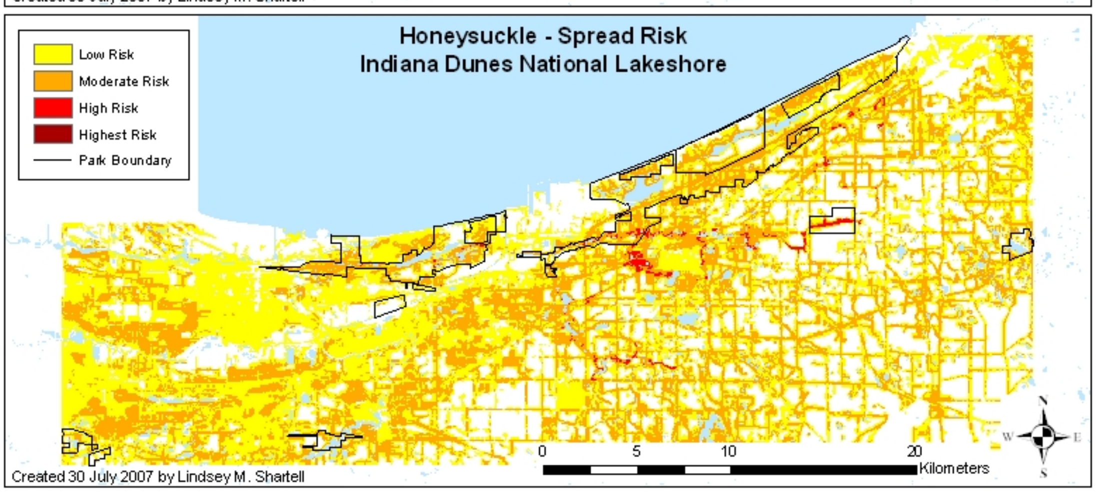




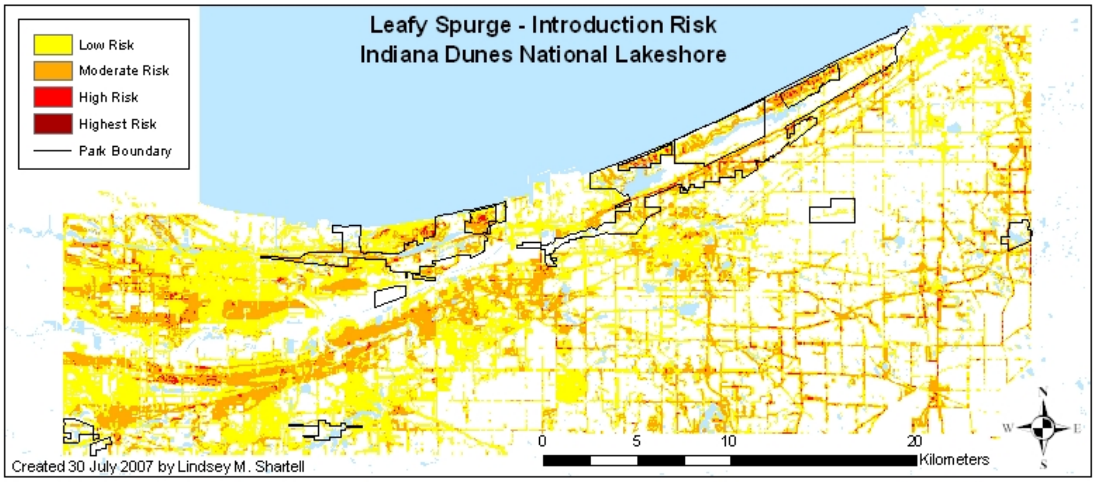
Appendix 3. Cont.

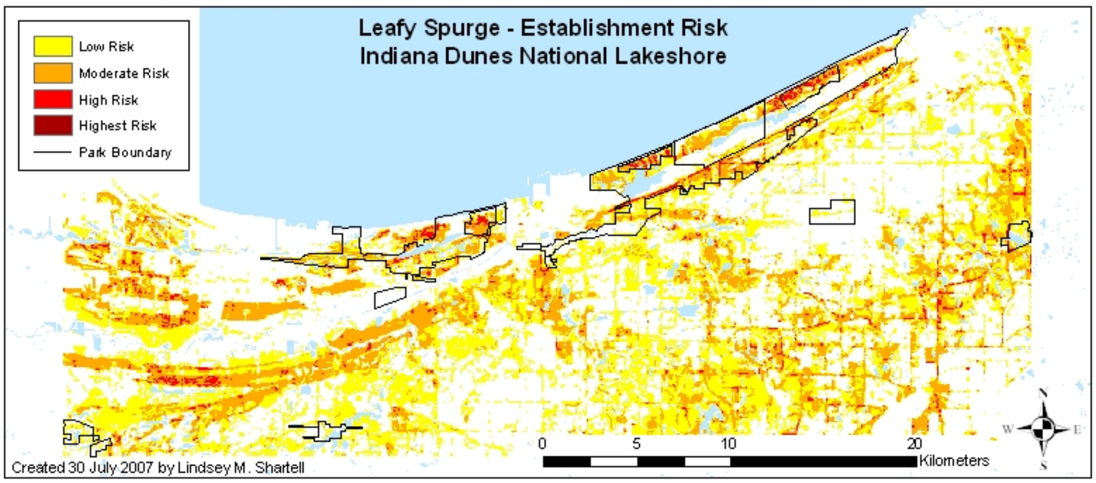


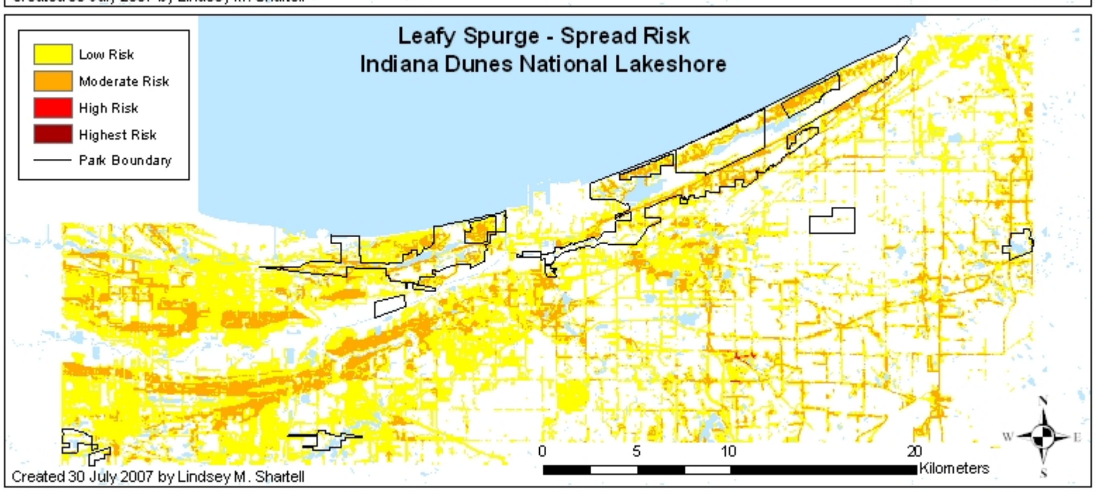




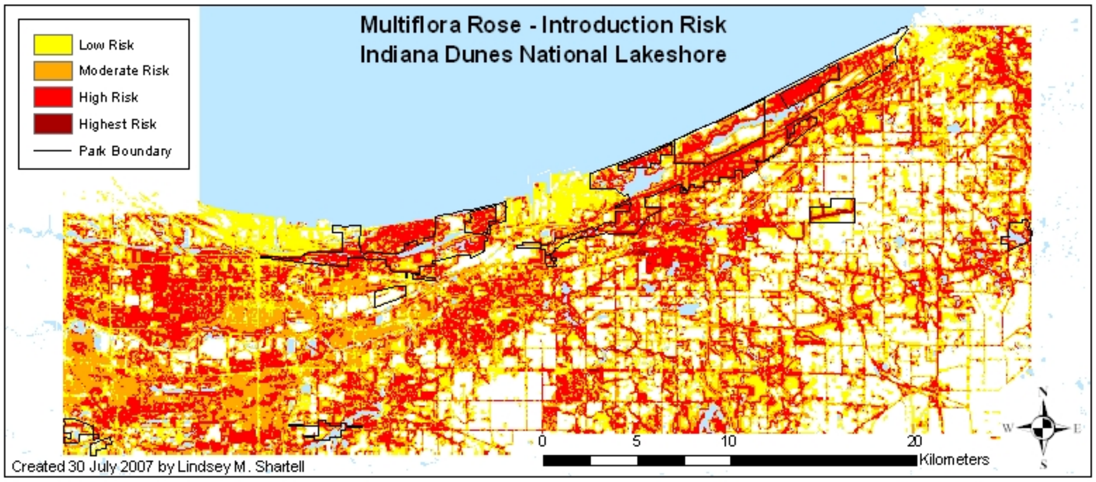
Appendix 3. Cont.

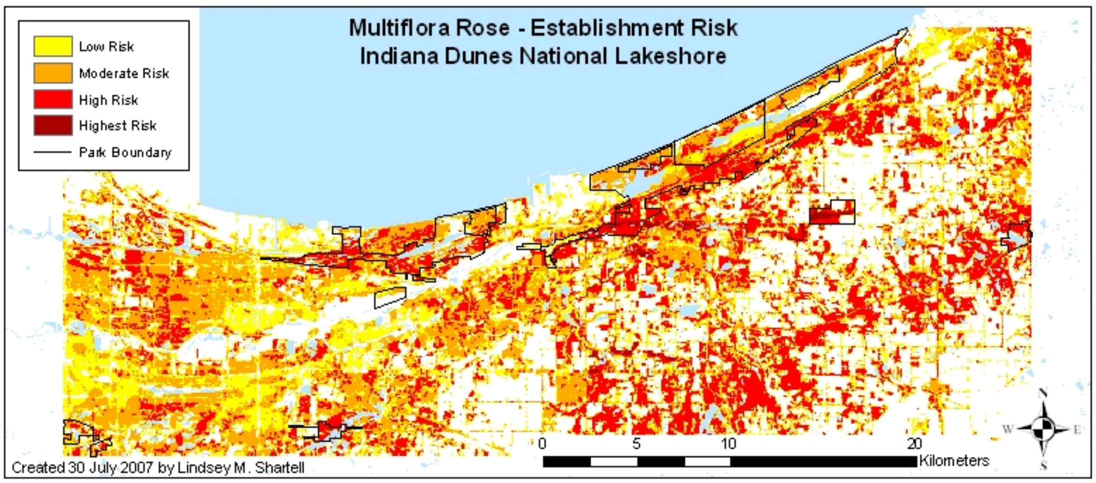


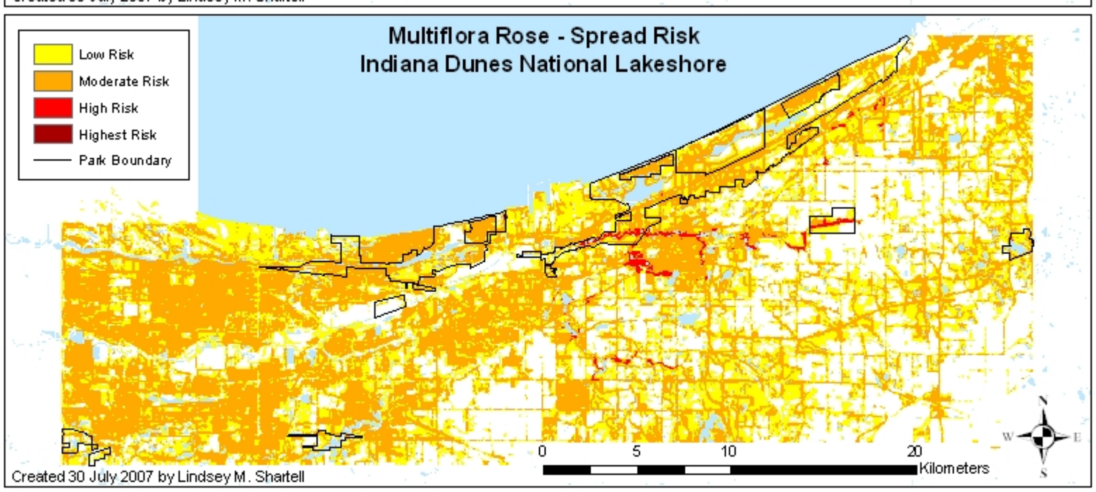




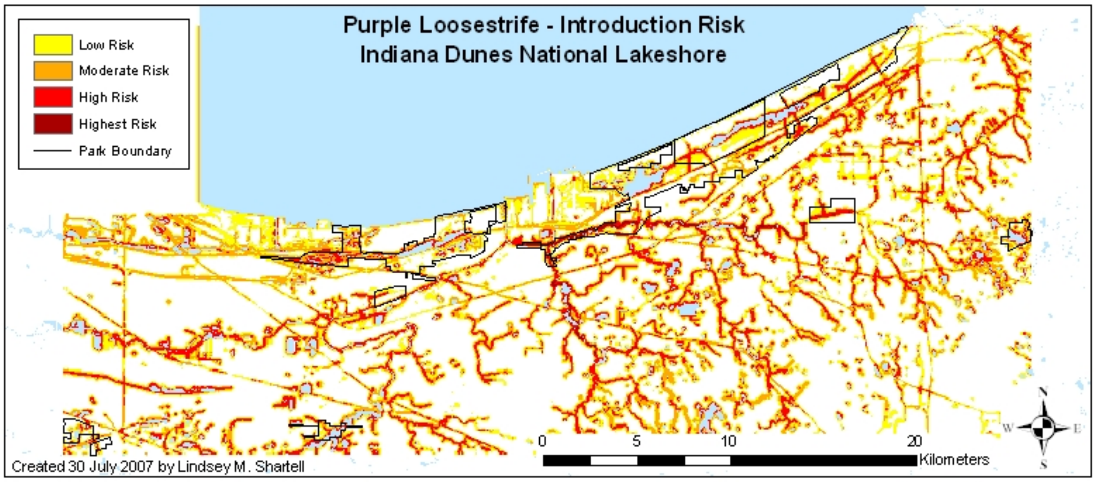
Appendix 3. Cont.

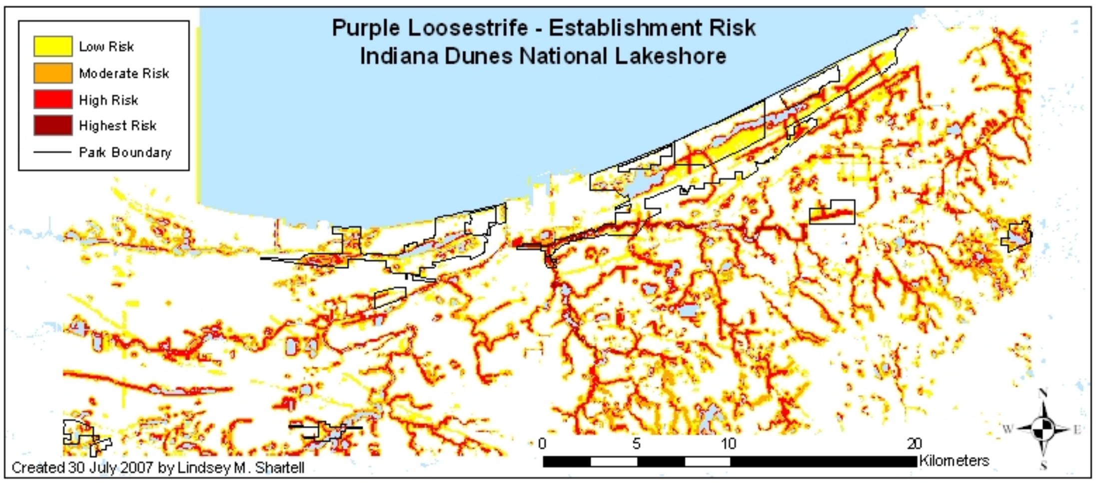


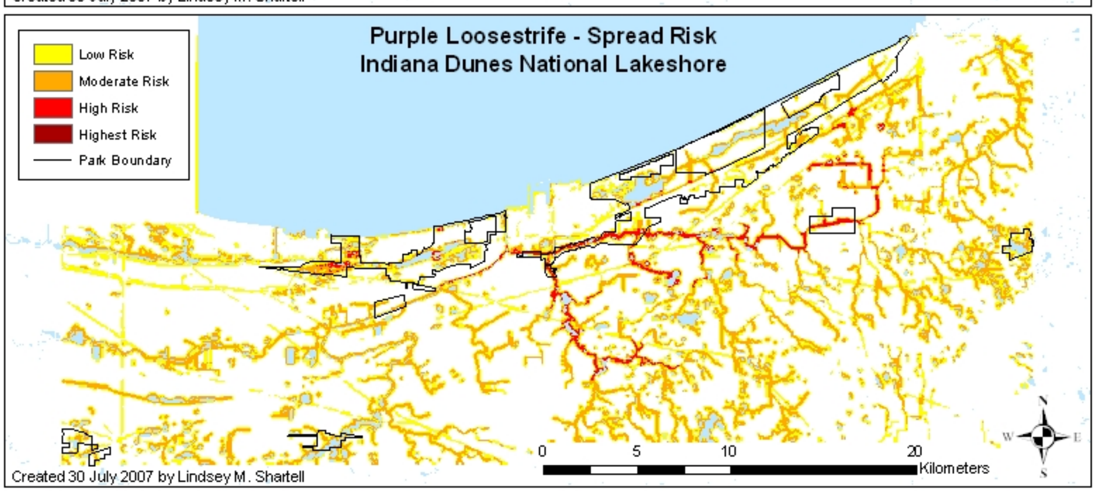




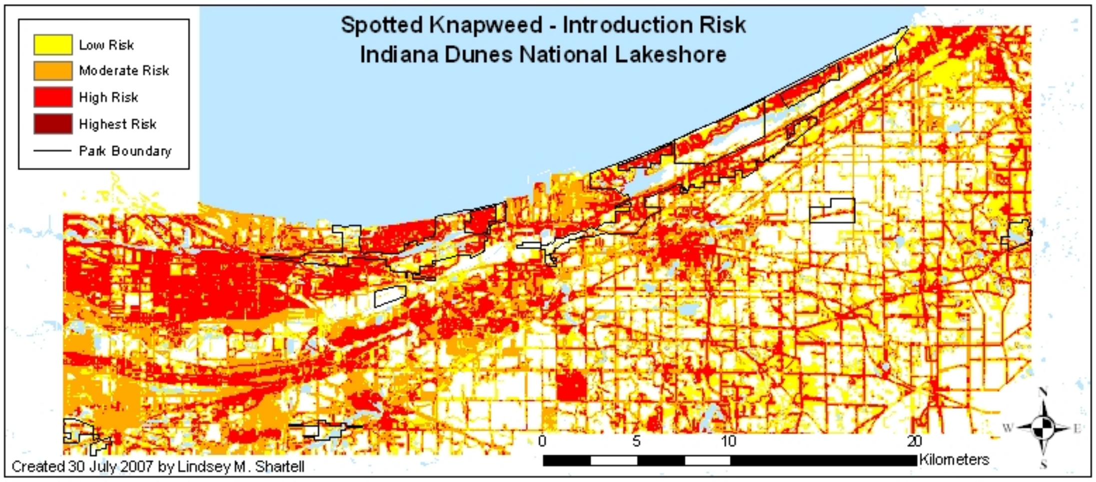
Appendix 3. Cont.

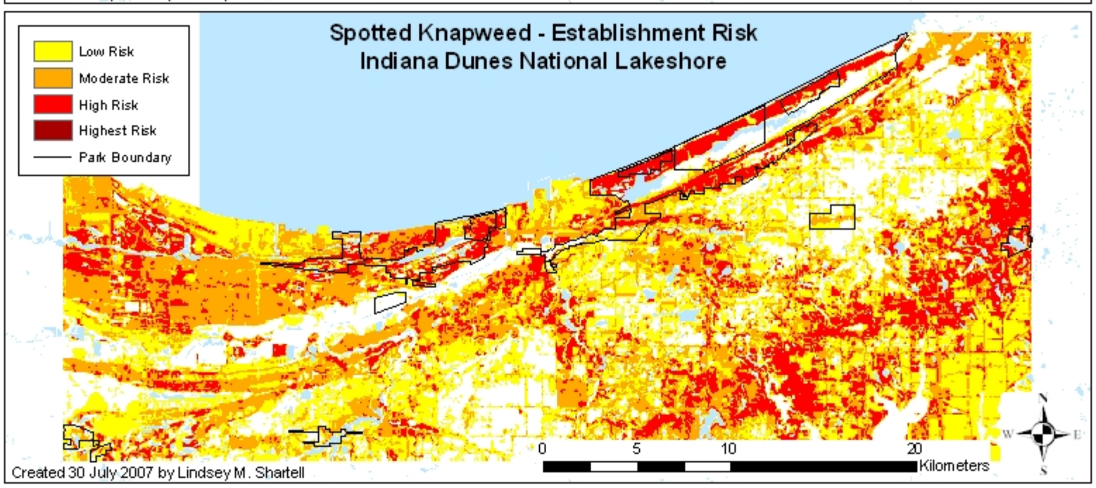


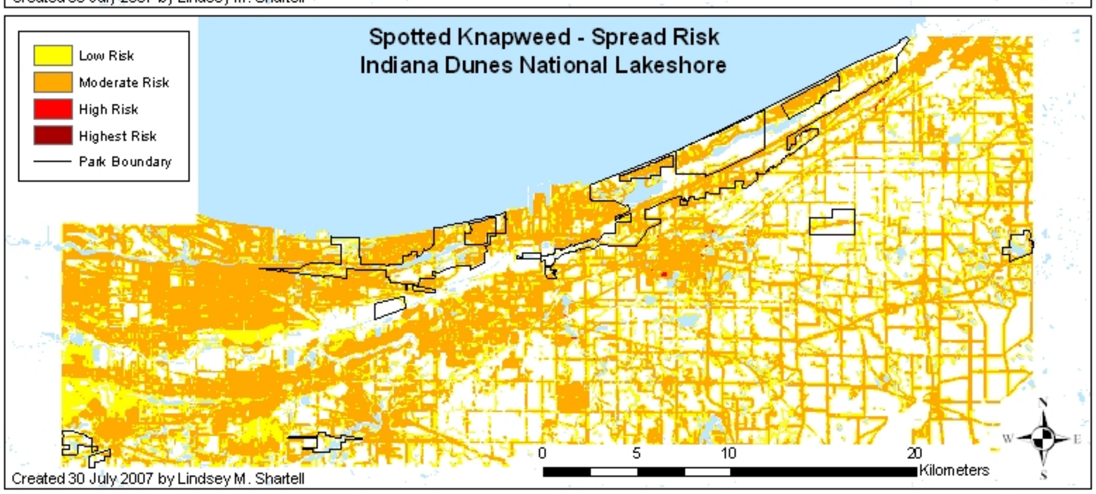




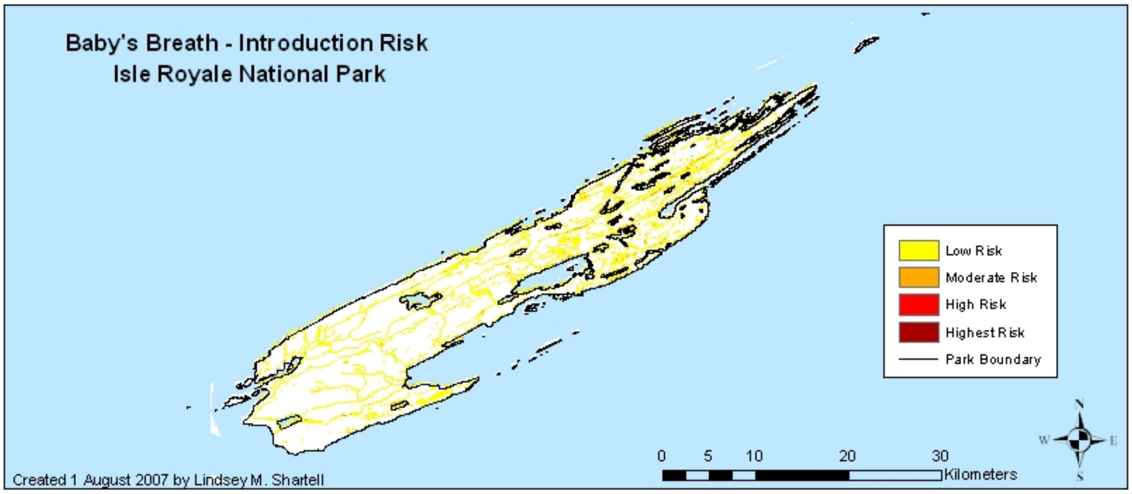
Appendix 3. Cont.

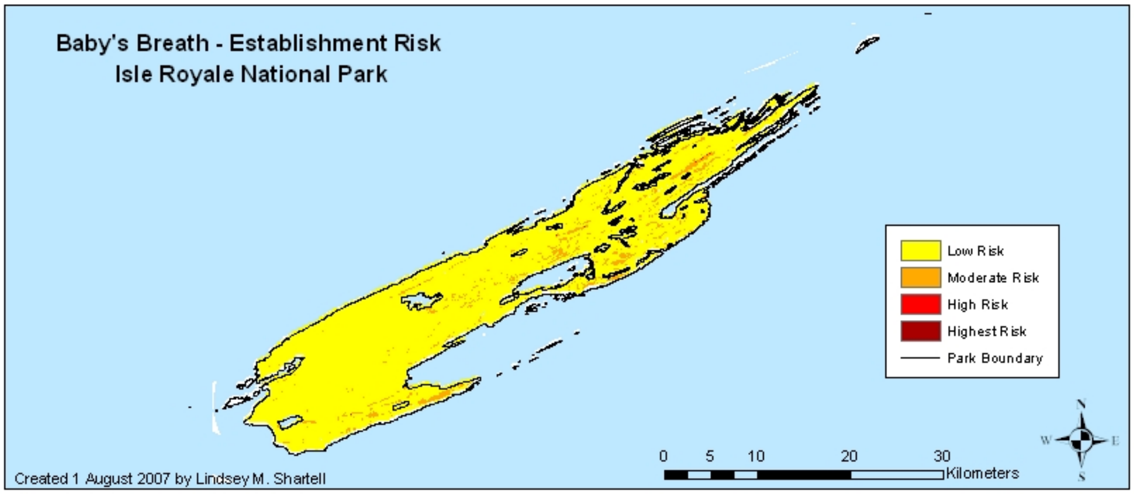


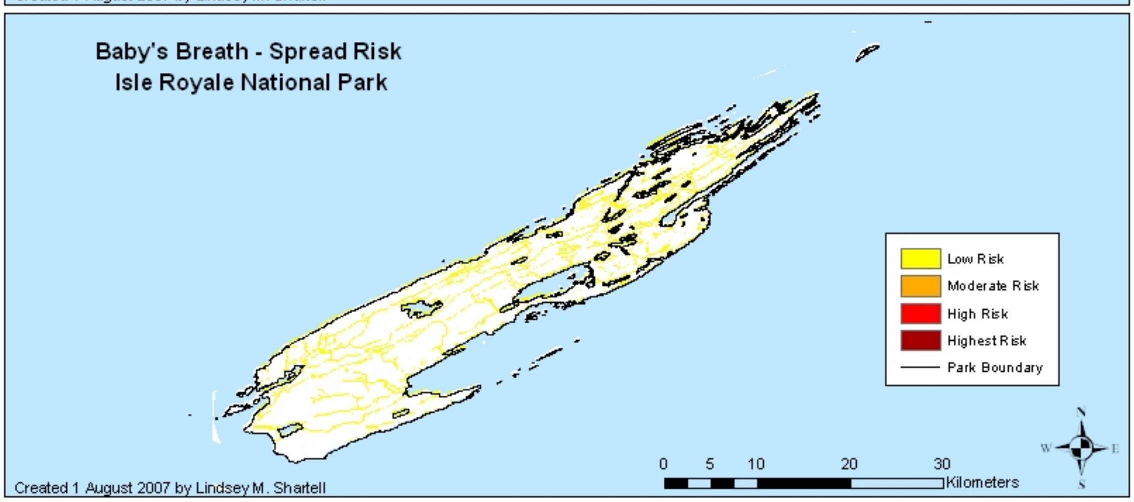




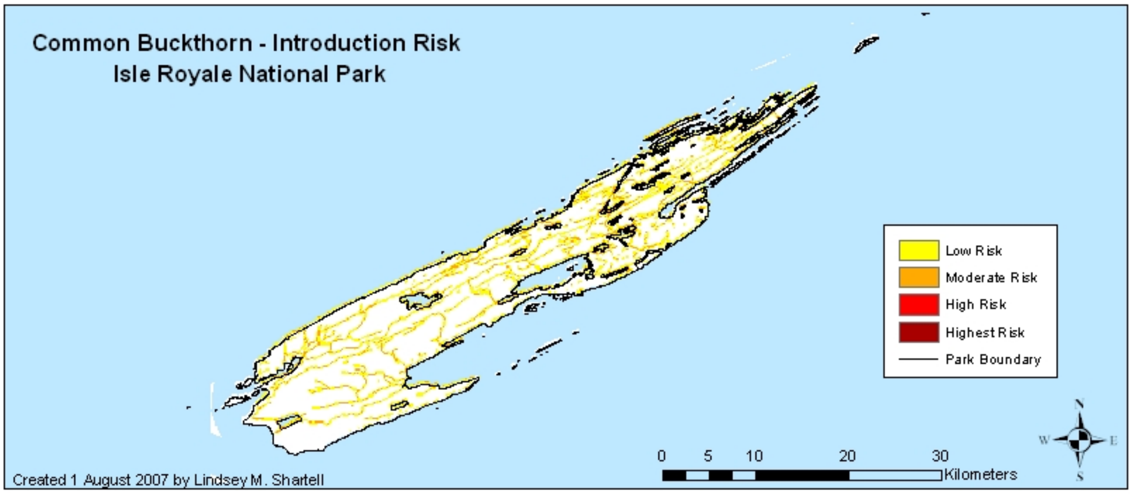
Appendix 3. Cont.

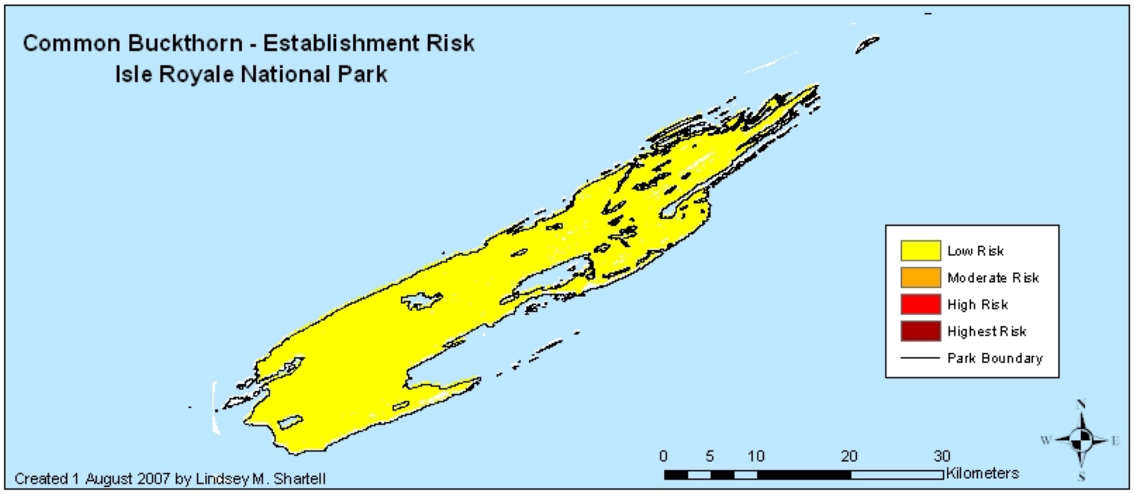


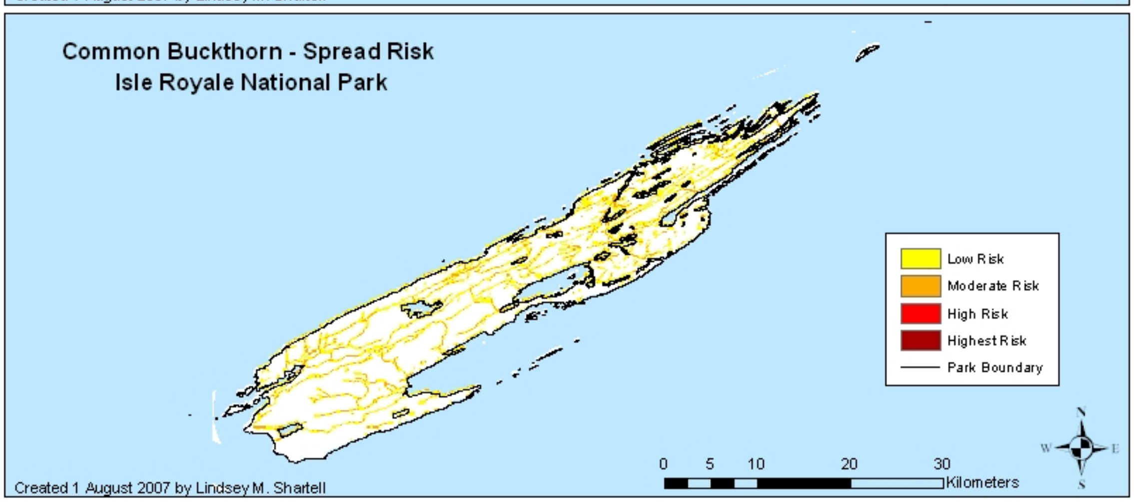




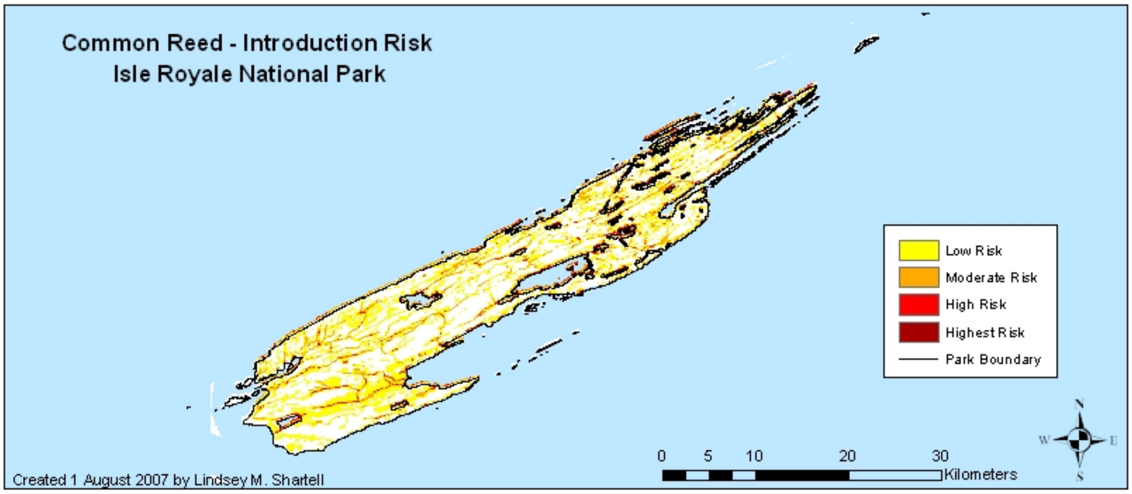
Appendix 3. Cont.

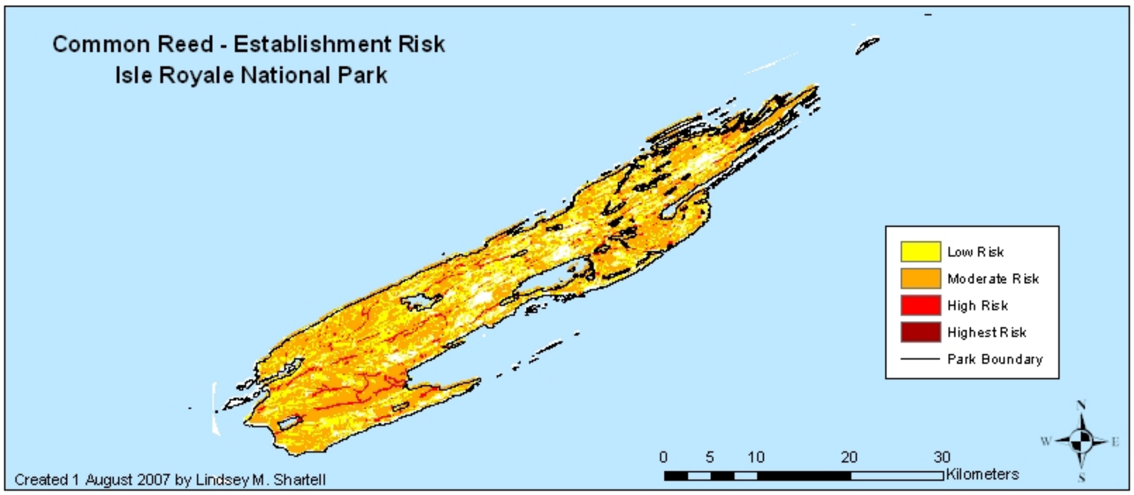


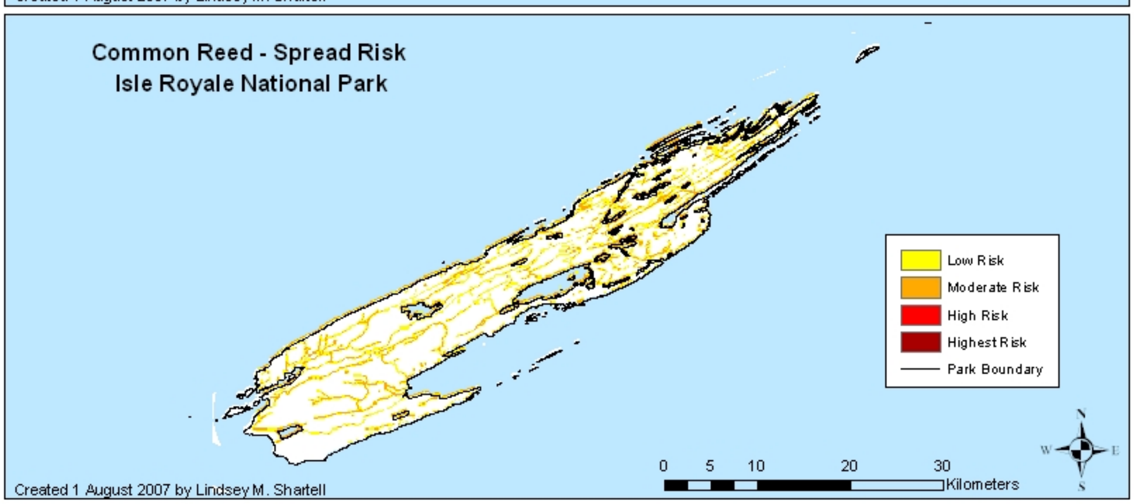




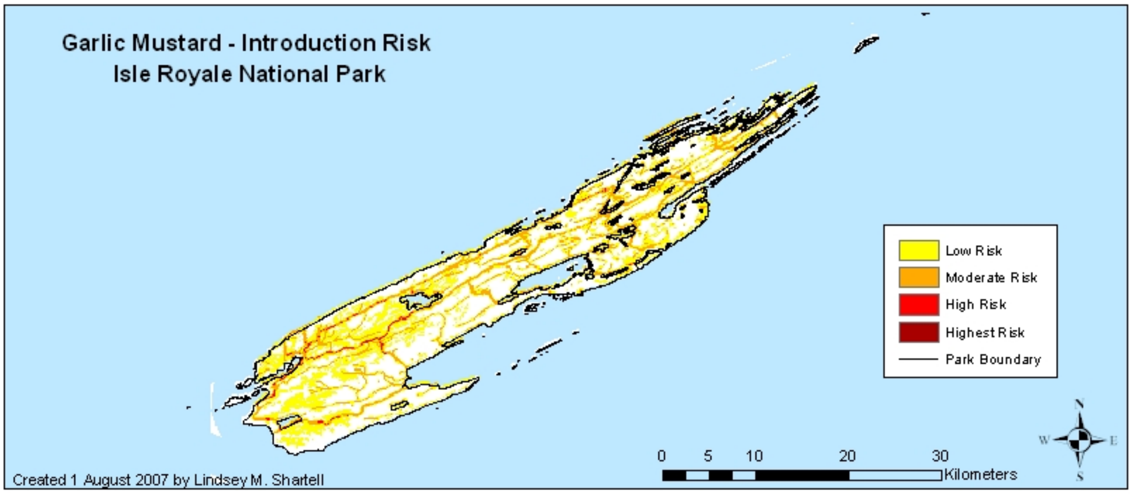
Appendix 3. Cont.

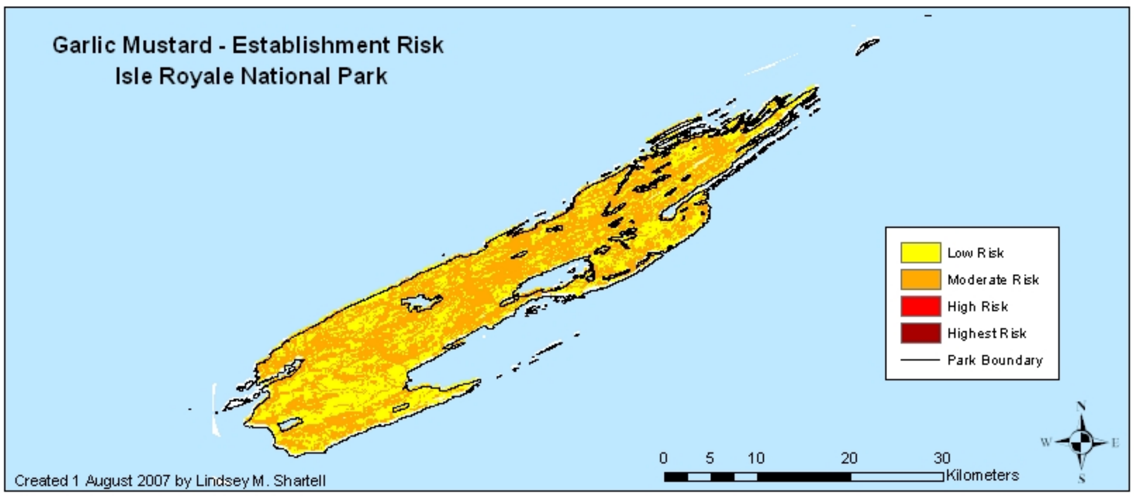


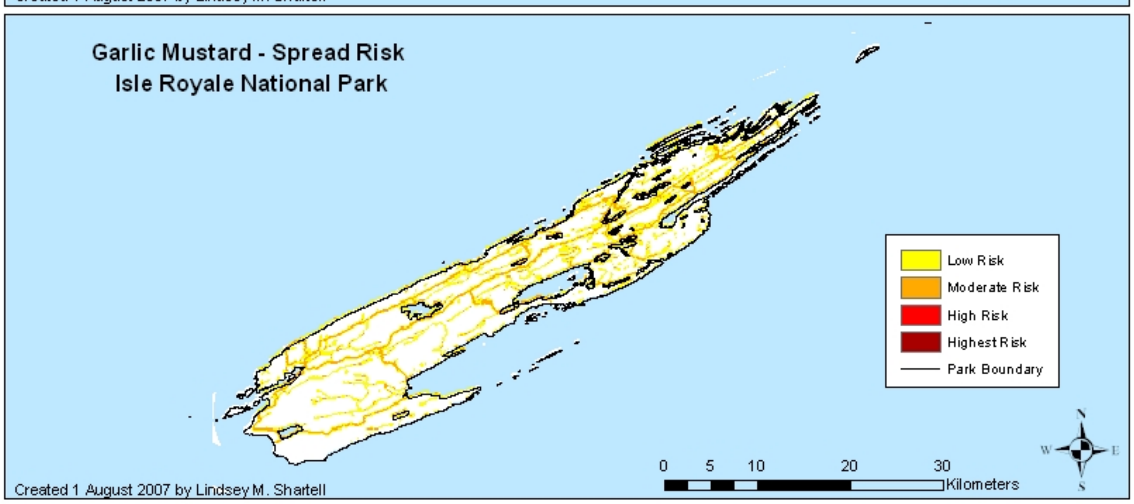




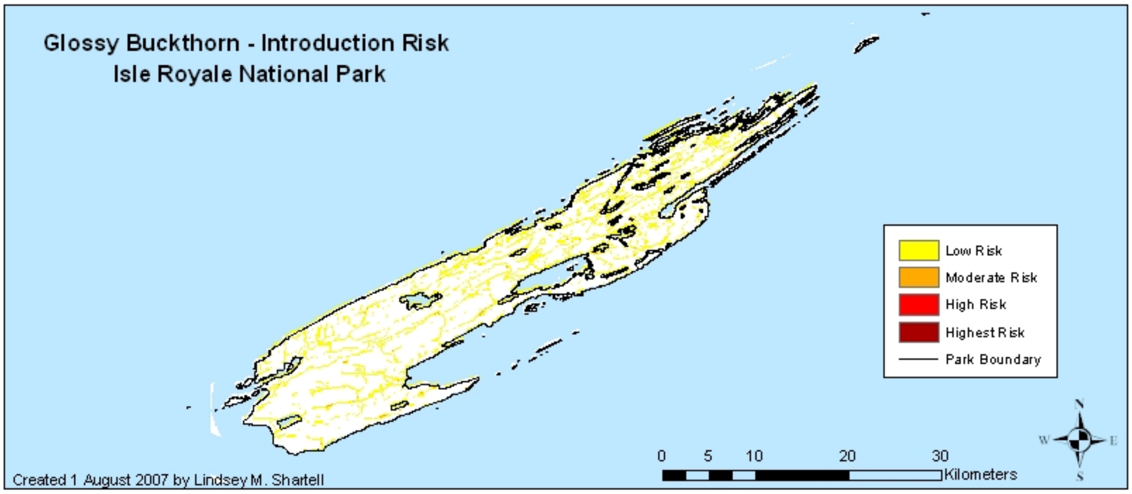
Appendix 3. Cont.

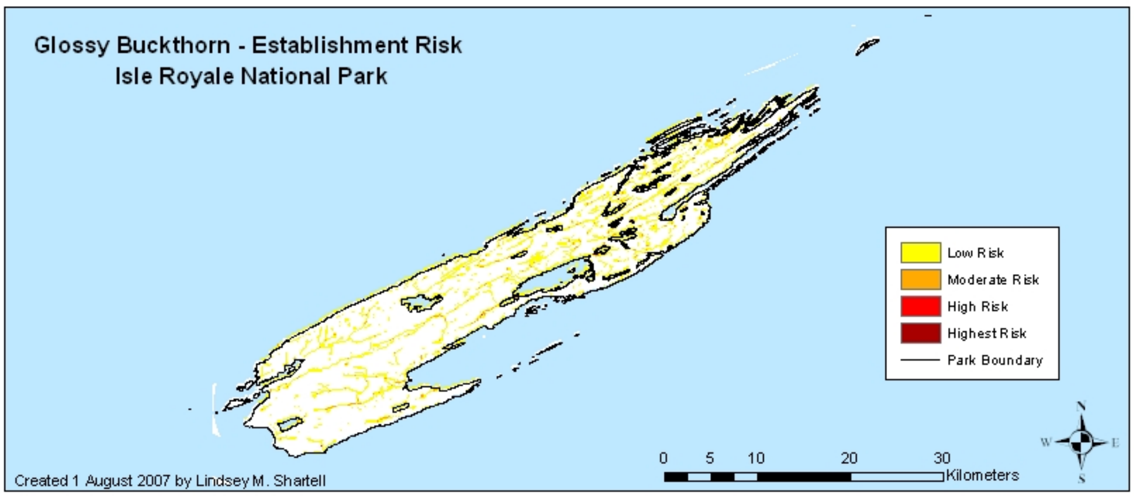


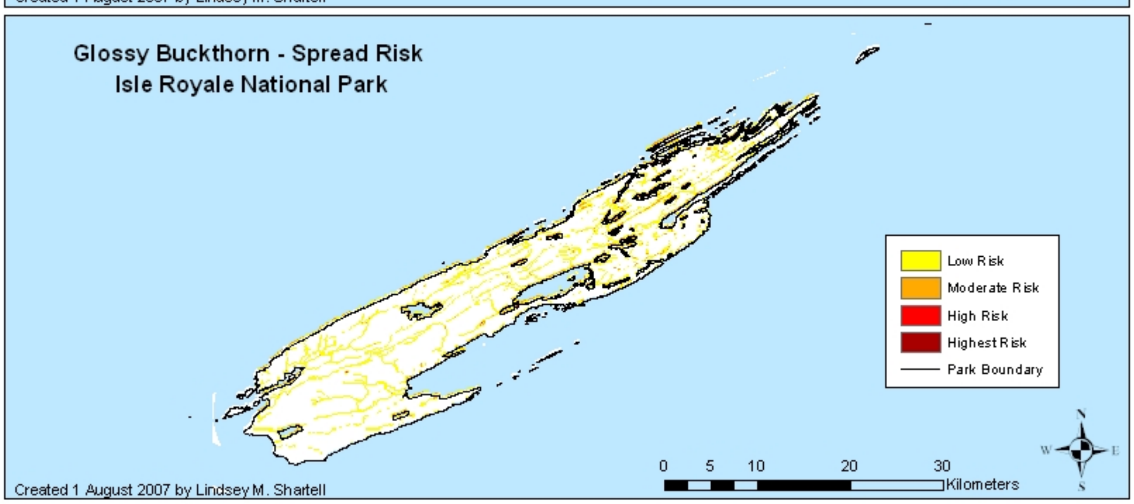




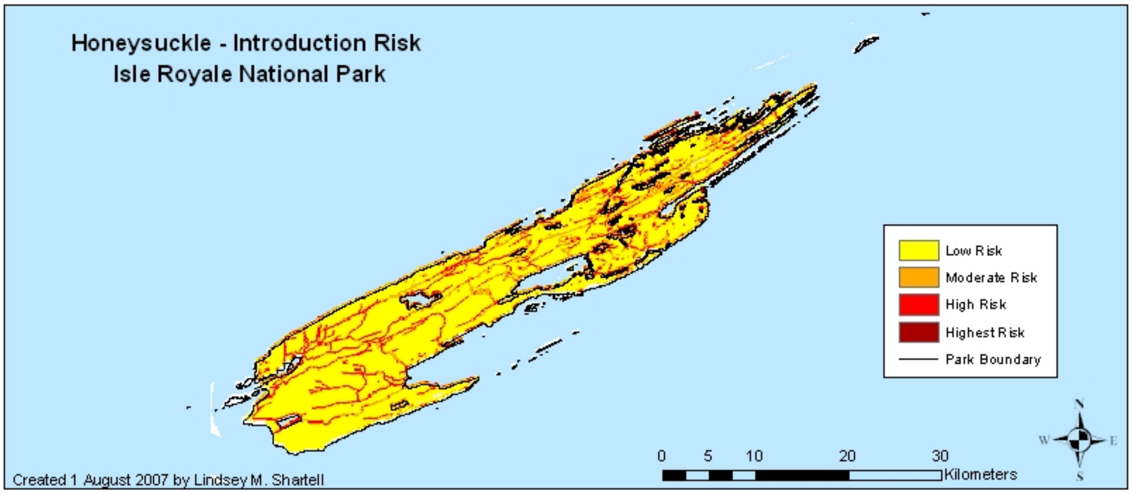
Appendix 3. Cont.

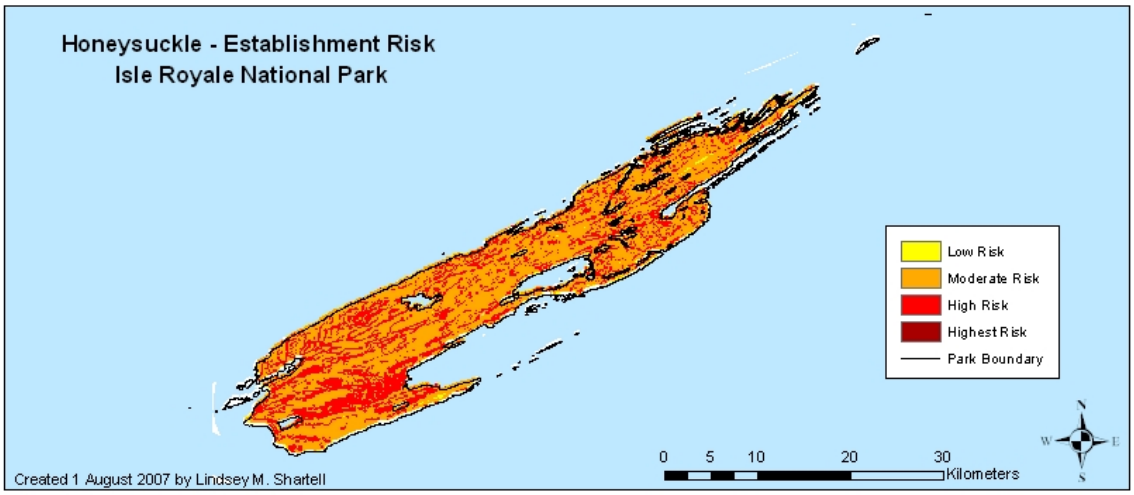


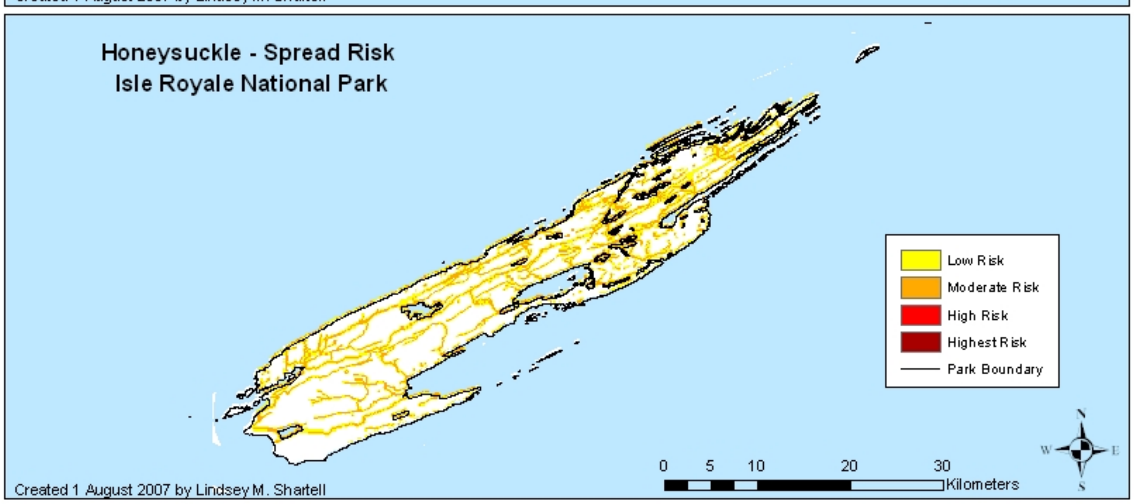




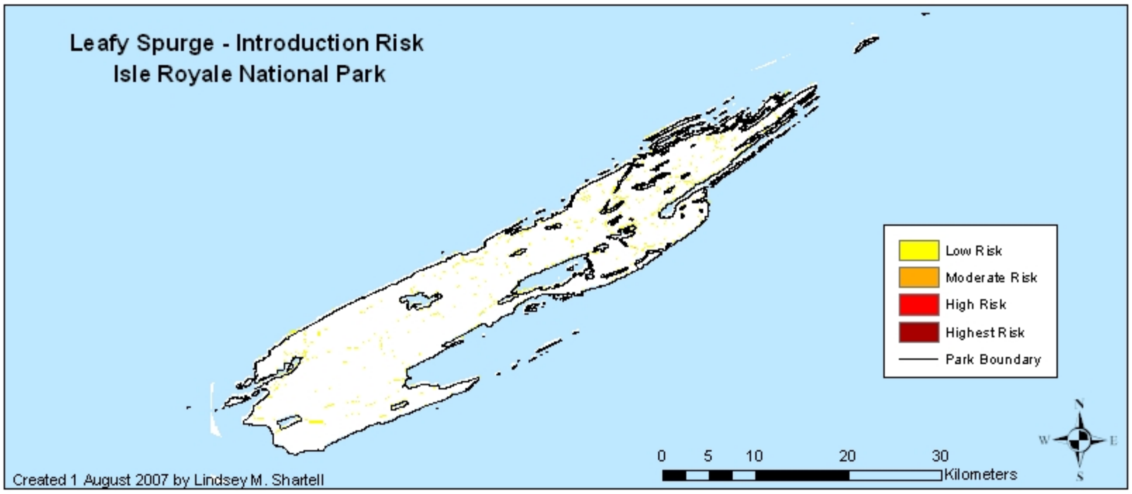
Appendix 3. Cont.

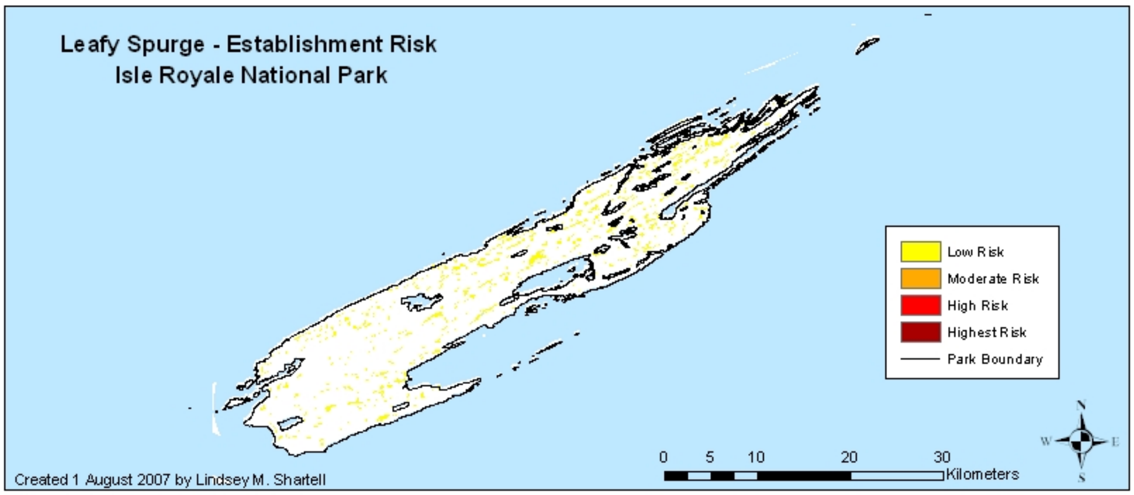


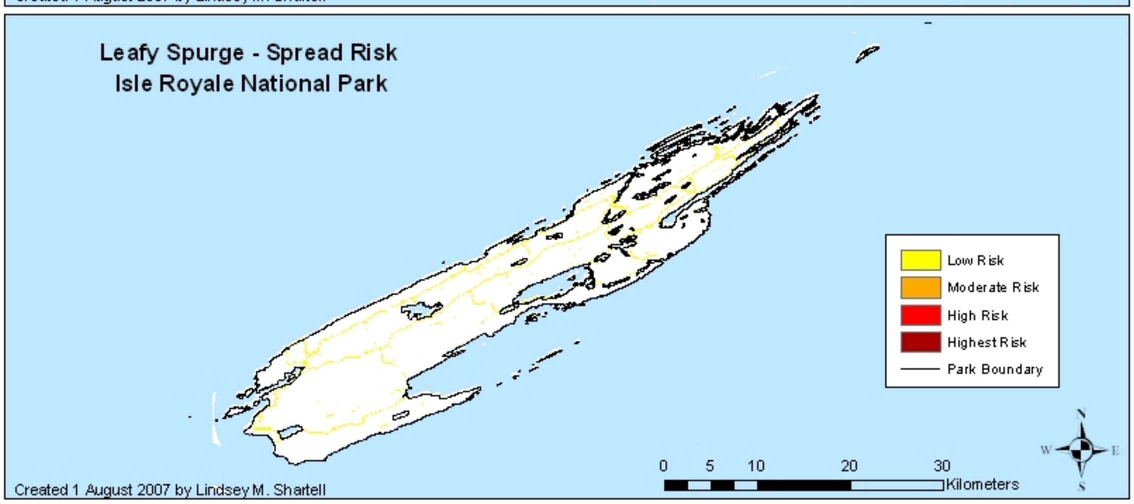




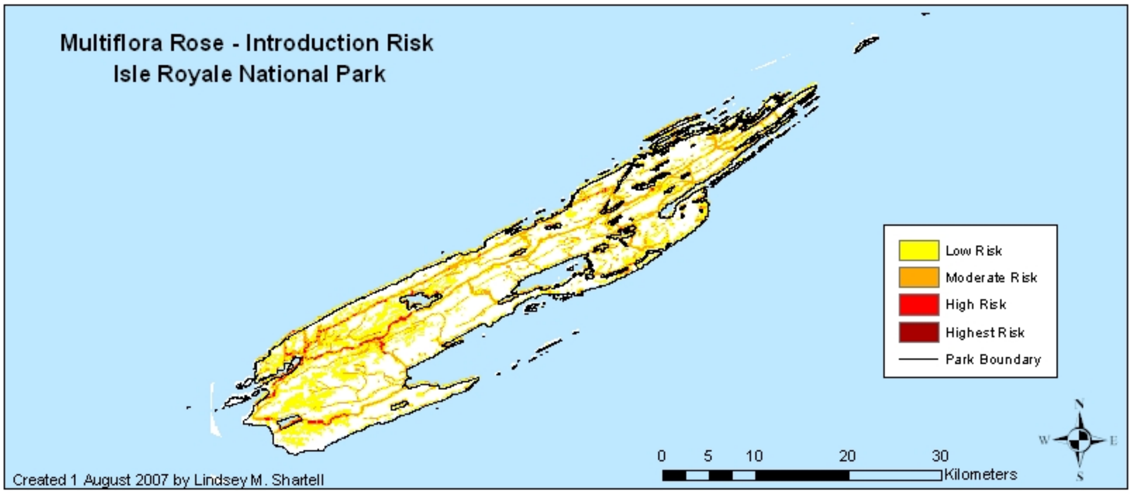
Appendix 3. Cont.

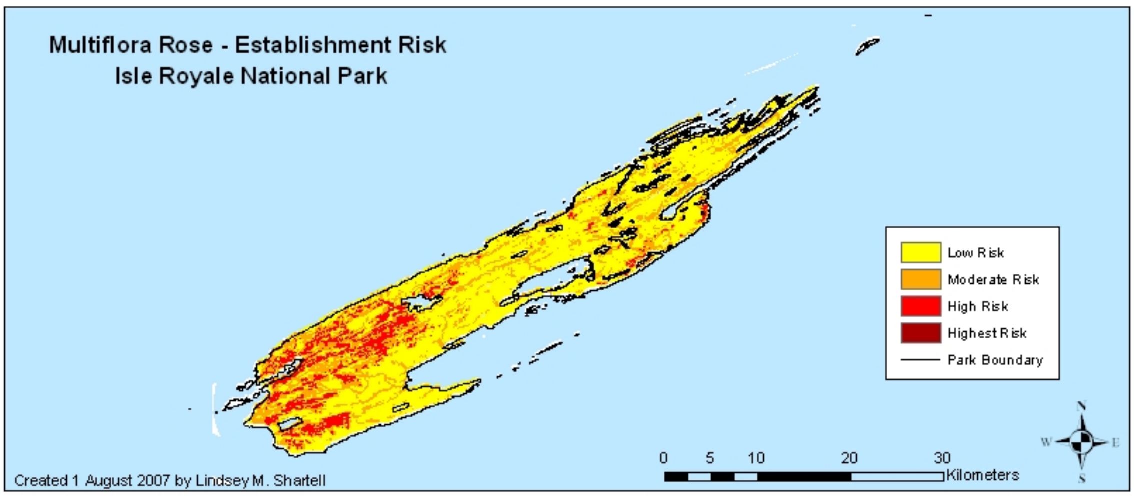


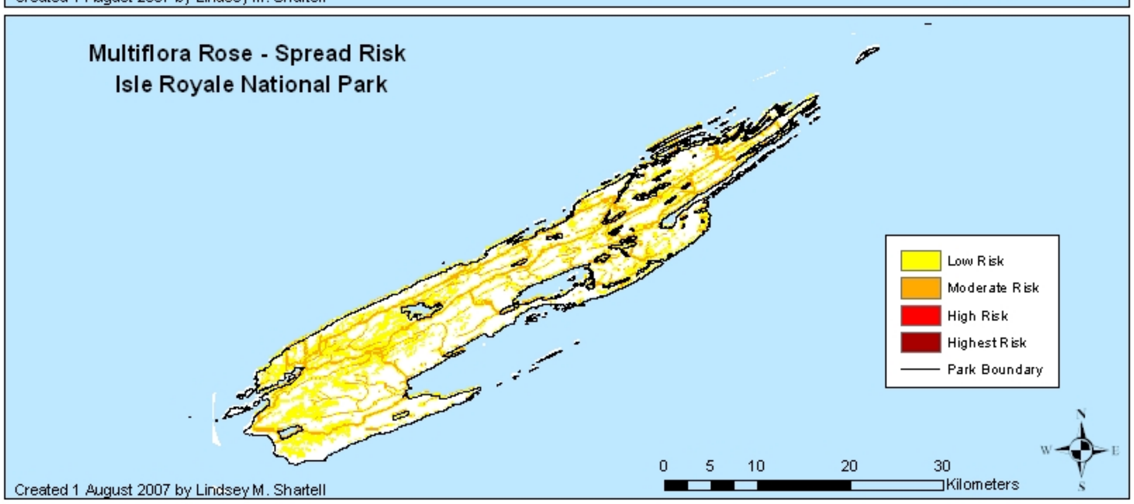




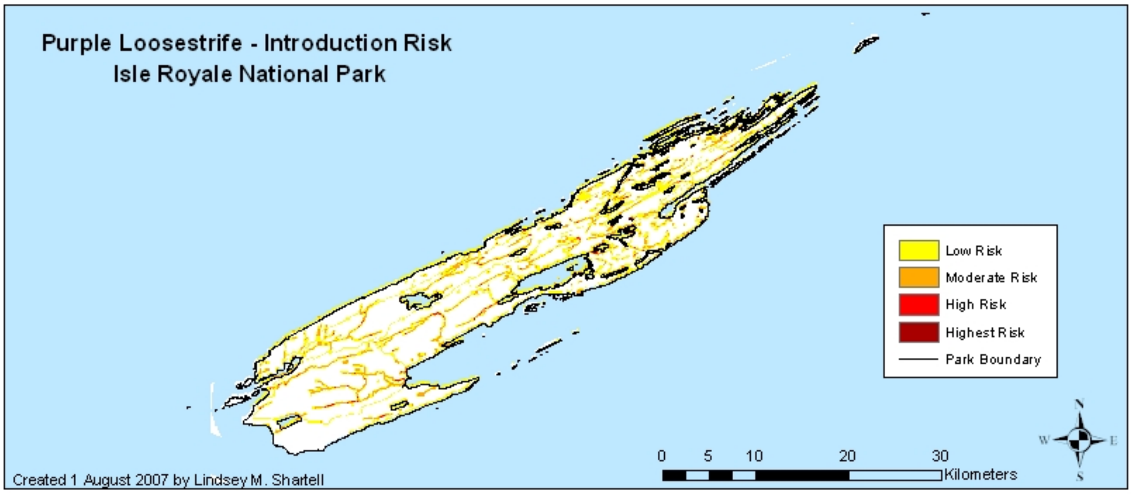
Appendix 3. Cont.

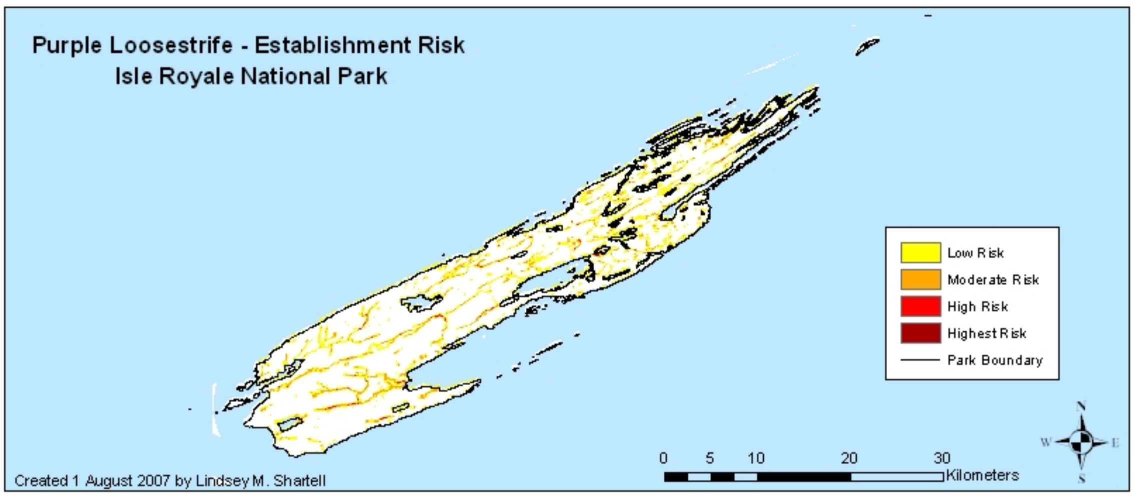


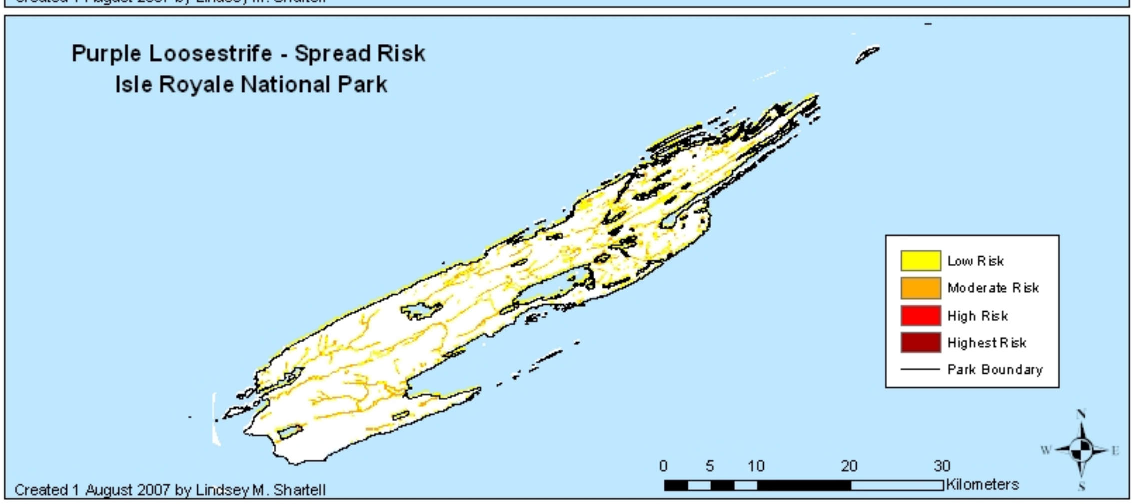




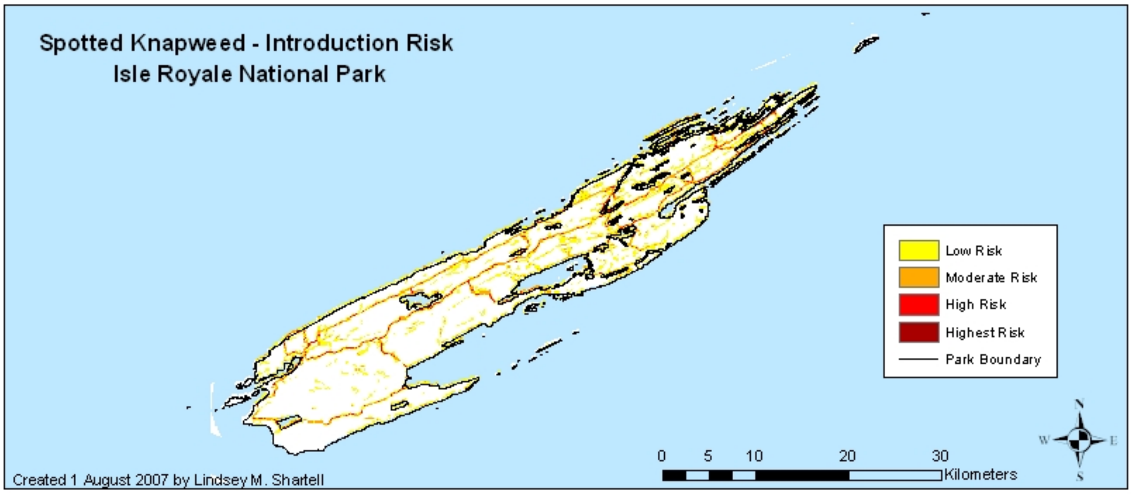
Appendix 3. Cont.

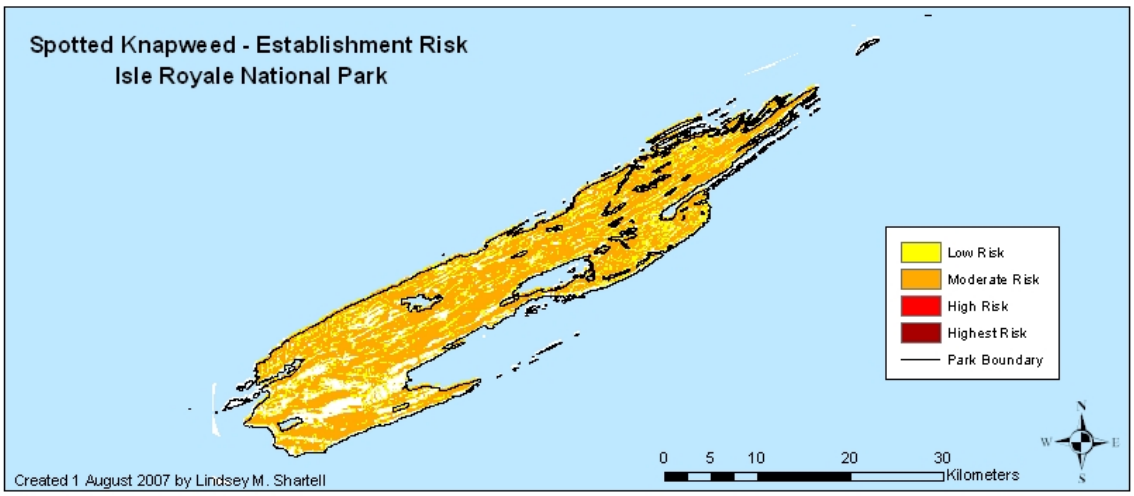


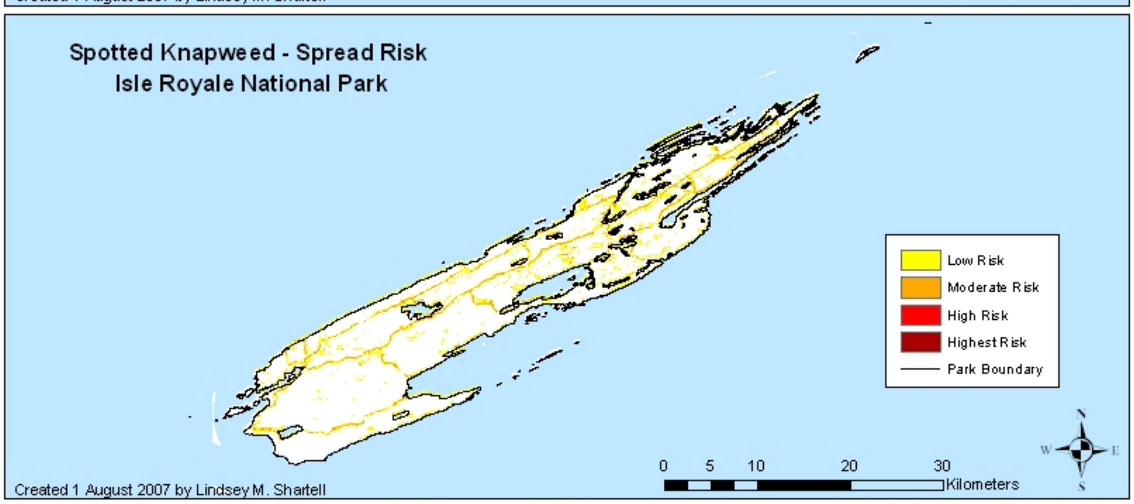




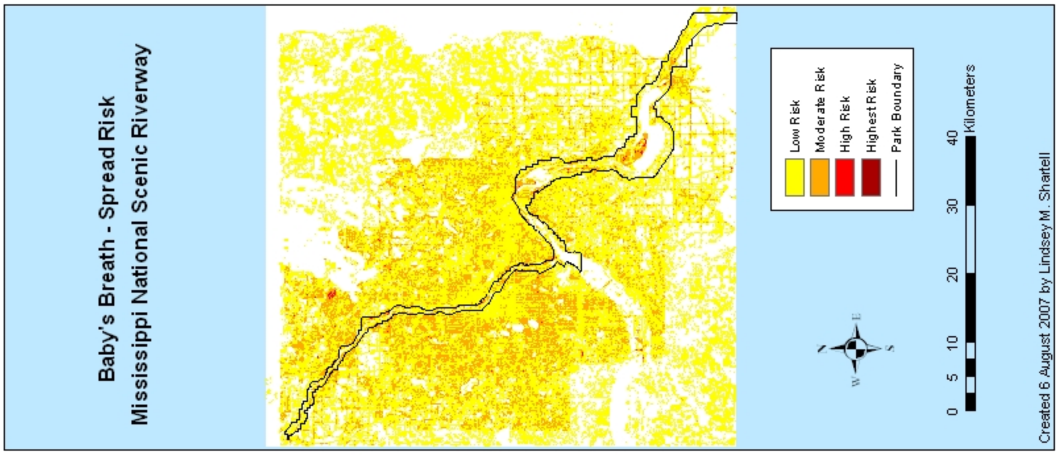
Appendix 3. Cont.

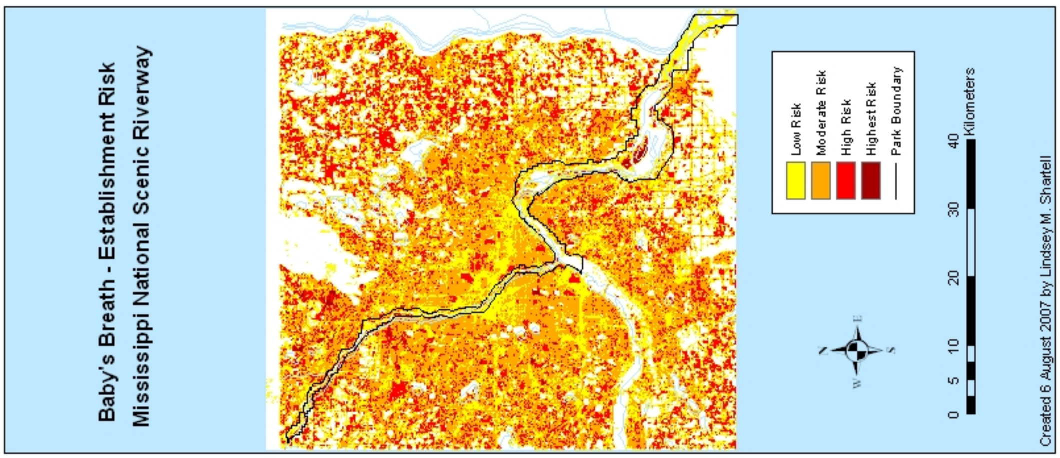


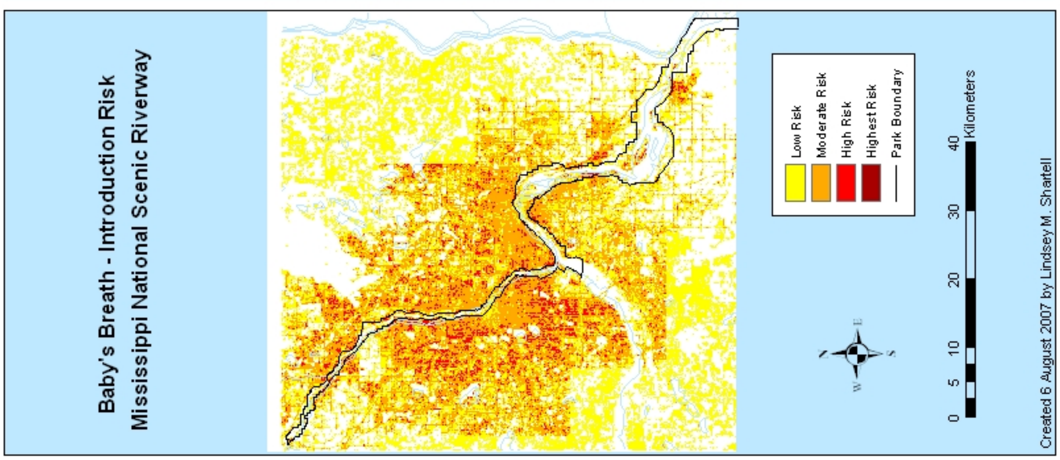




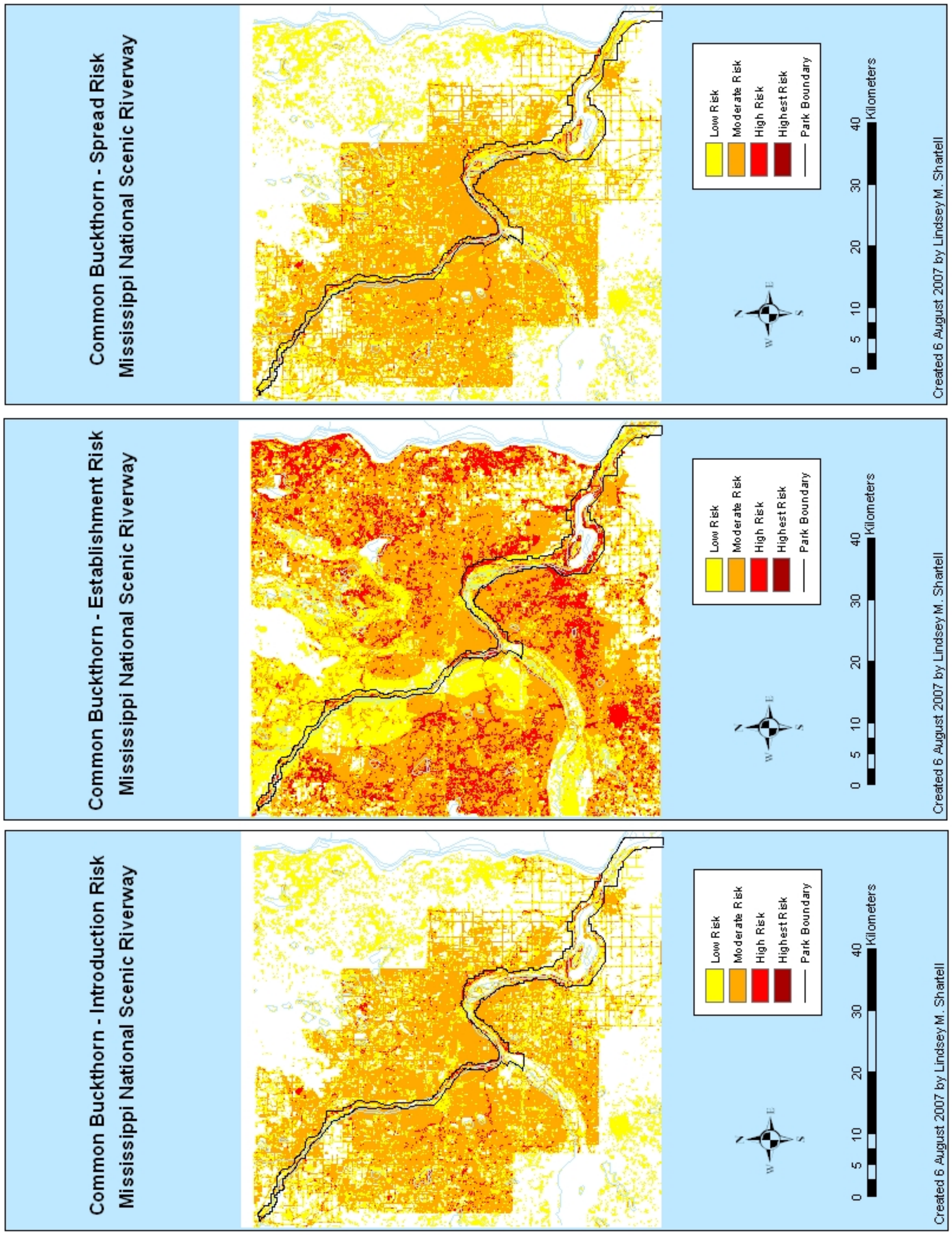
Appendix 3. Cont.



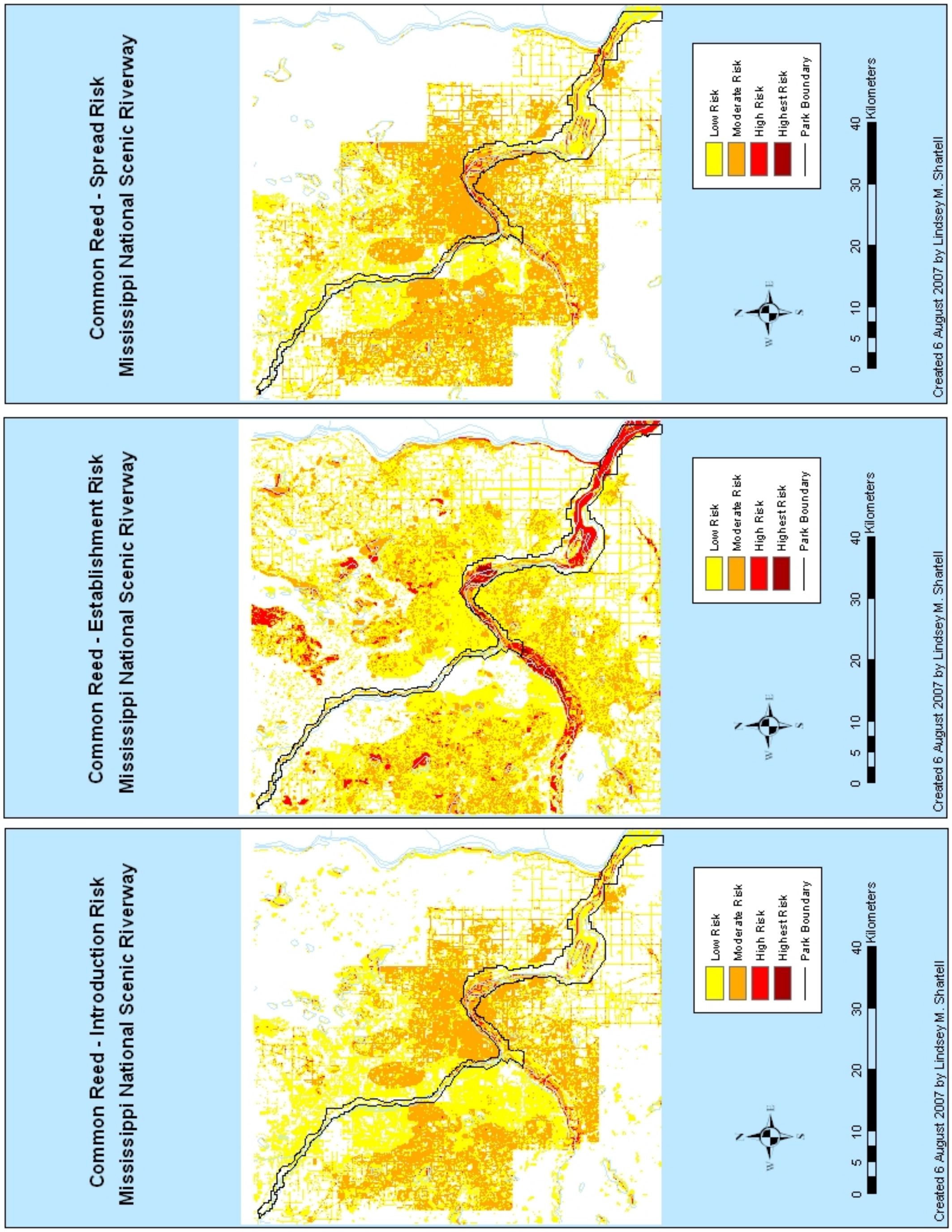




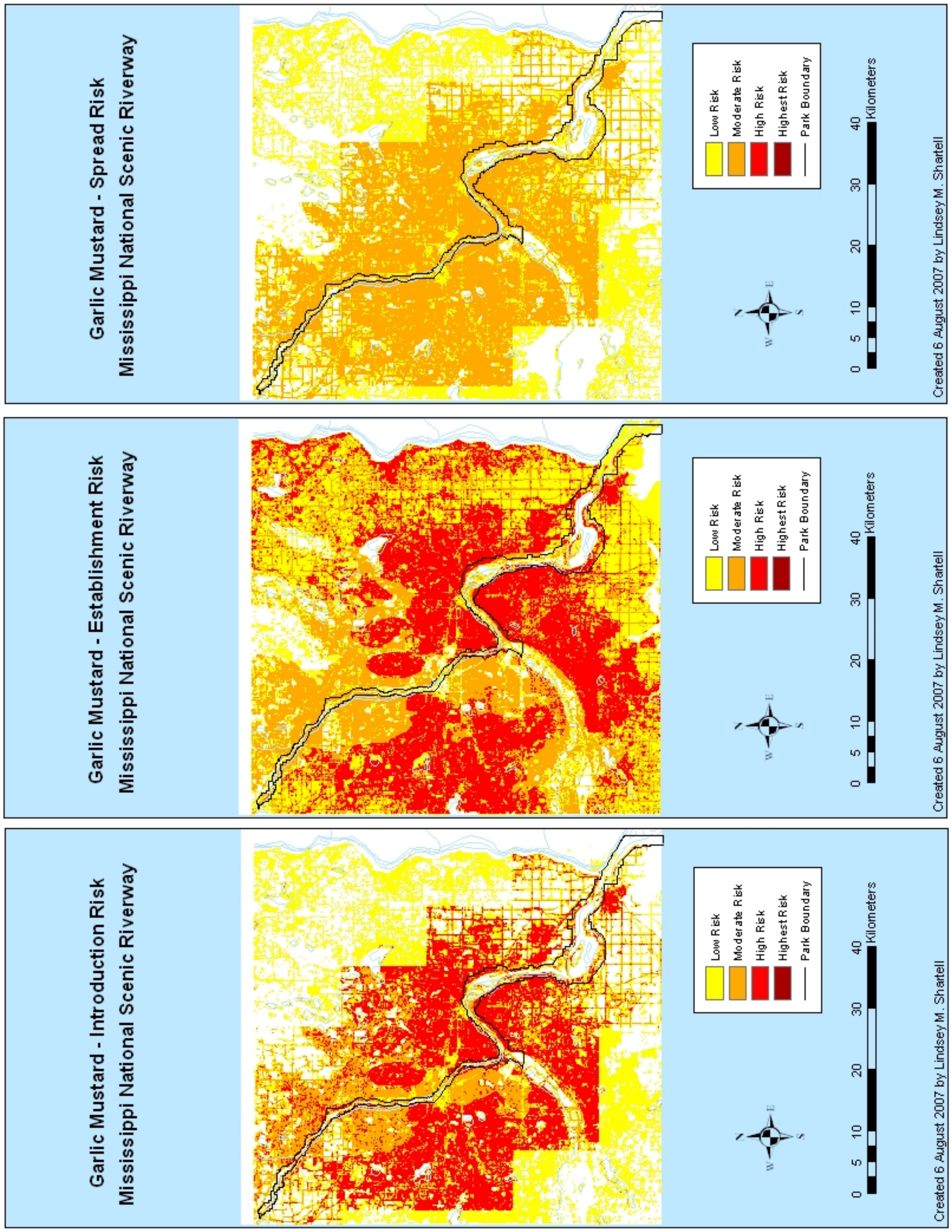
Appendix 3. Cont.



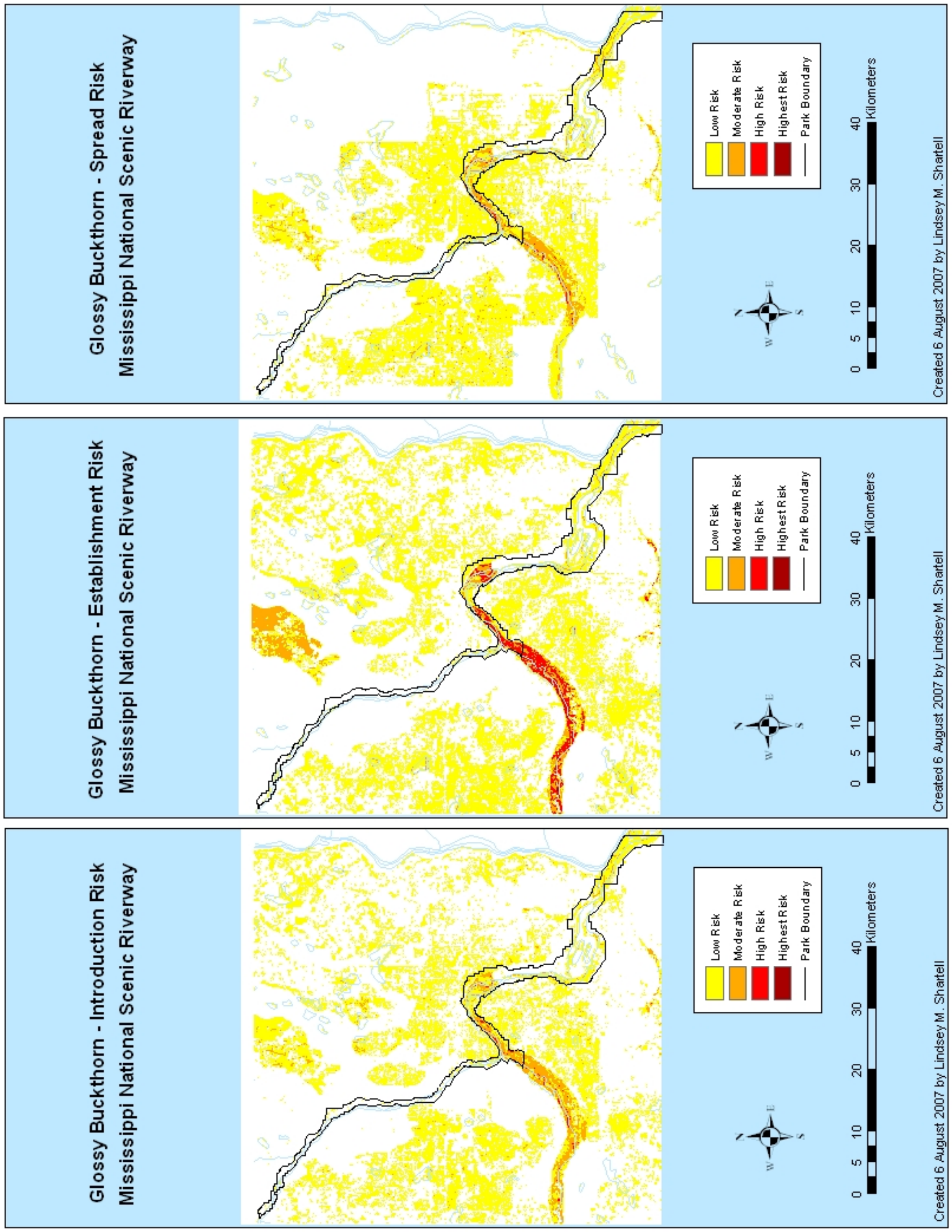
Appendix 3. Cont.



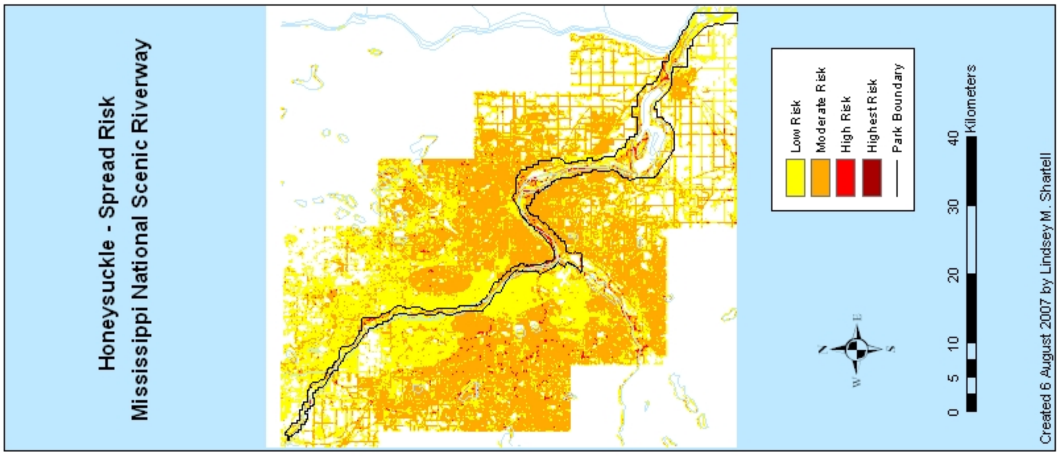
Appendix 3. Cont.

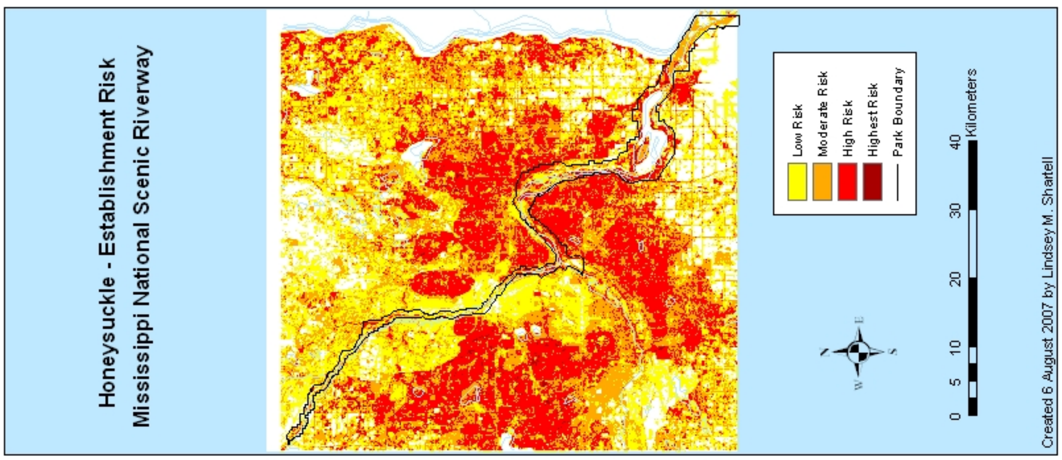


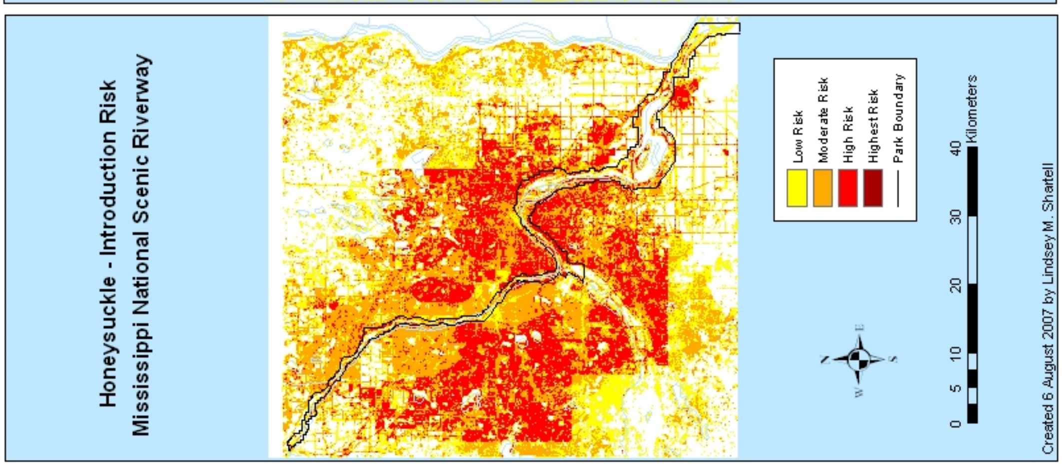
Appendix 3. Cont.



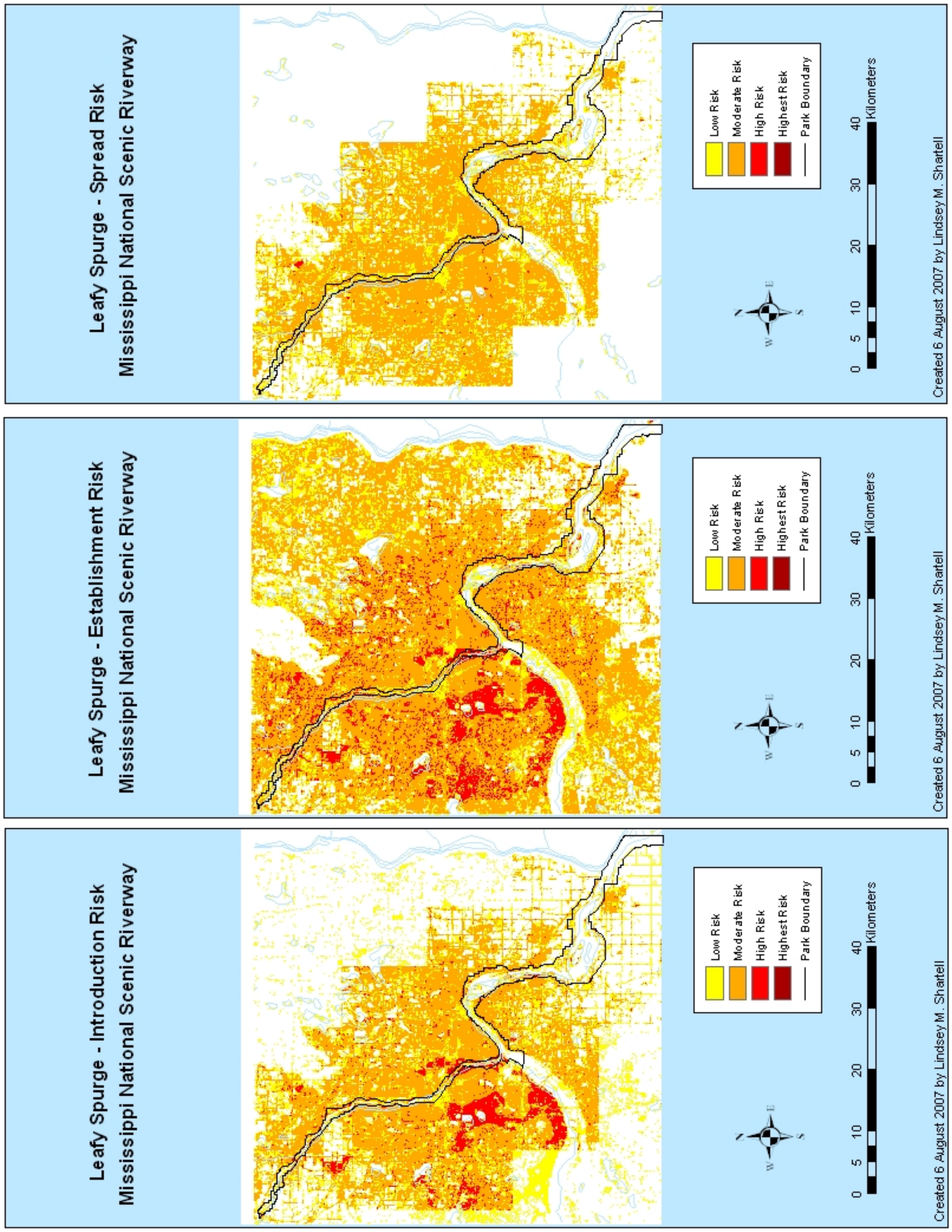
Appendix 3. Cont.



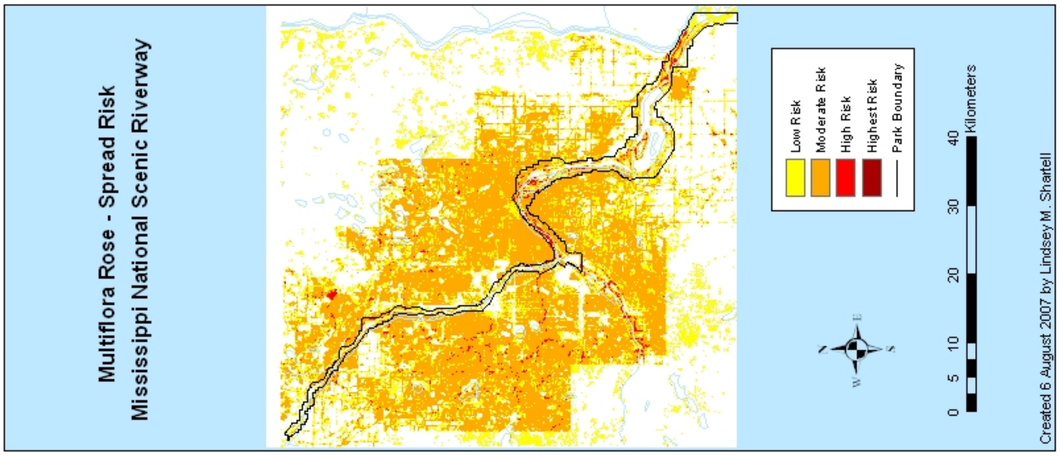


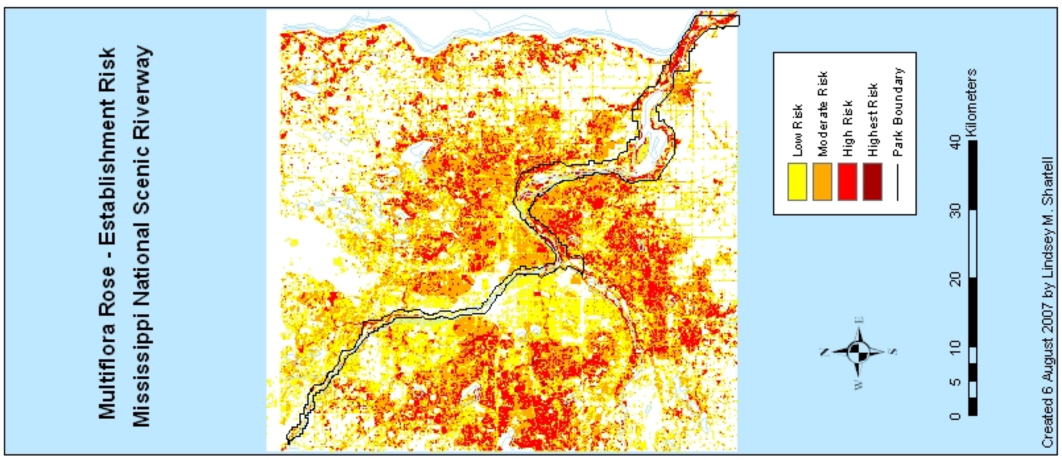


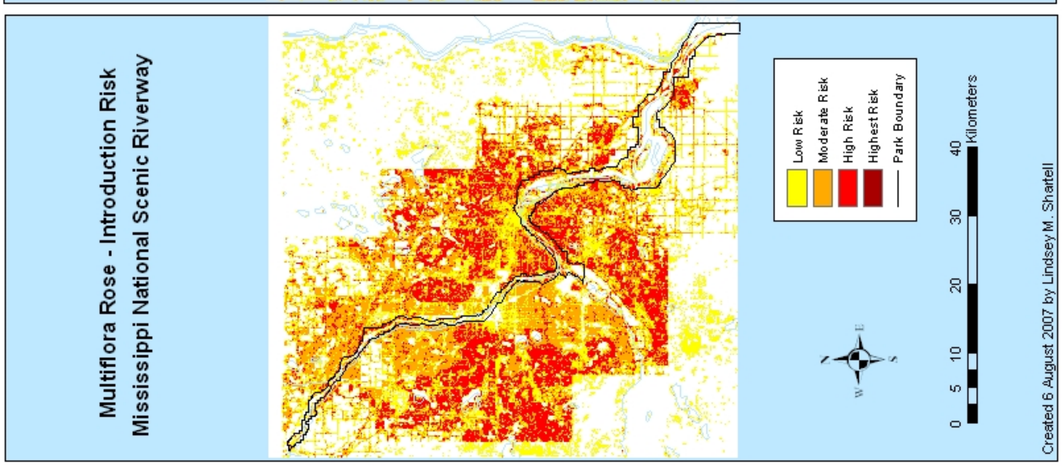
Appendix 3. Cont.



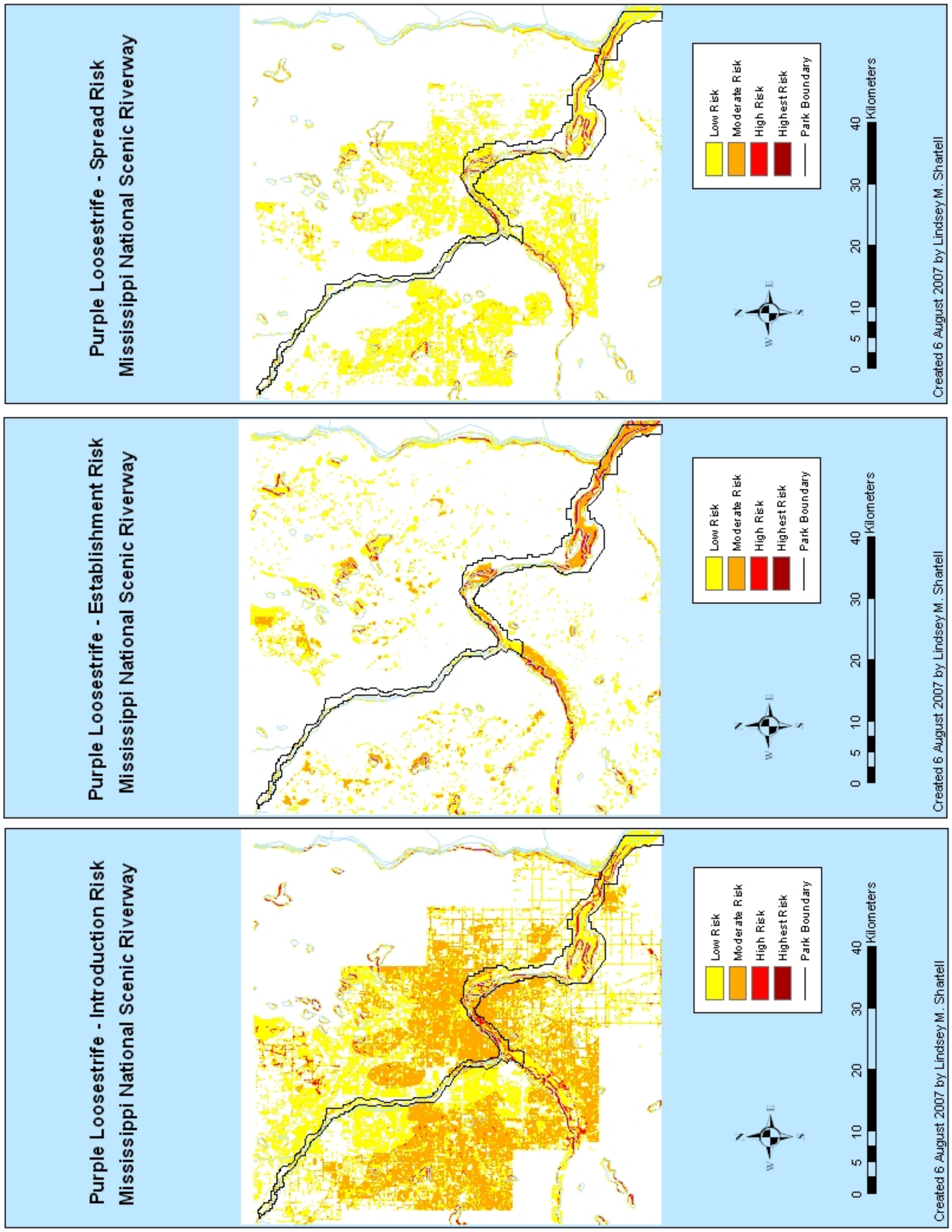
Appendix 3. Cont.



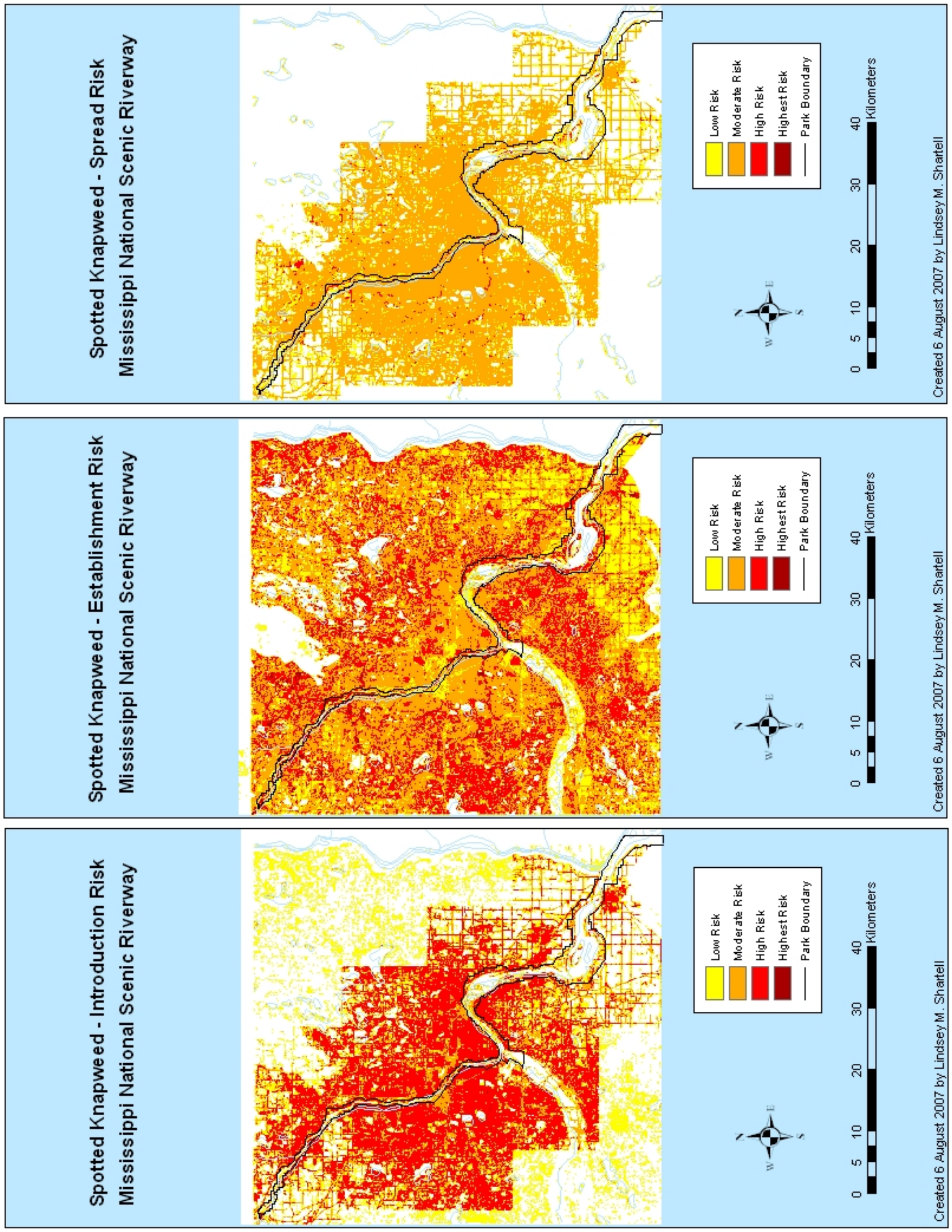




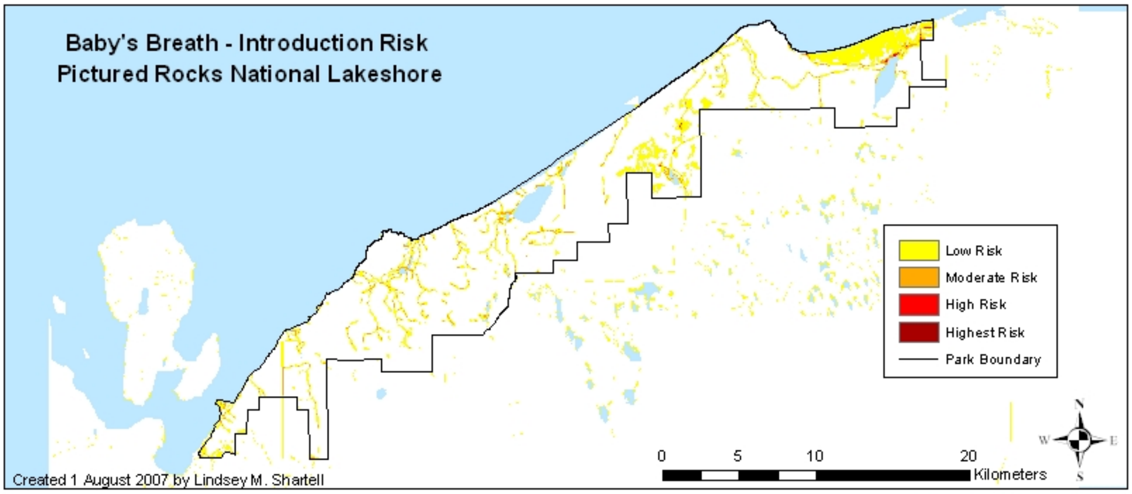
Appendix 3. Cont.

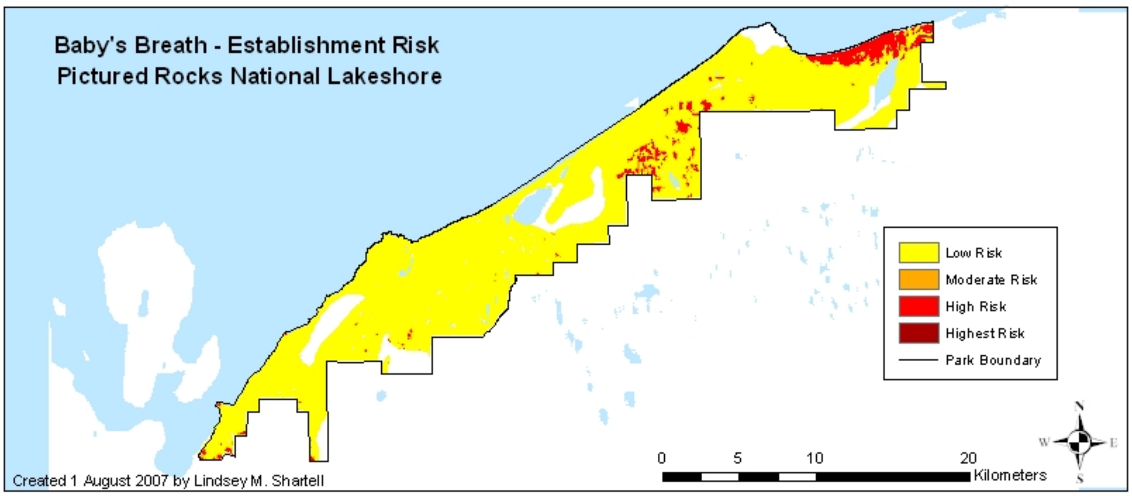


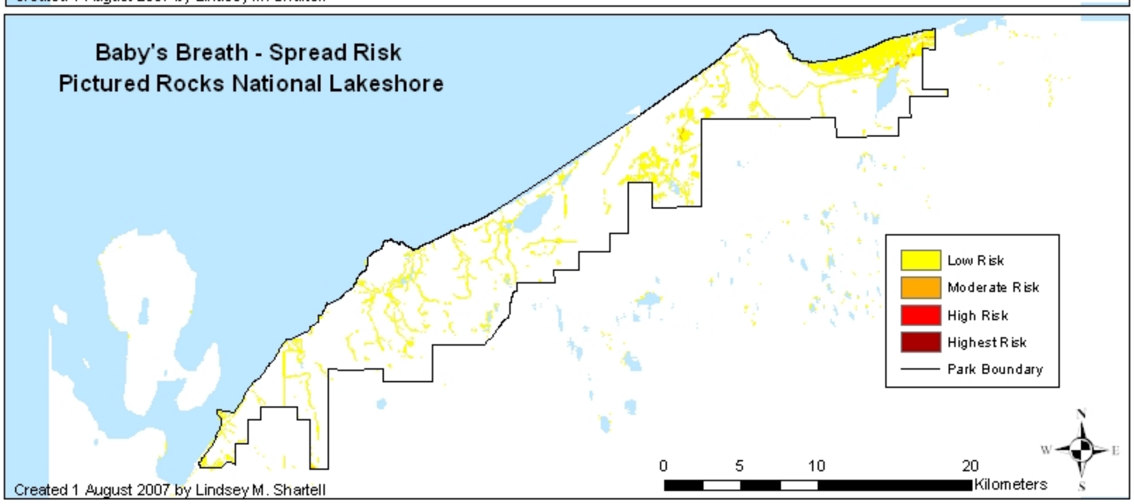
Appendix 3. Cont.



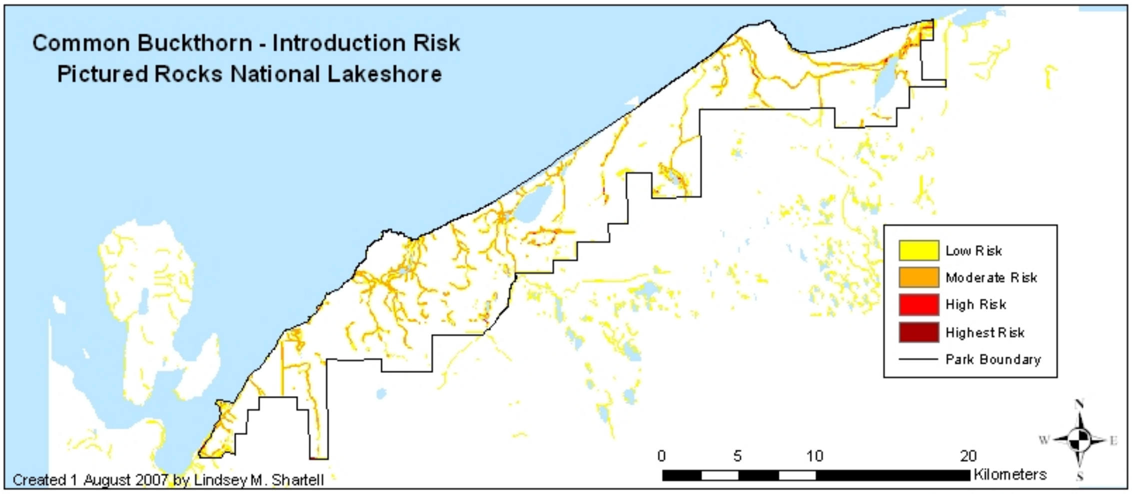
Appendix 3. Cont.

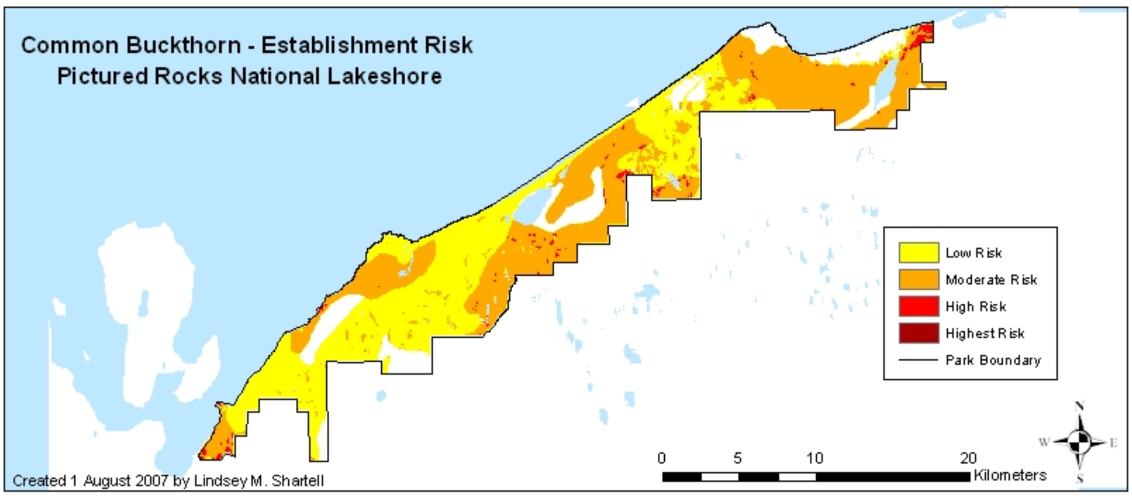


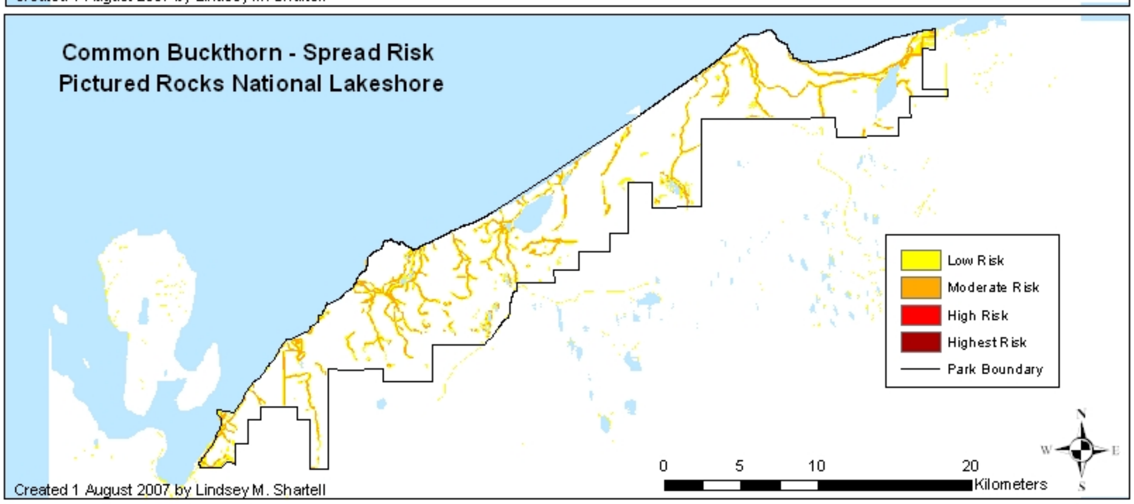




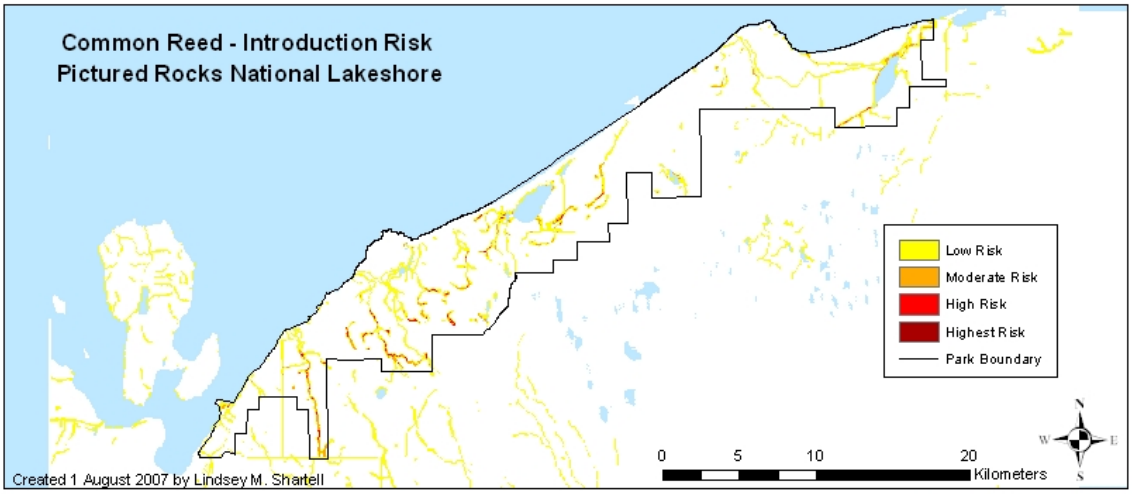
Appendix 3. Cont.

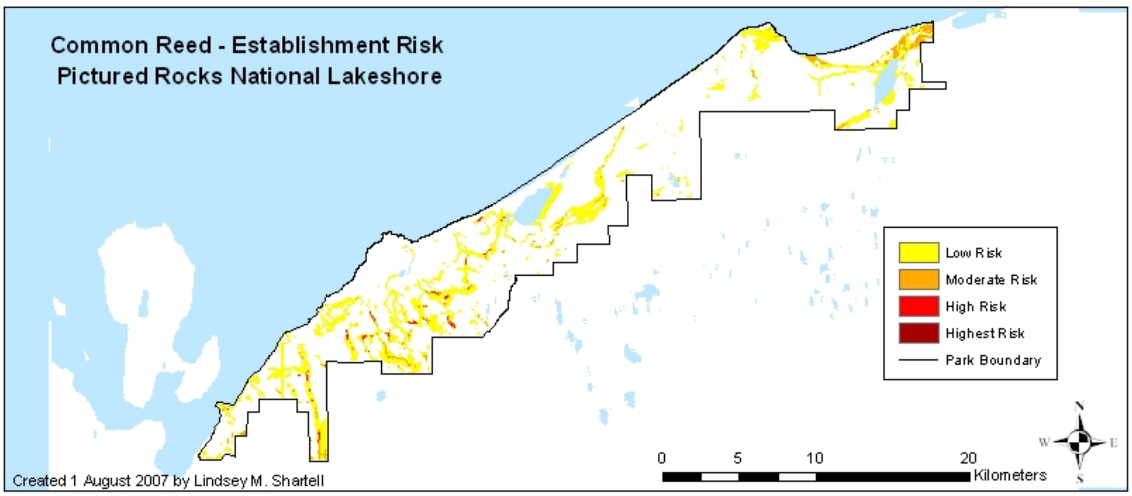


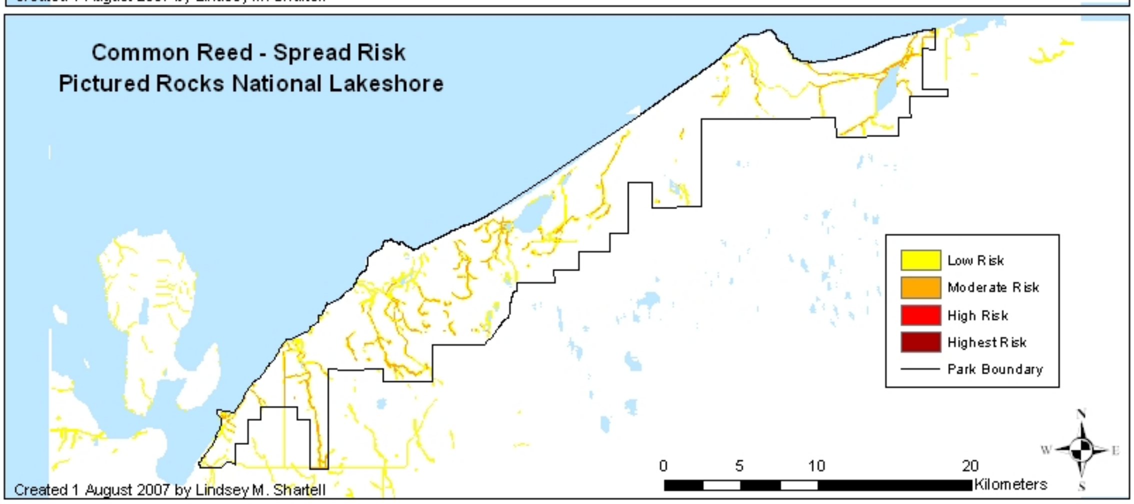




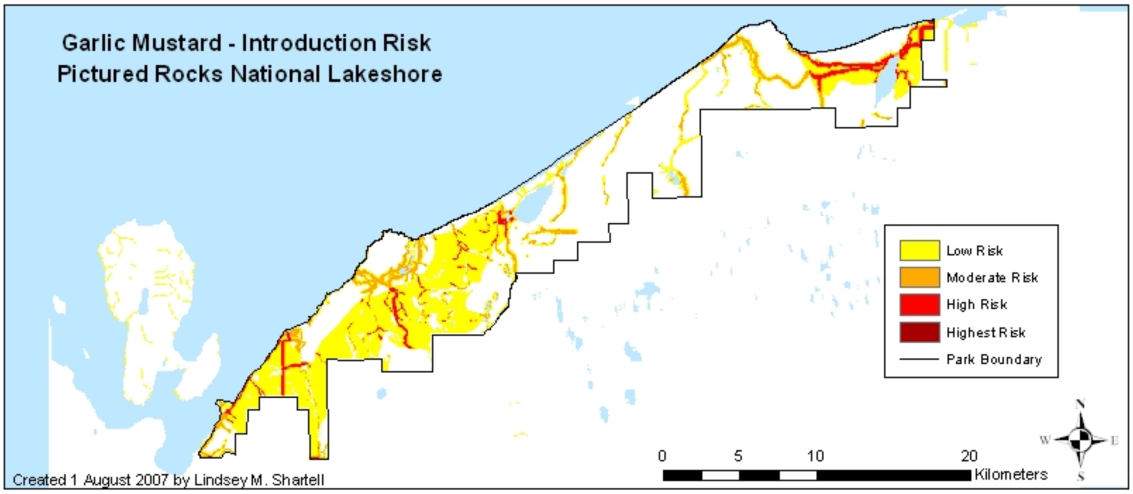
Appendix 3. Cont.

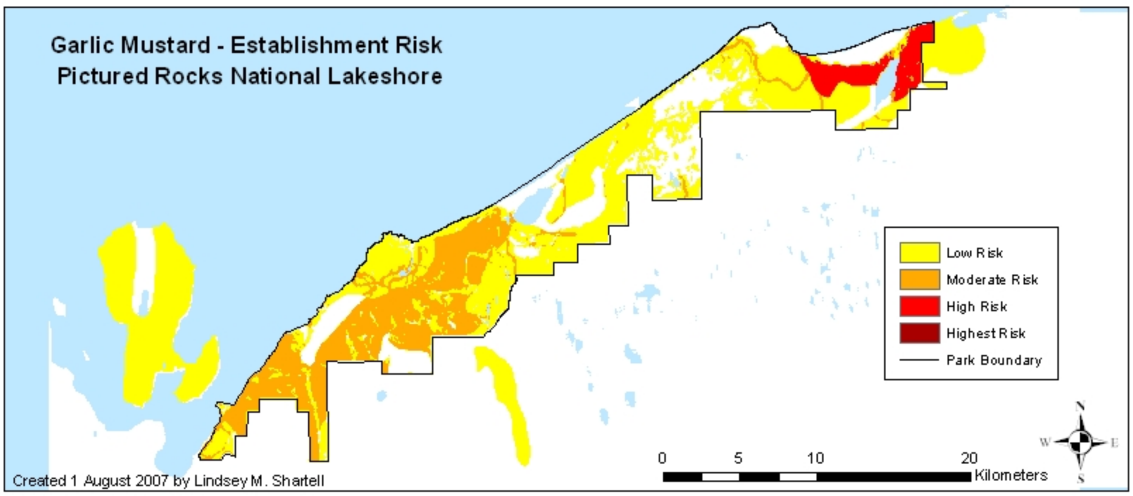


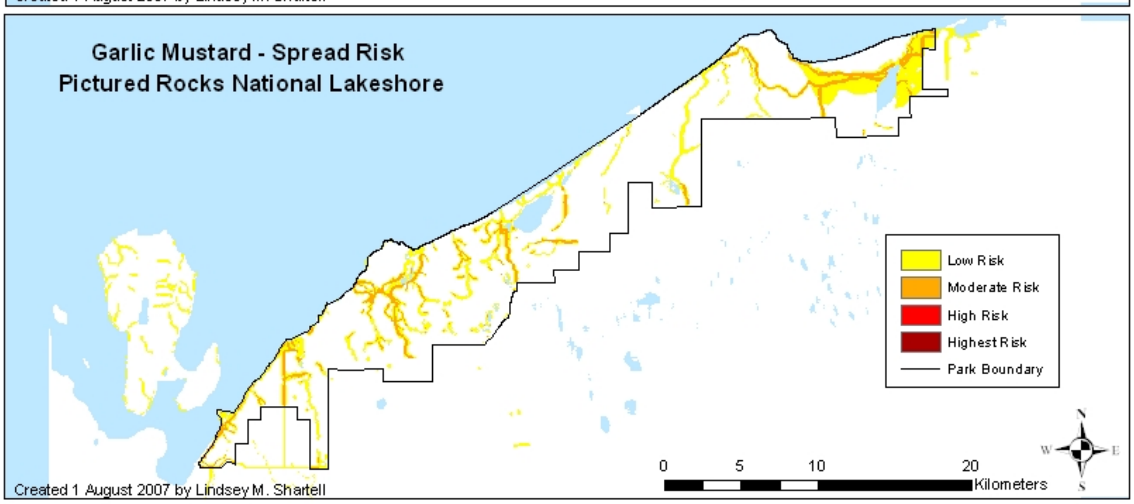




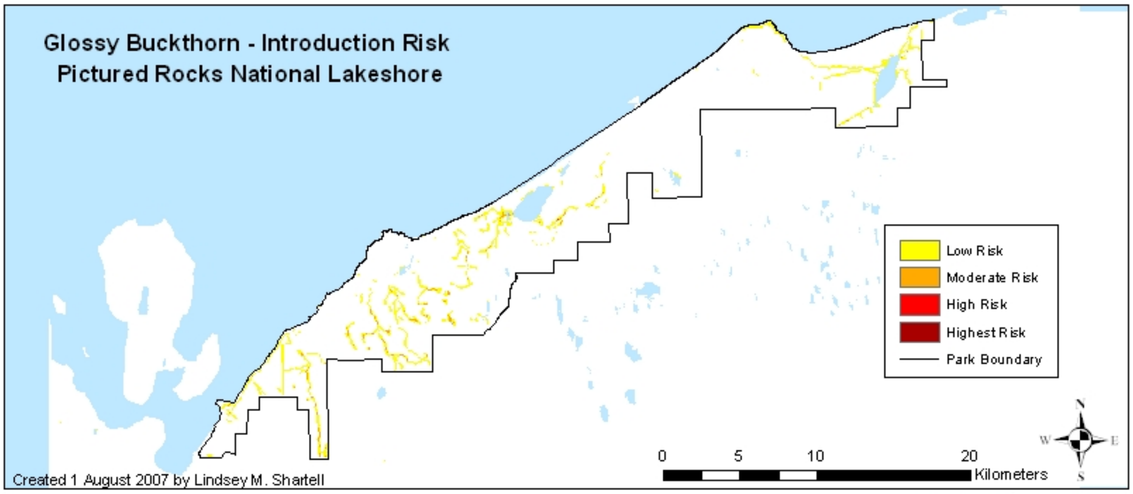
Appendix 3. Cont.

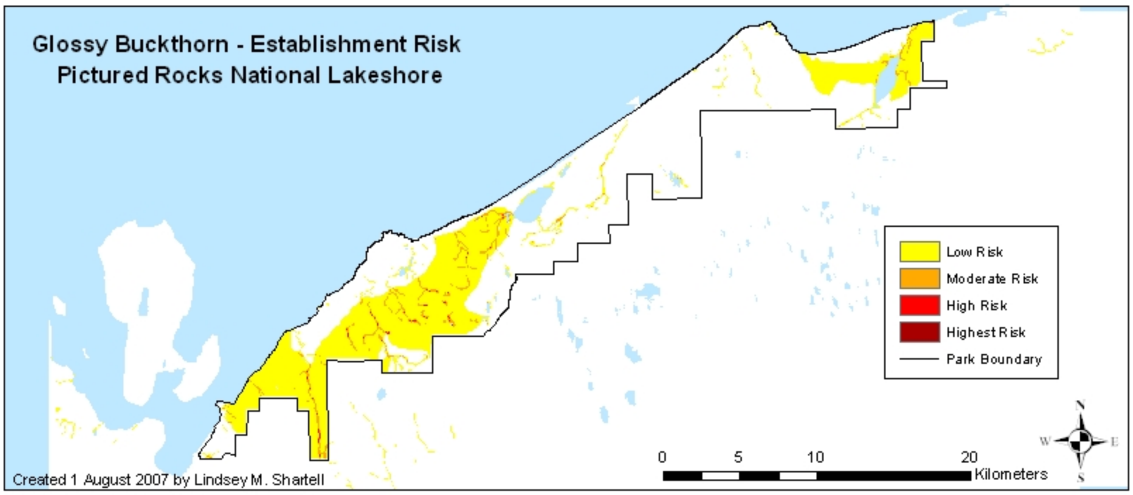


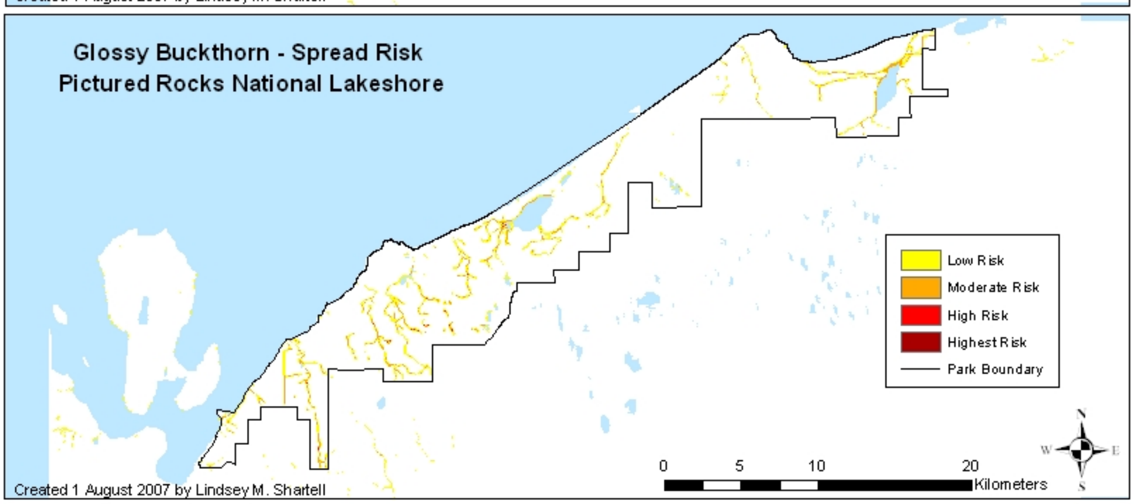




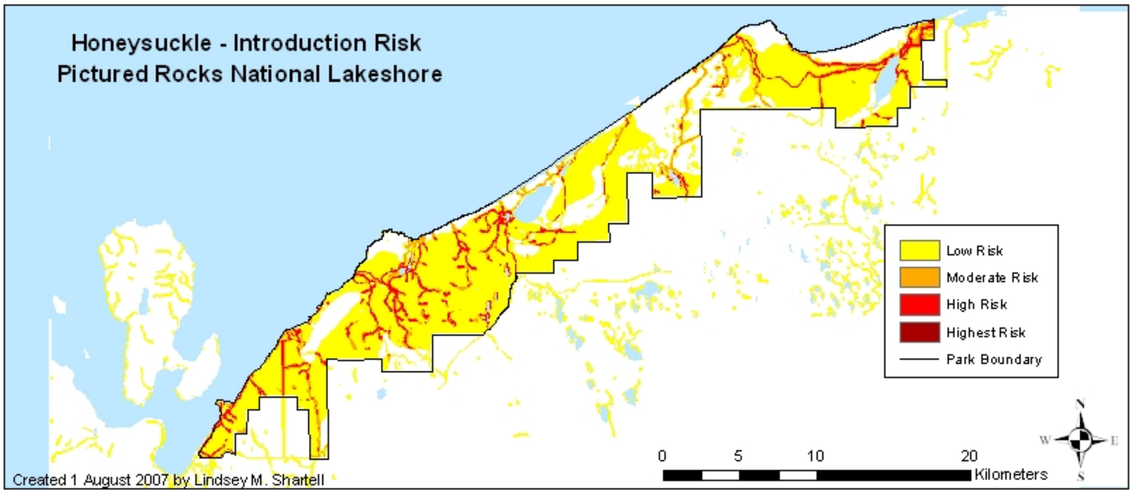
Appendix 3. Cont.

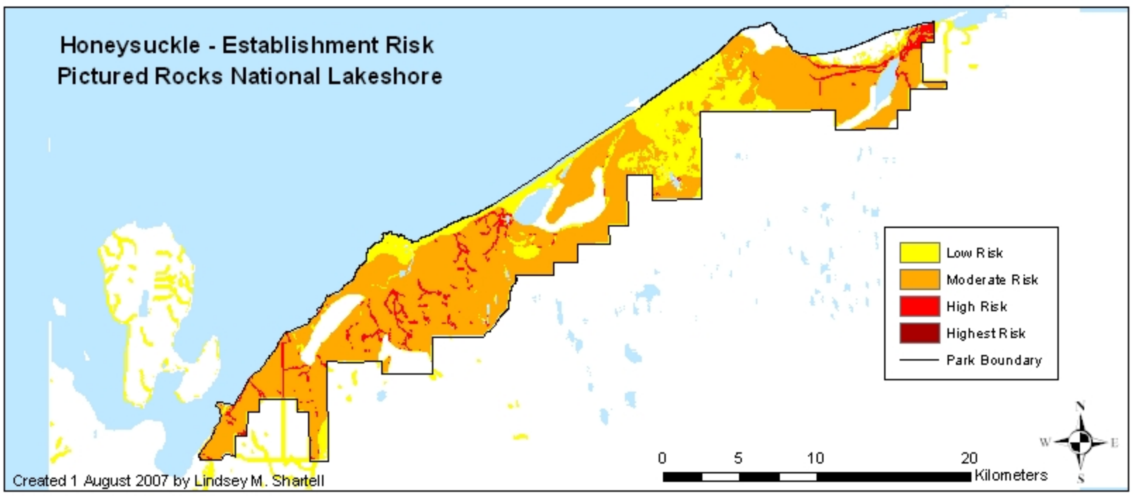


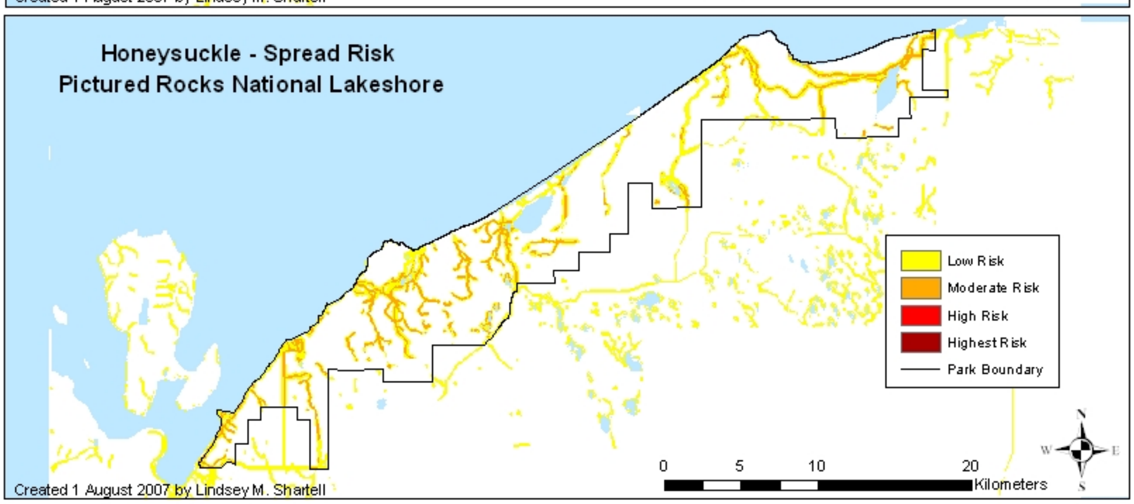




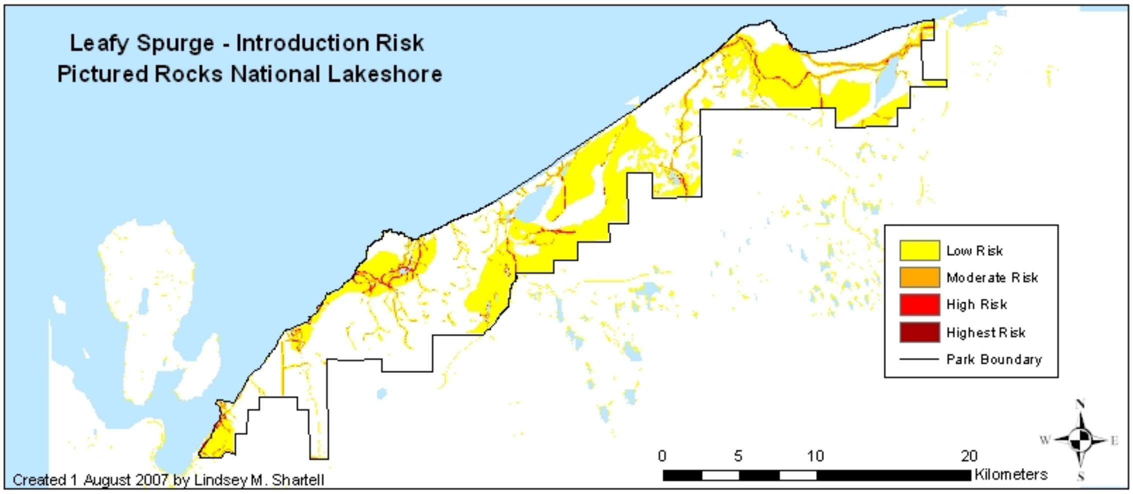
Appendix 3. Cont.

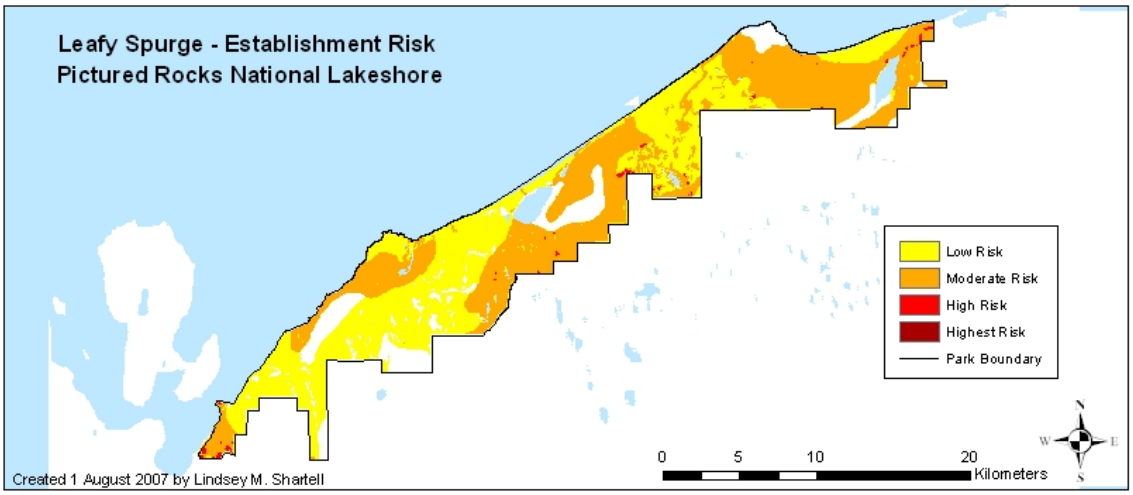


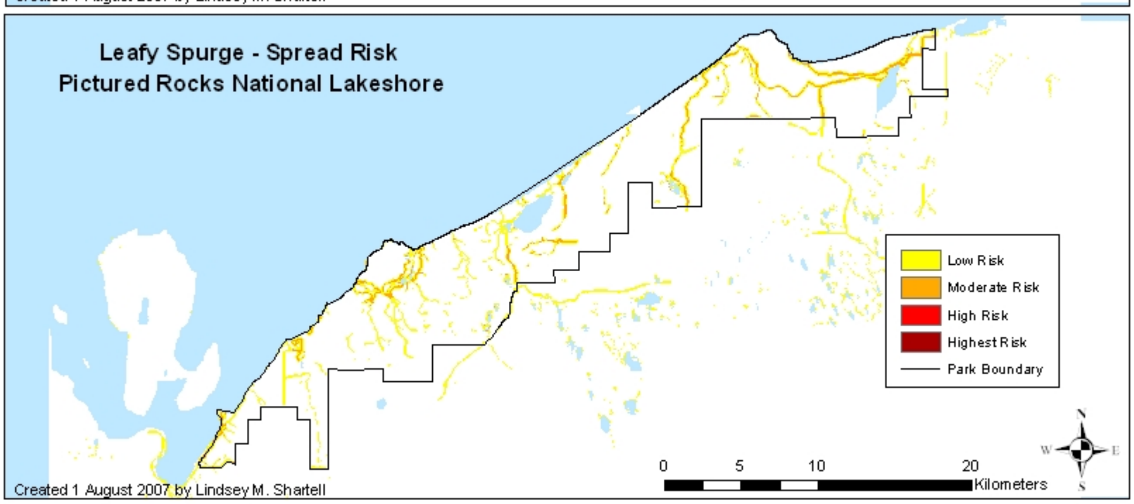




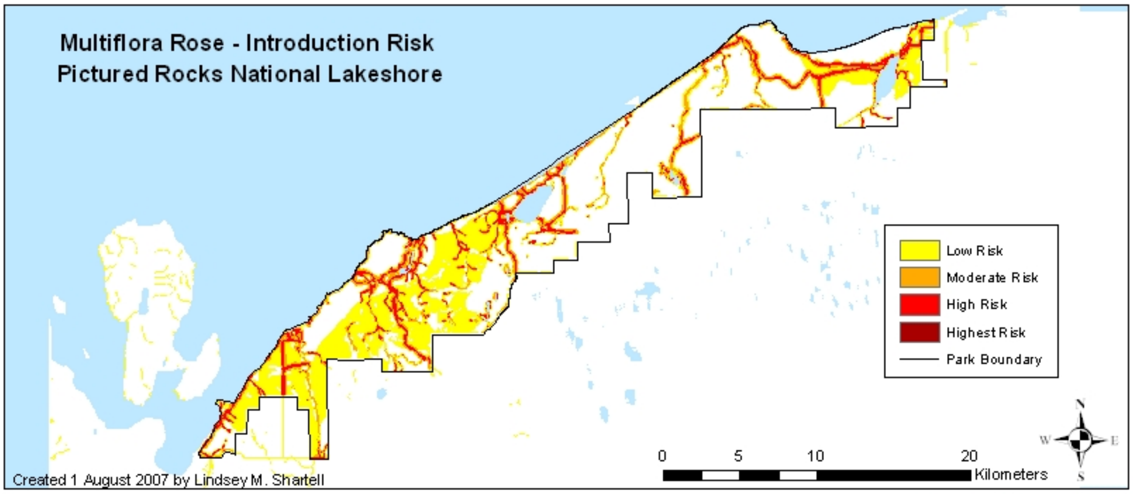
Appendix 3. Cont.

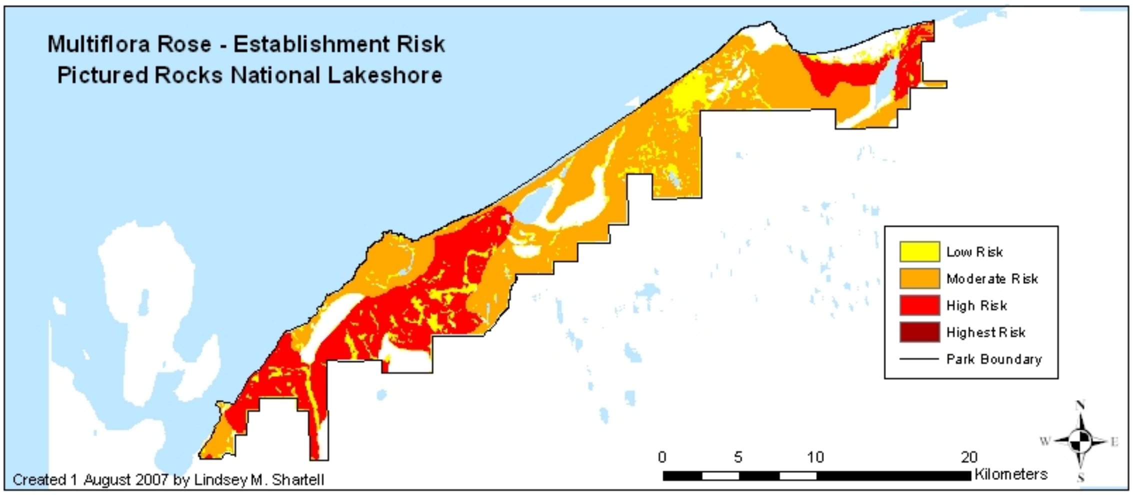


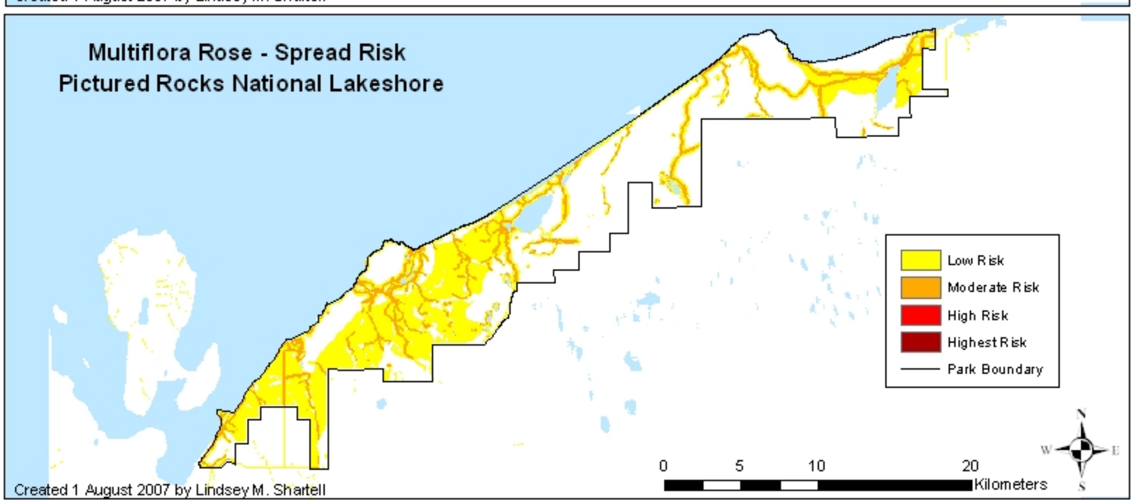




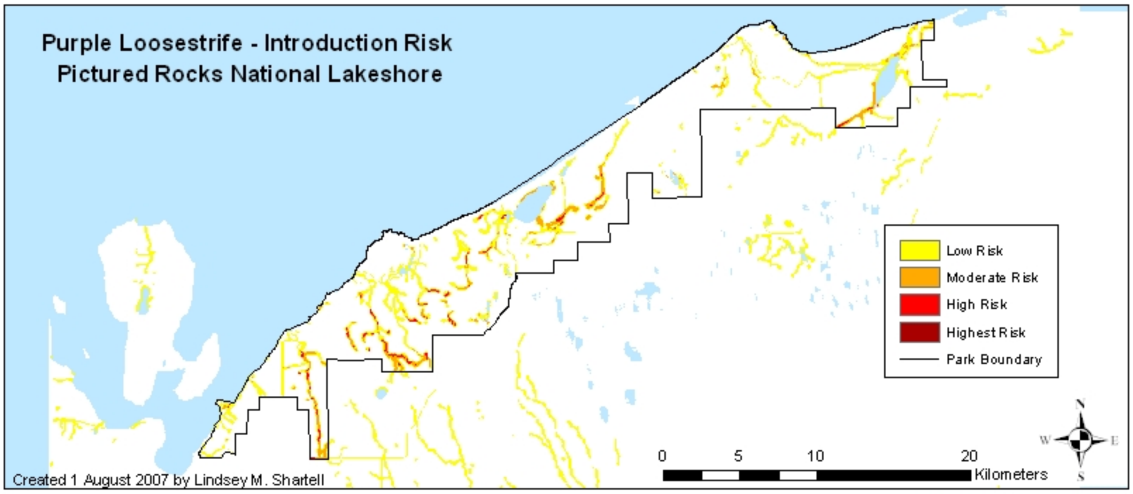
Appendix 3. Cont.

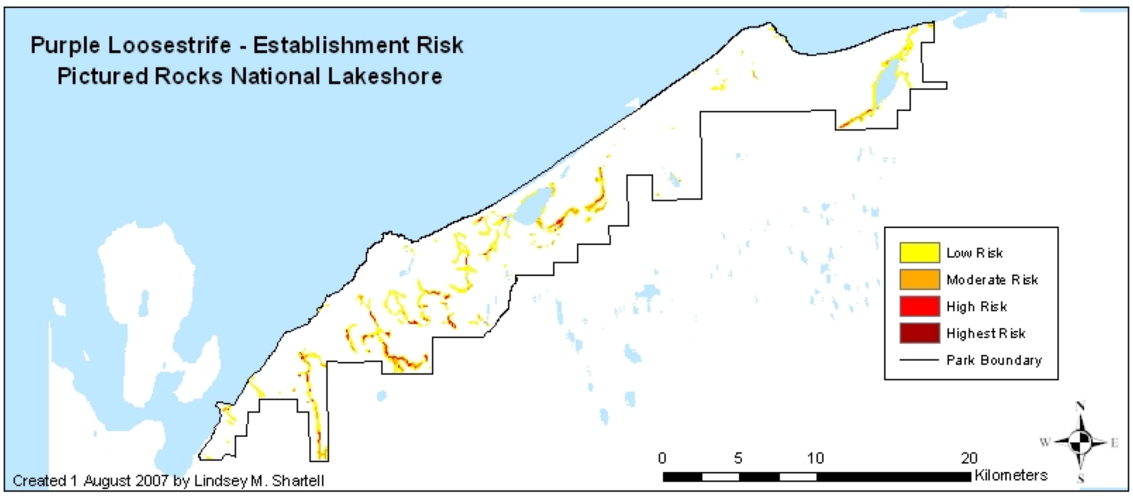


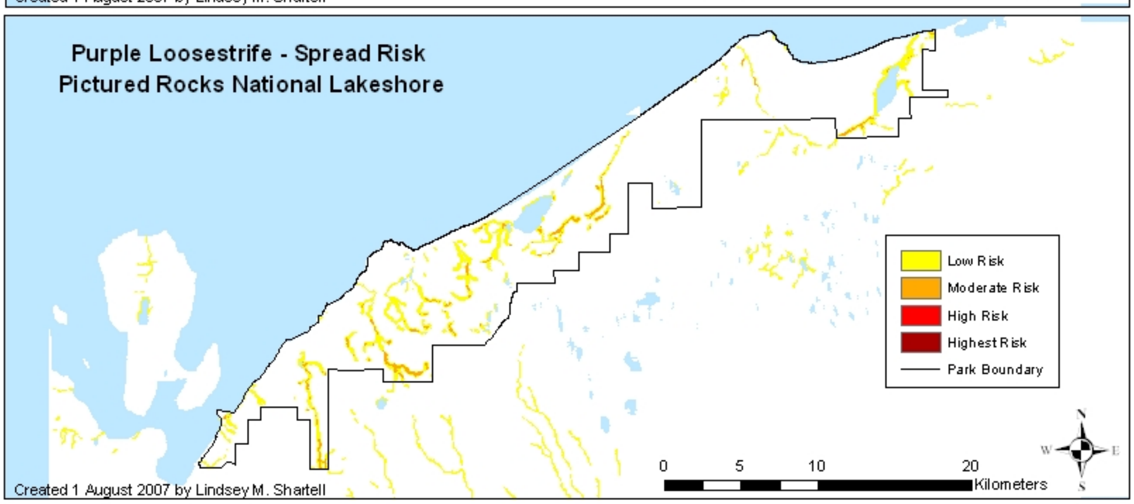




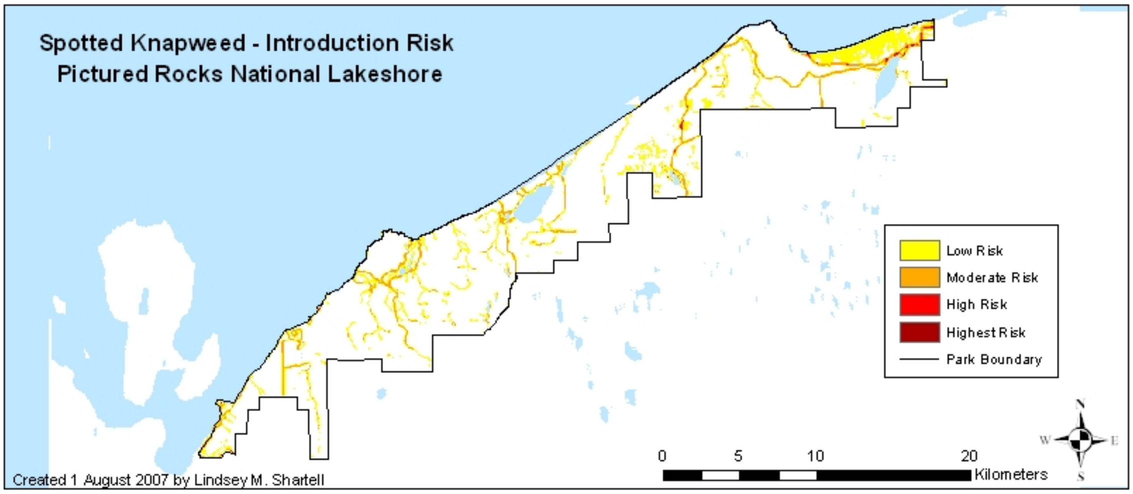
Appendix 3. Cont.

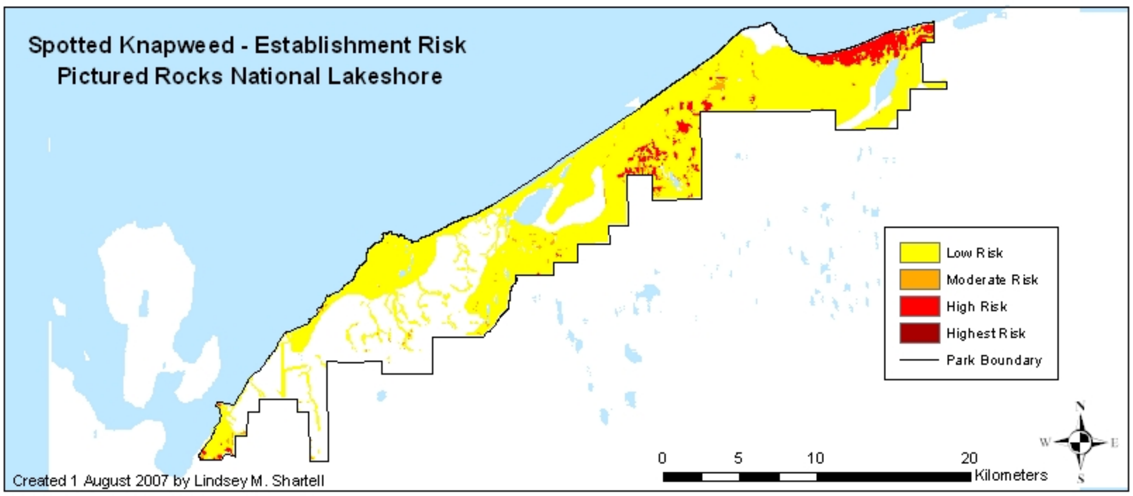


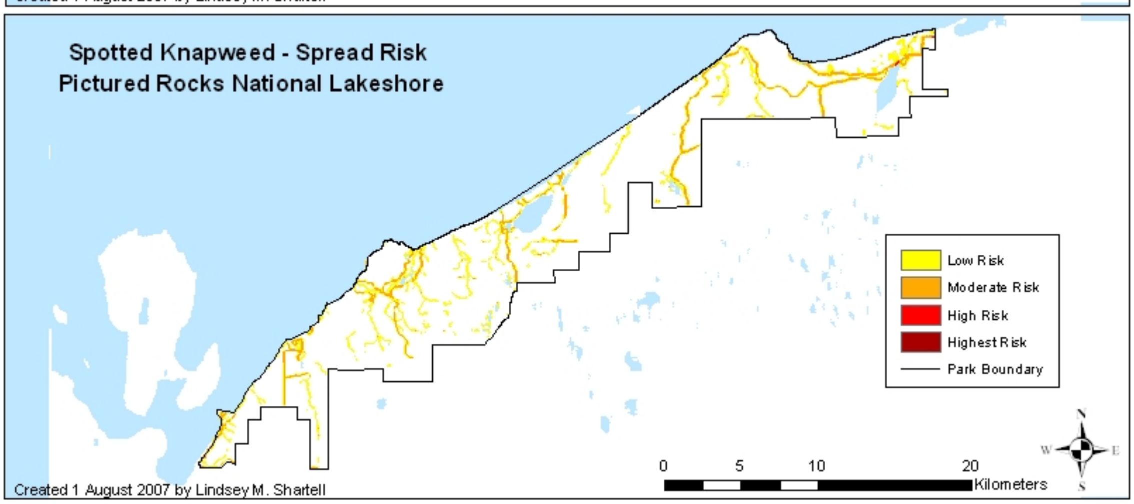




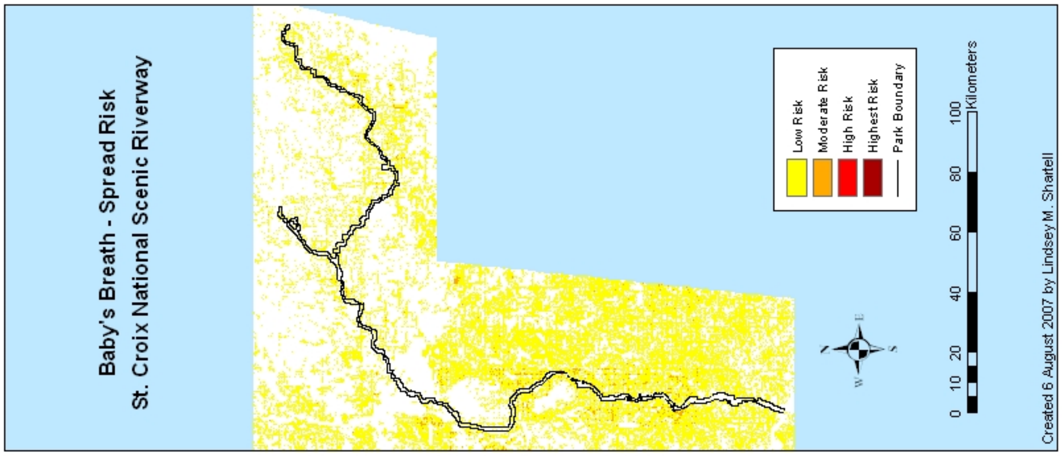
Appendix 3. Cont.

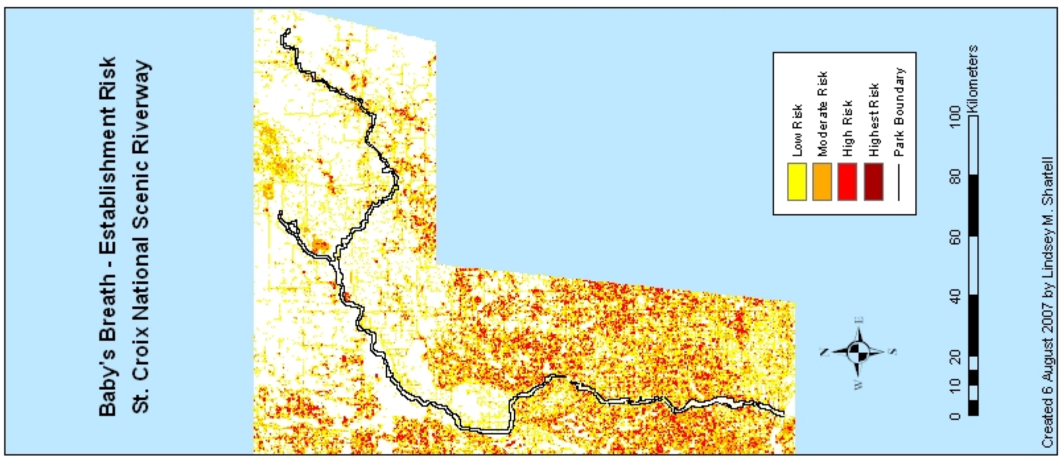


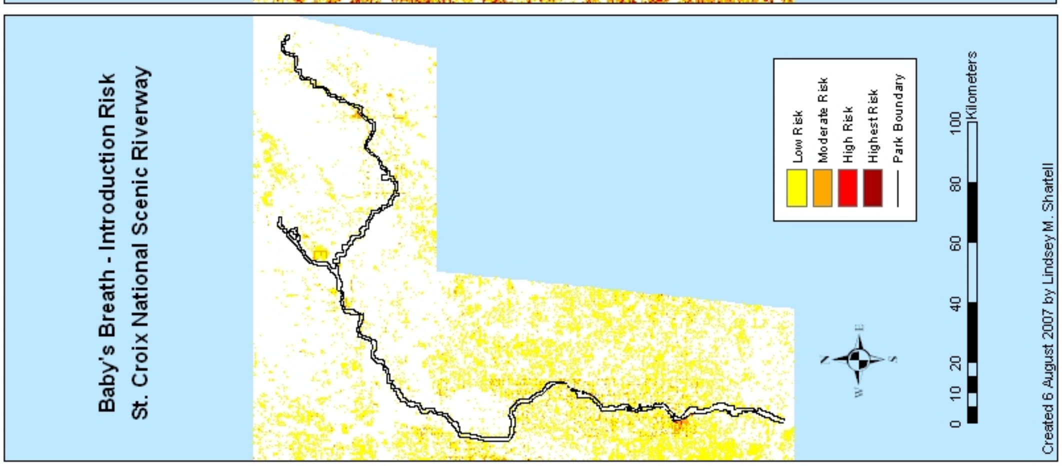




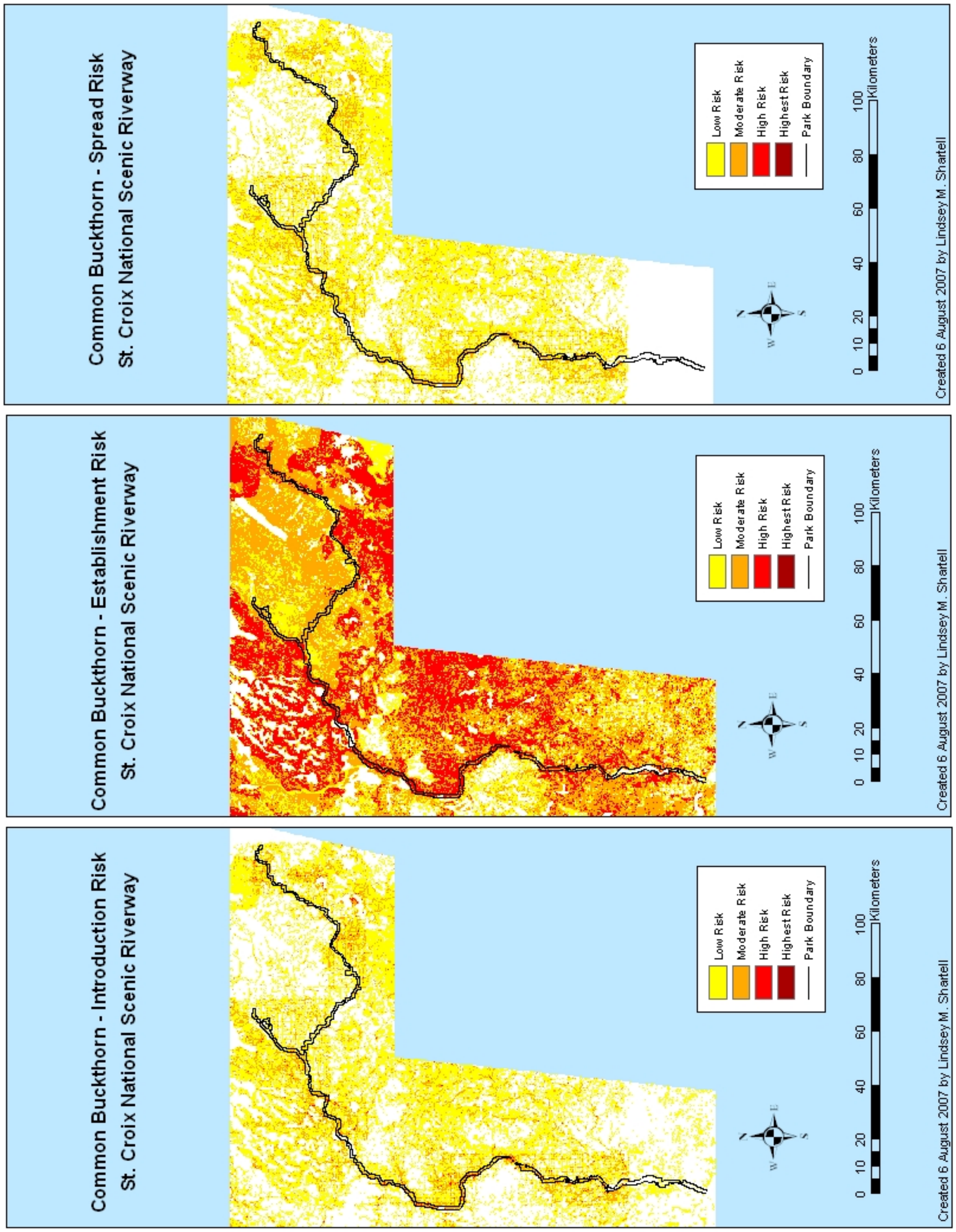
Appendix 3. Cont.



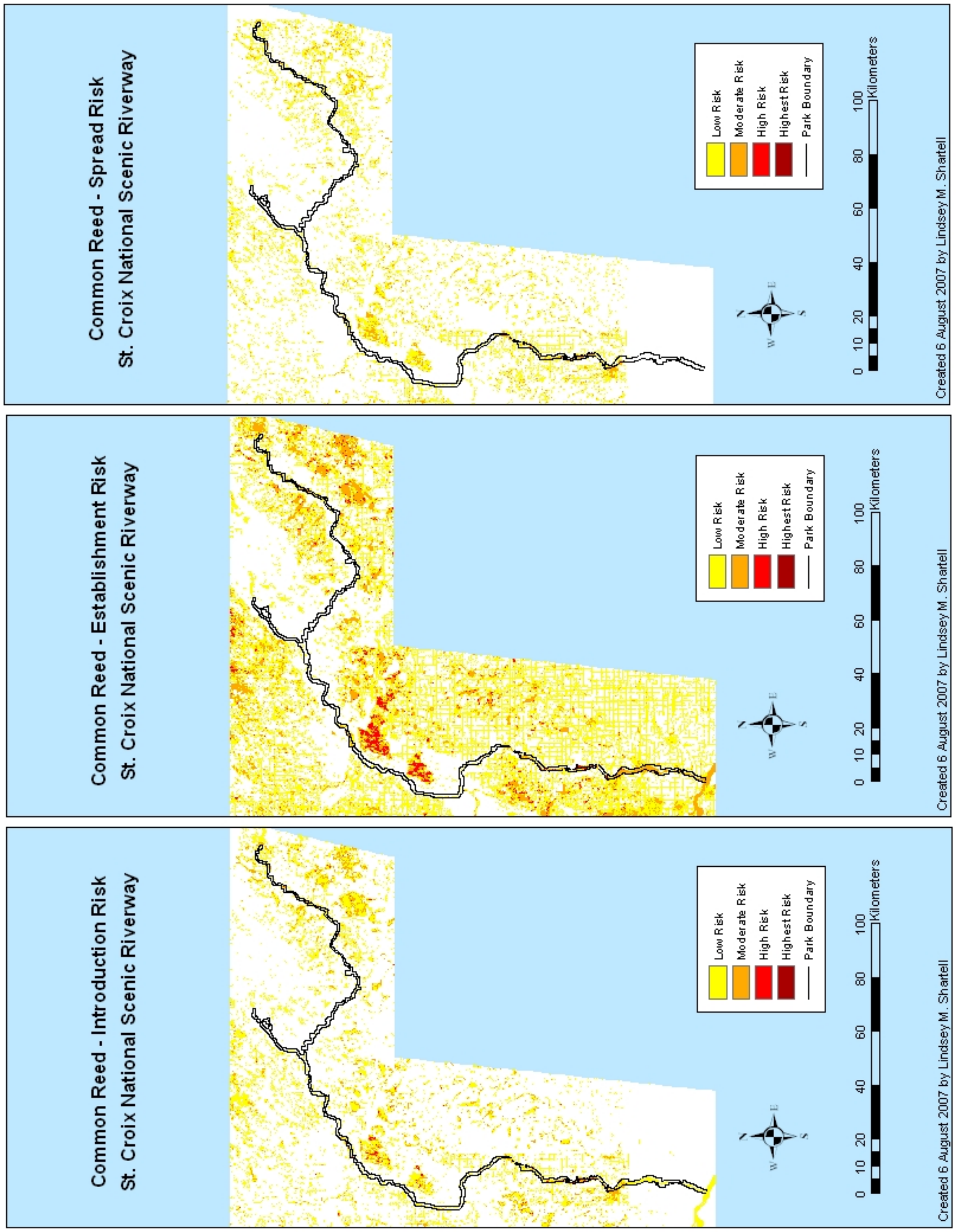




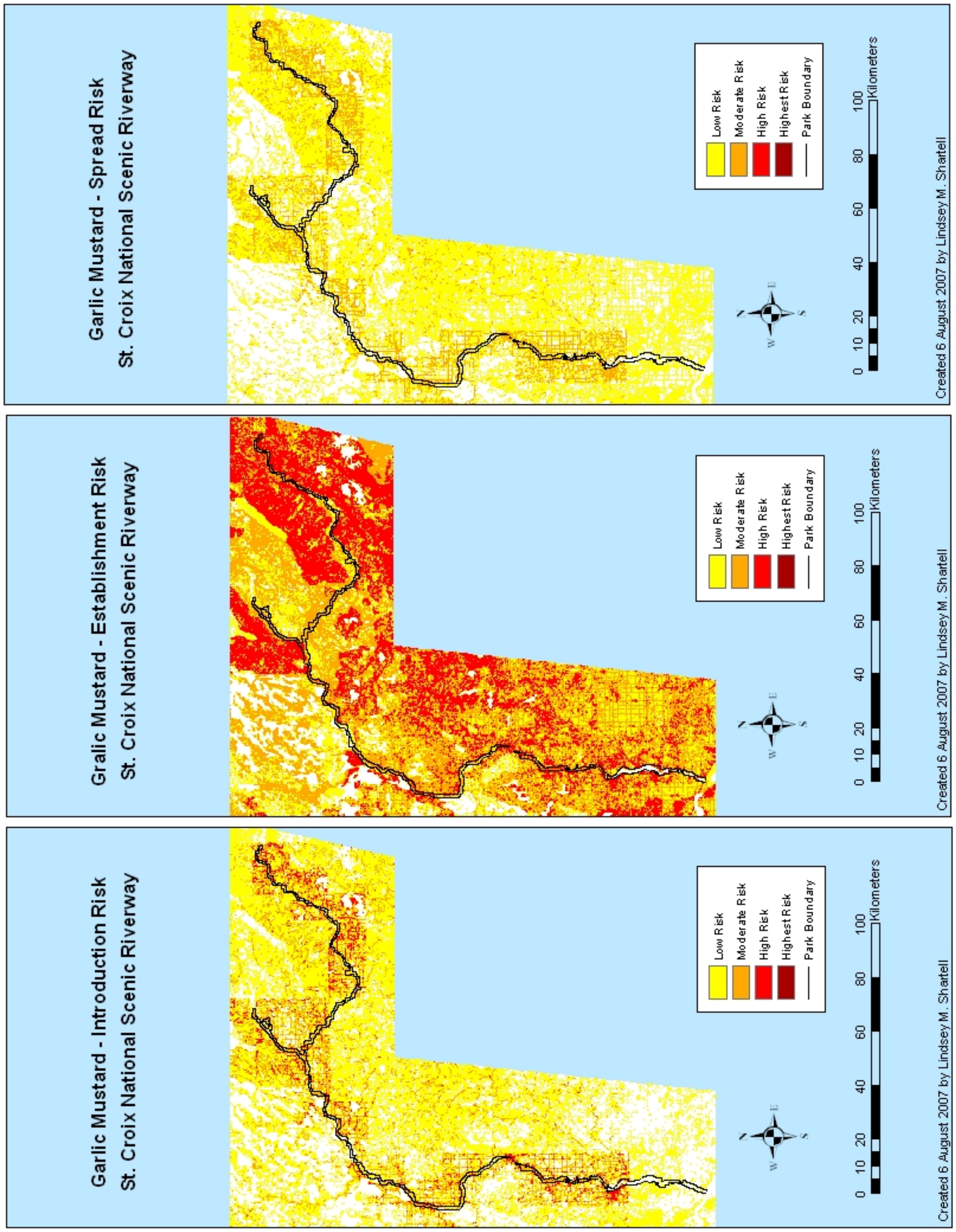
Appendix 3. Cont.



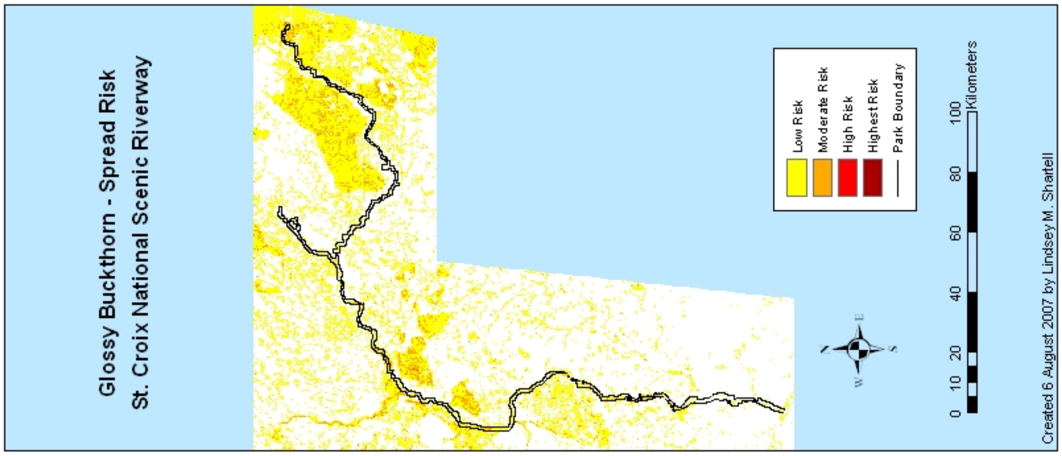
Appendix 3. Cont.

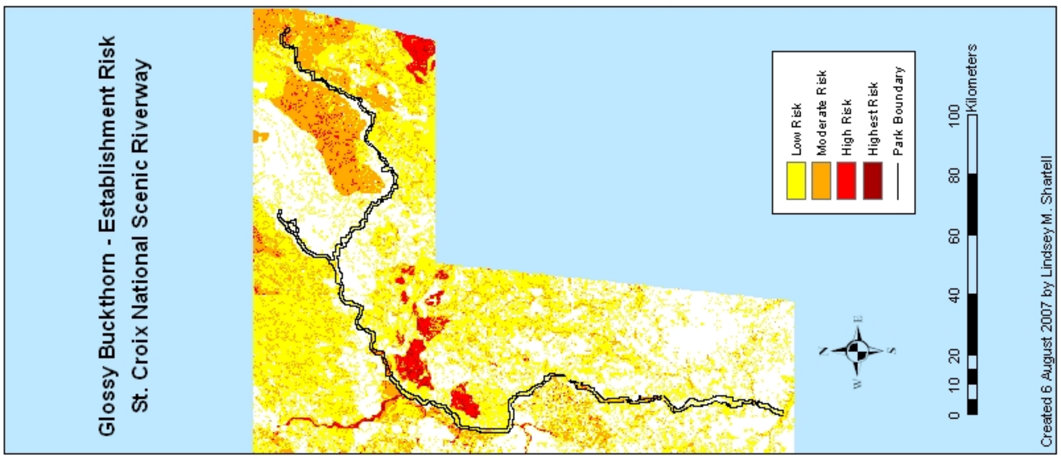


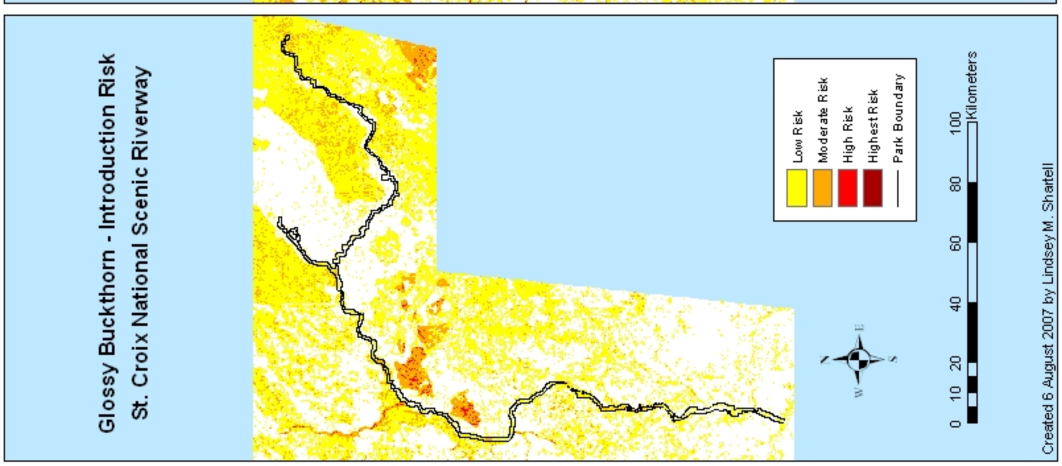
Appendix 3. Cont.



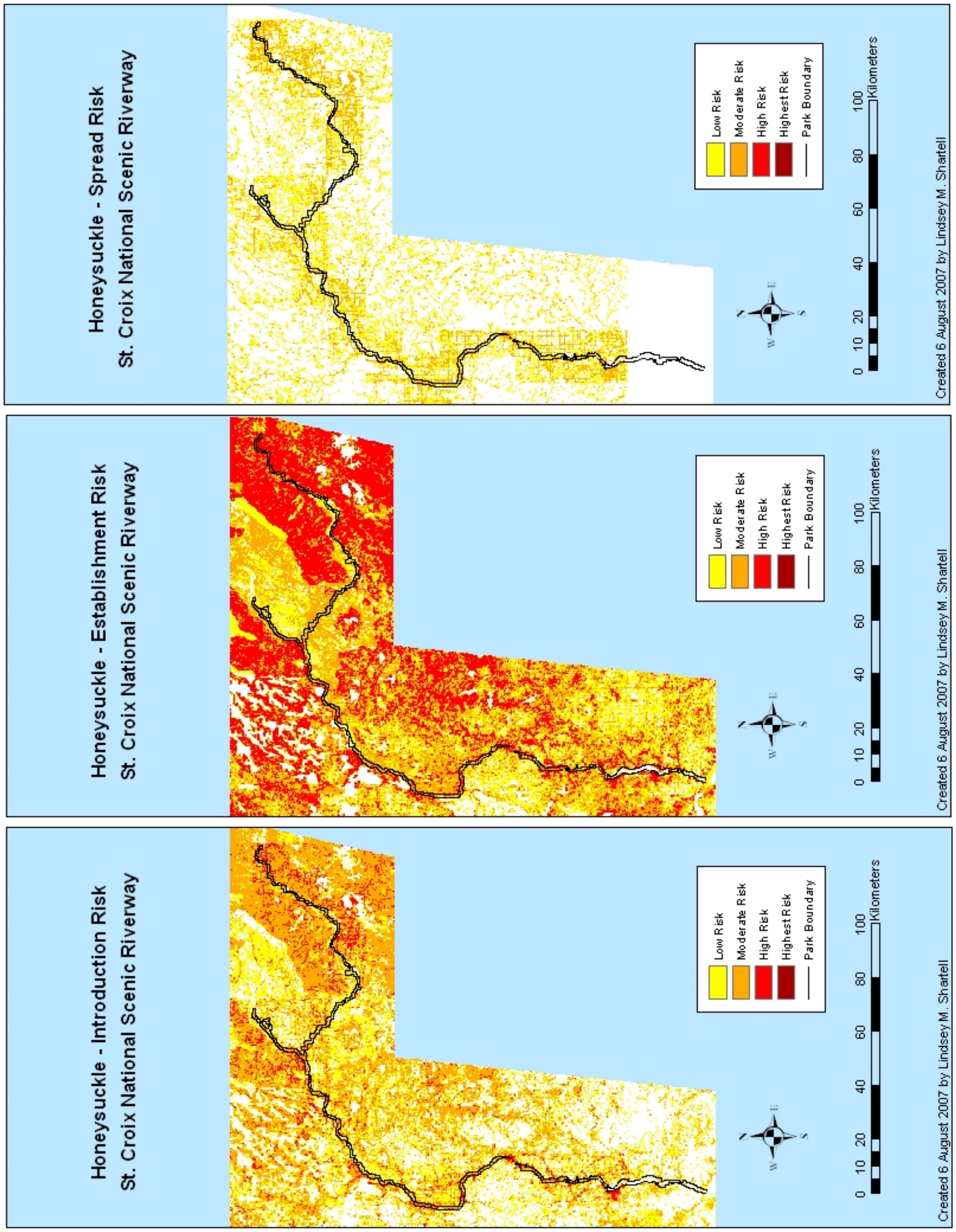
Appendix 3. Cont.



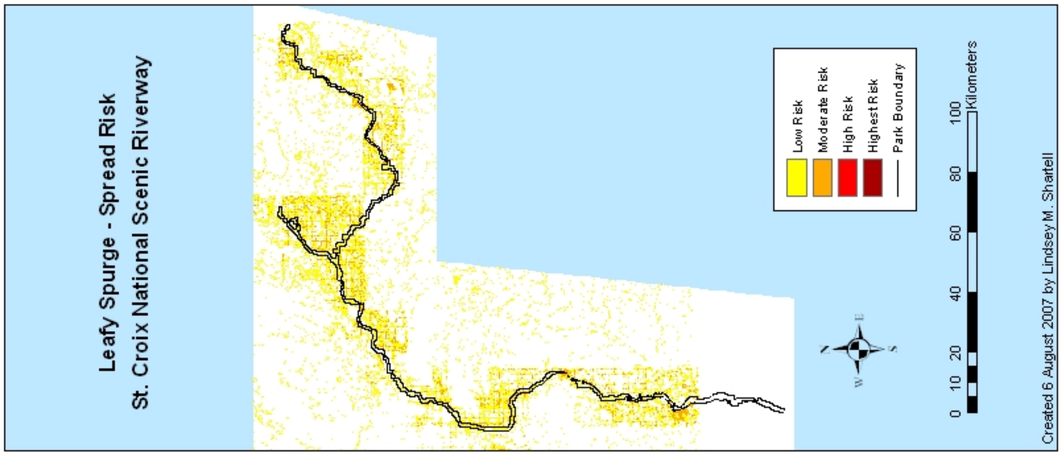


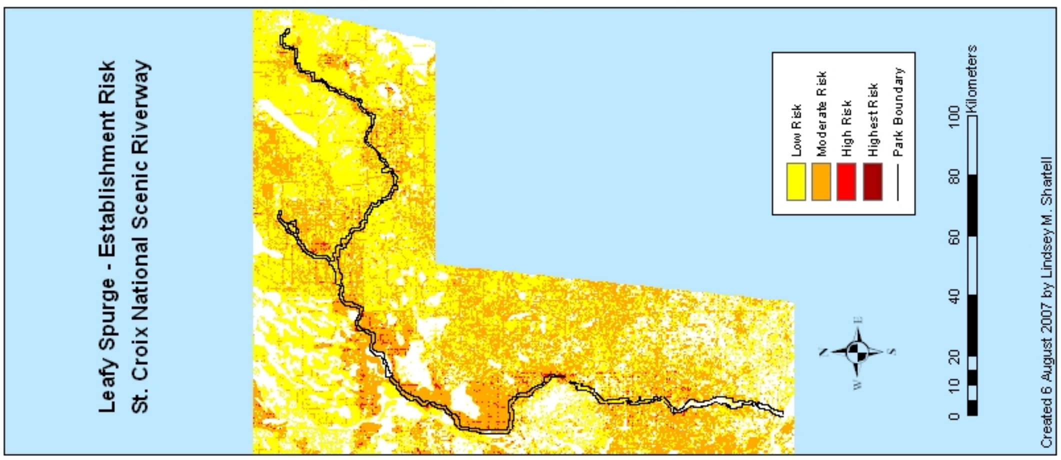


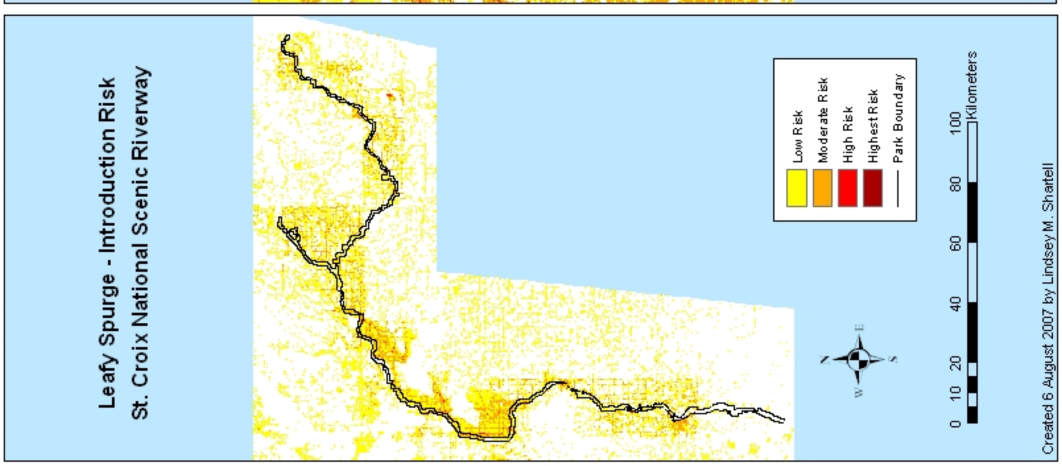
Appendix 3. Cont.



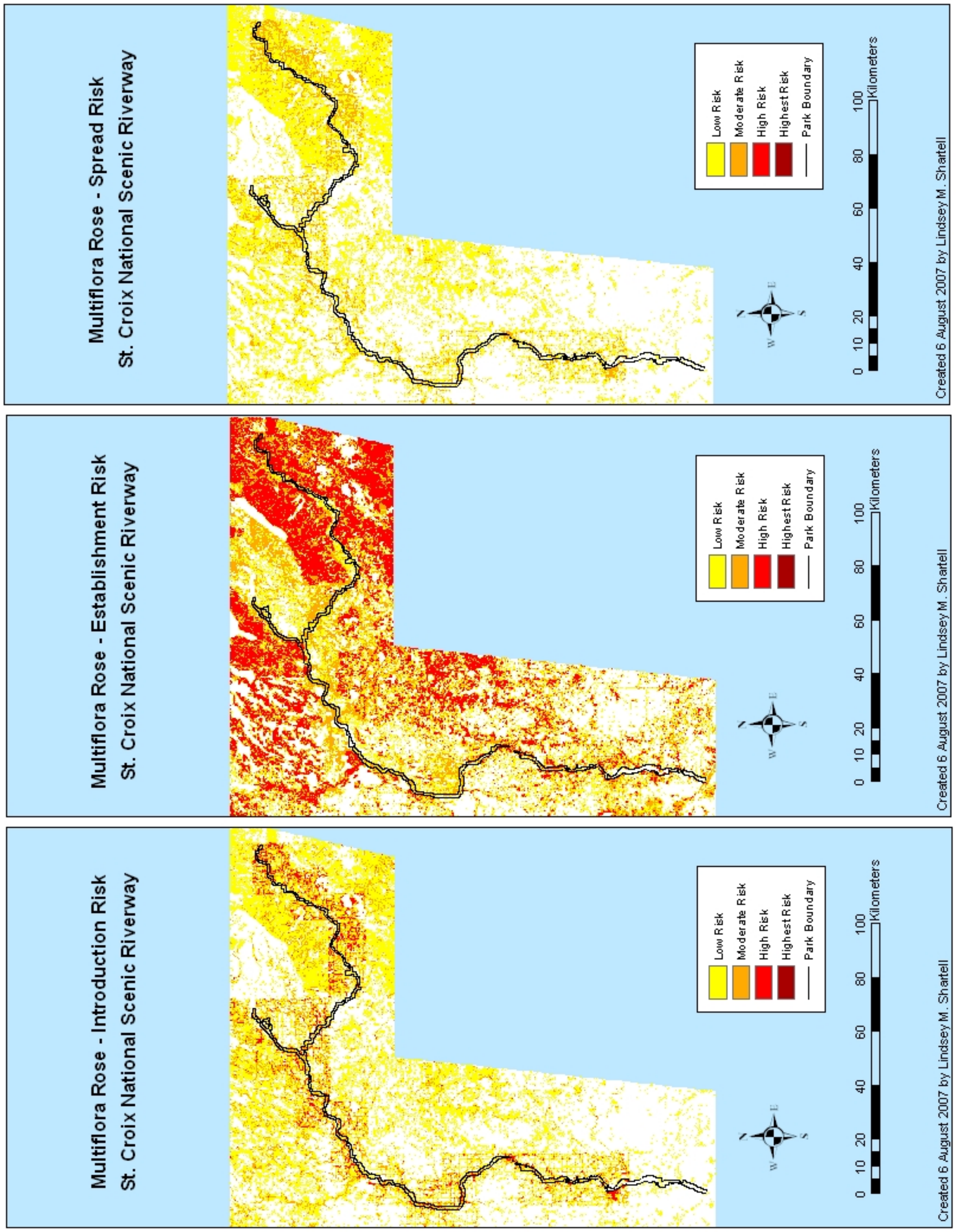
Appendix 3. Cont.



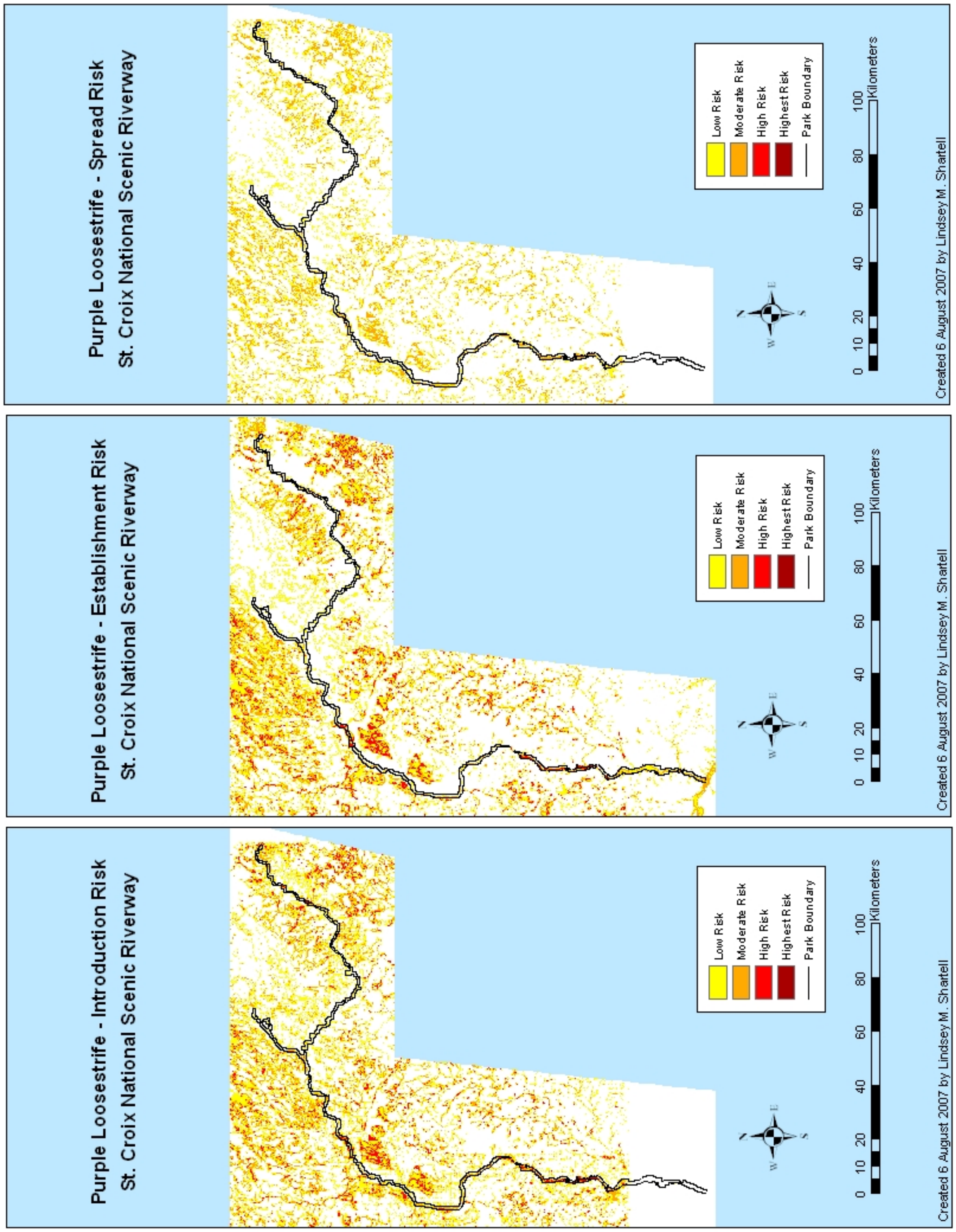




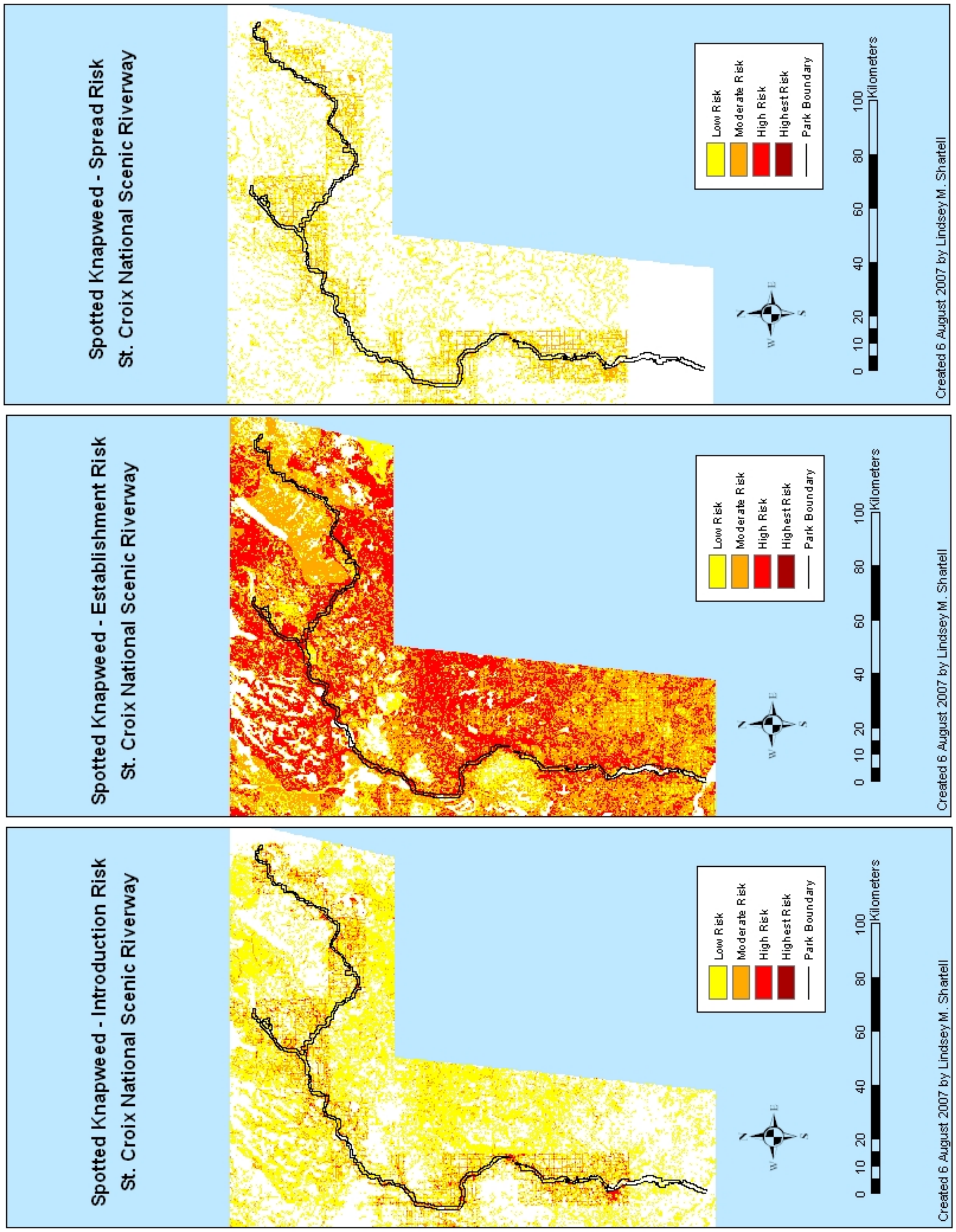
Appendix 3. Cont.



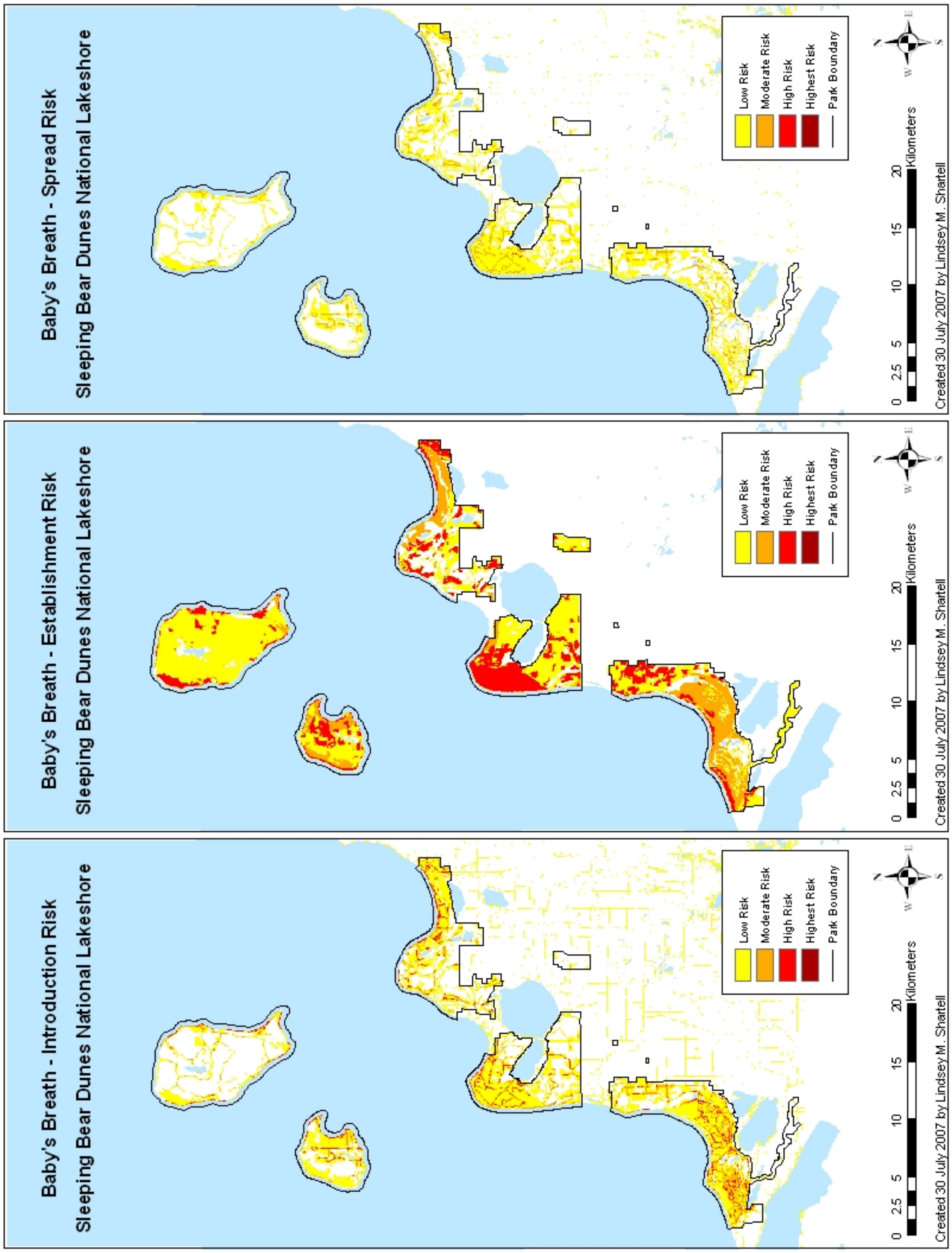
Appendix 3. Cont.



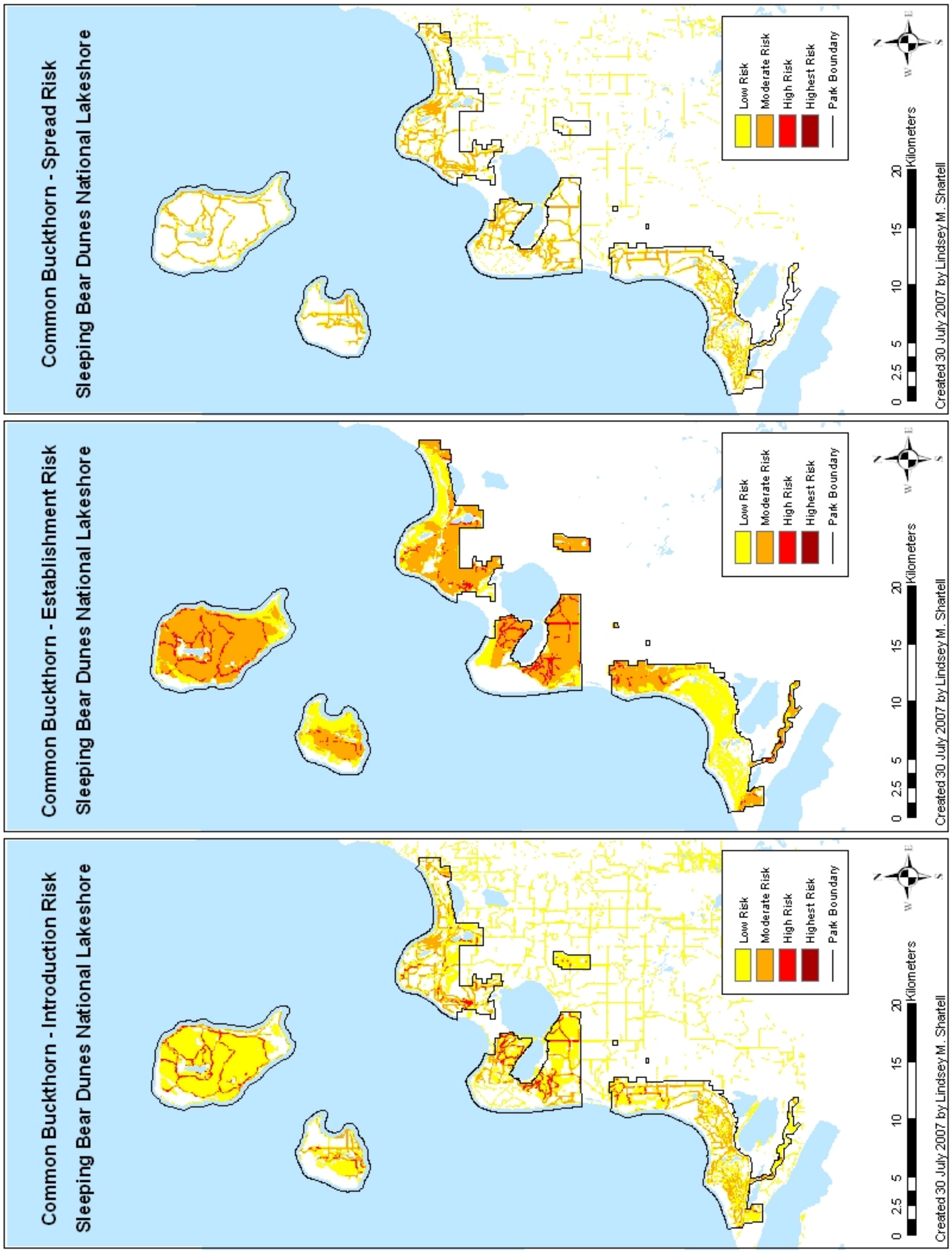
Appendix 3. Cont.



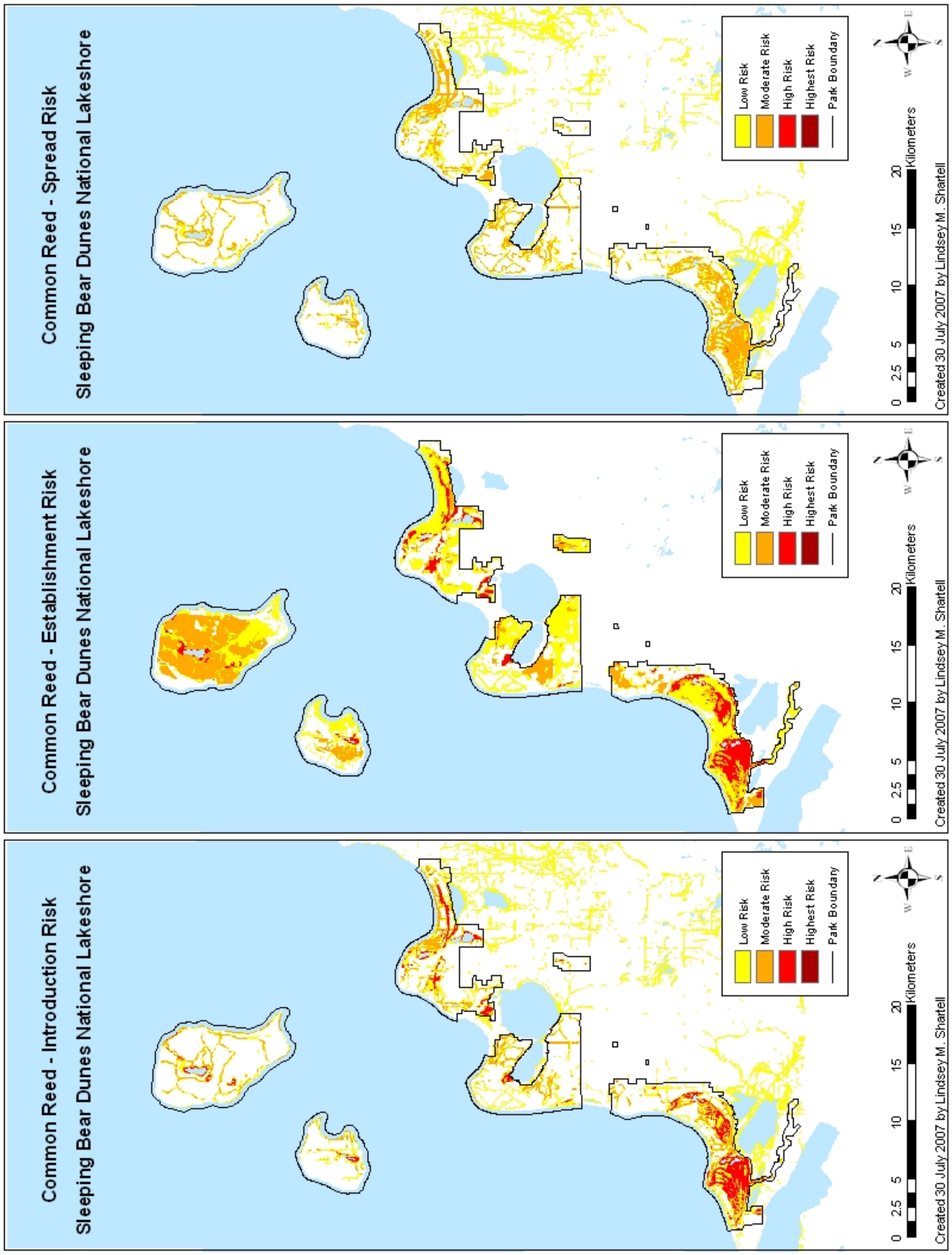
Appendix 3. Cont.



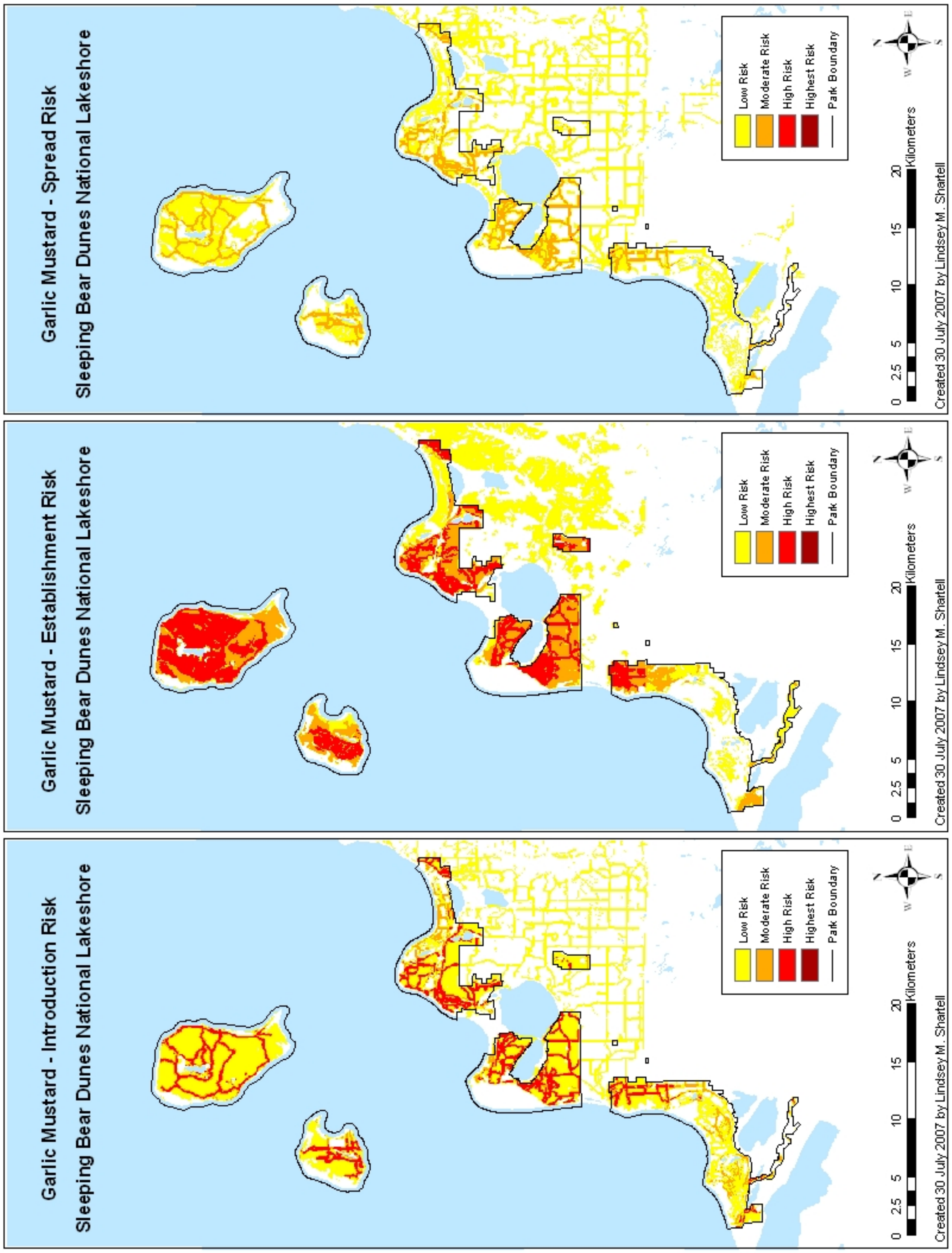
Appendix 3. Cont.



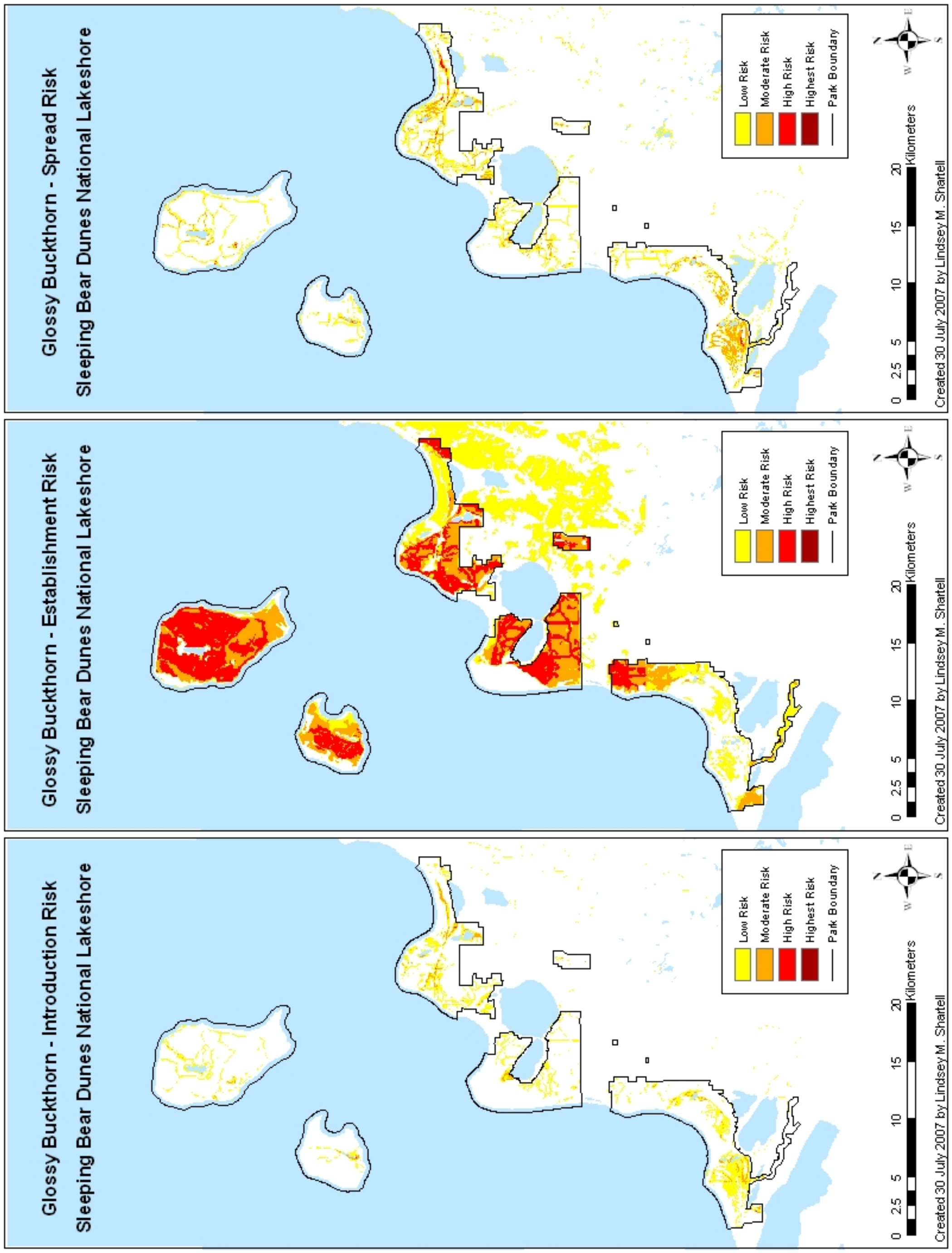
Appendix 3. Cont.



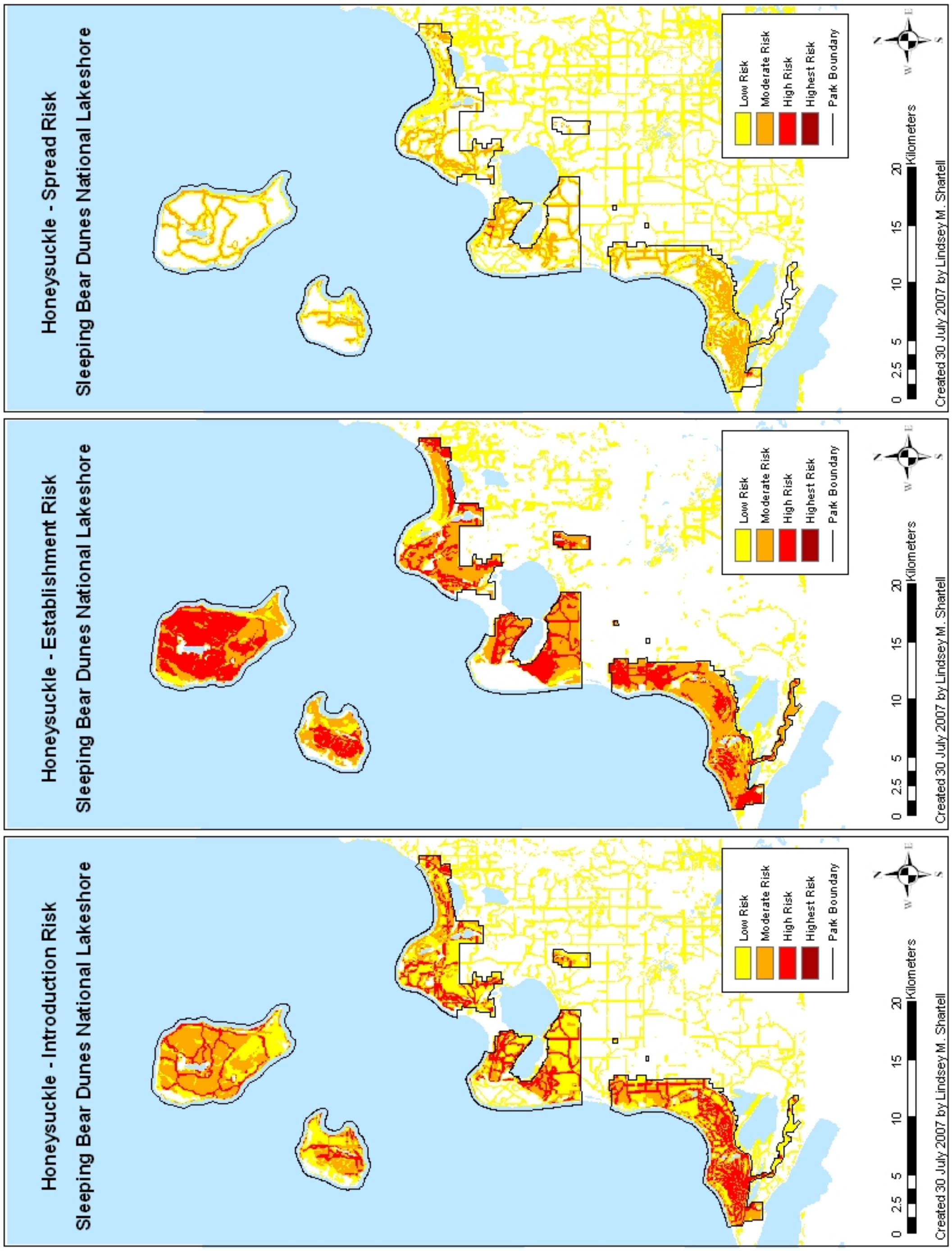
Appendix 3. Cont.



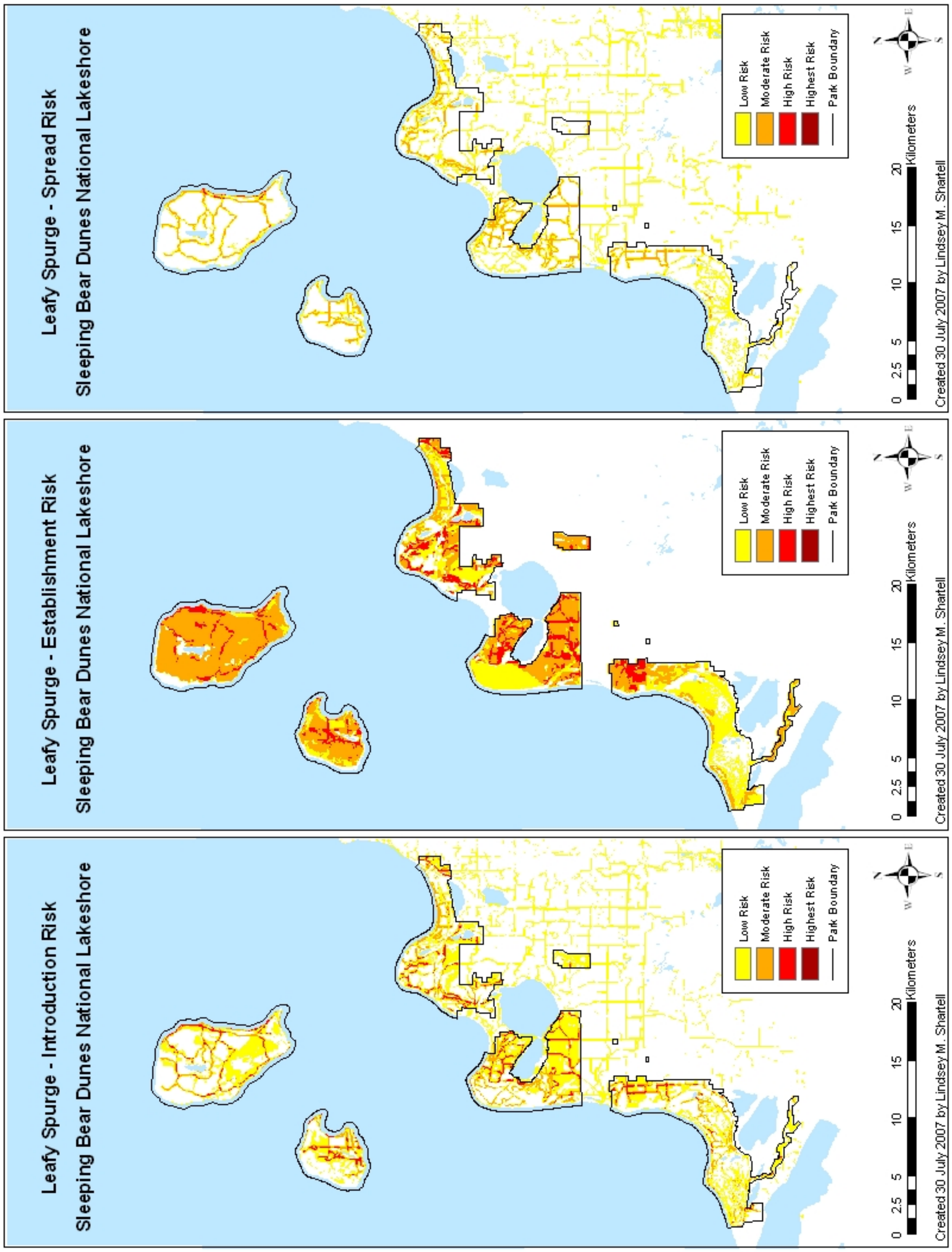
Appendix 3. Cont.



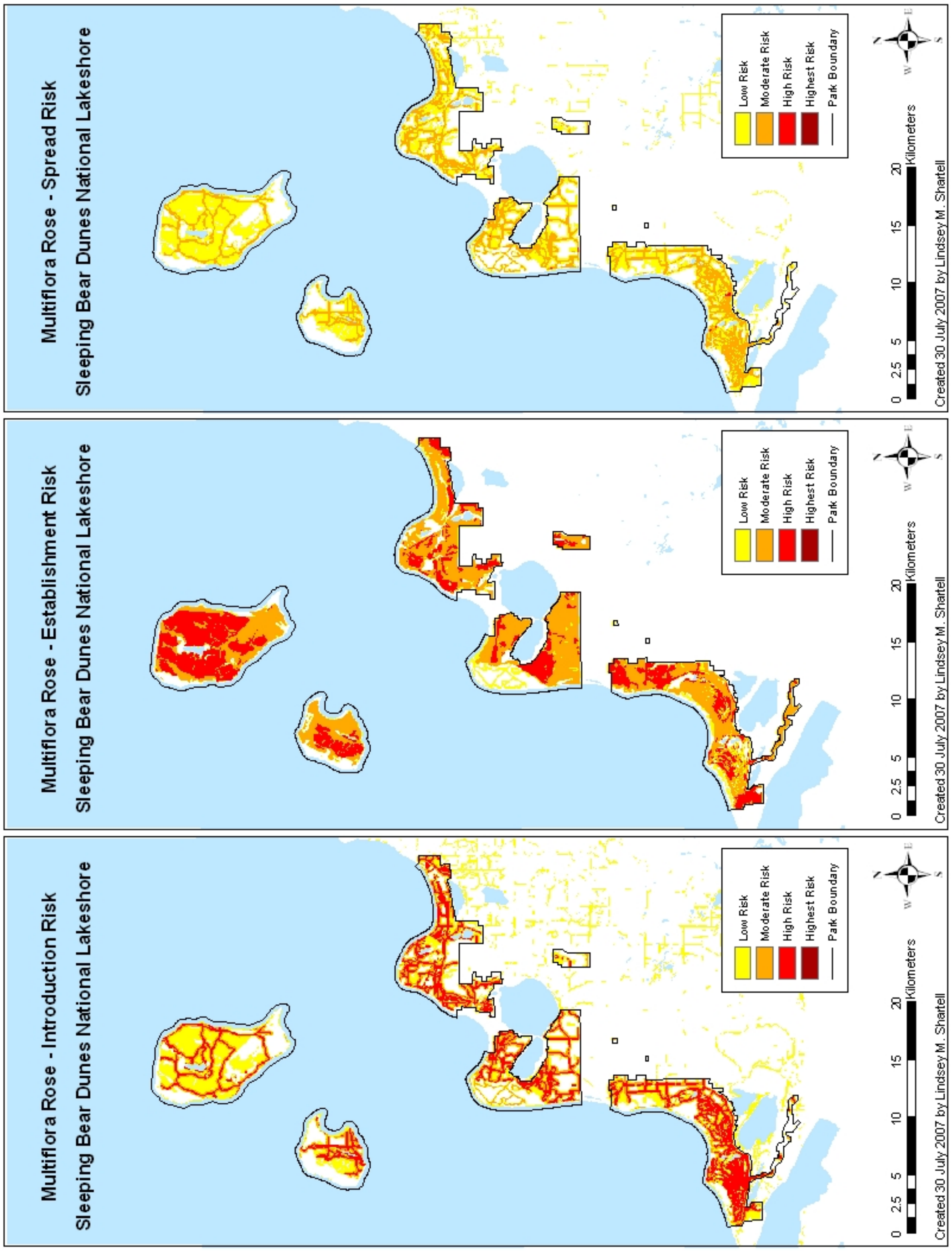
Appendix 3. Cont.



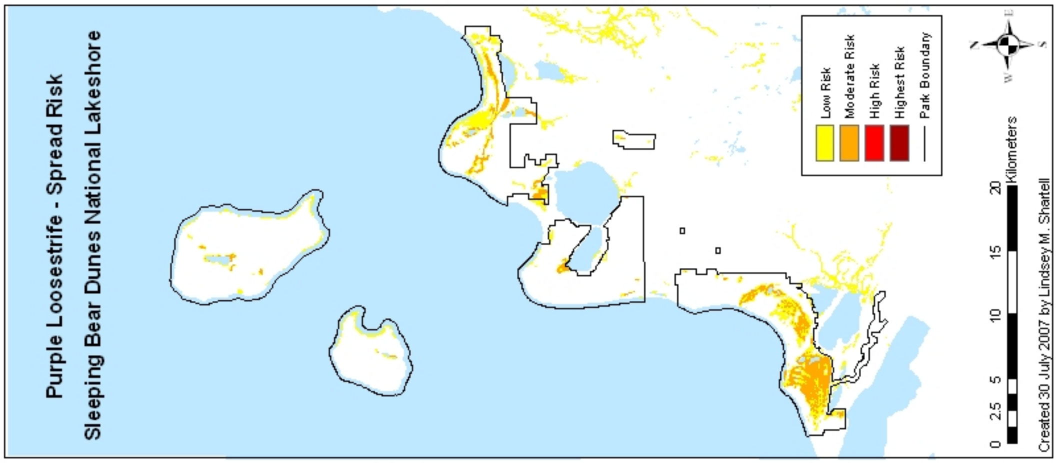
Appendix 3. Cont.

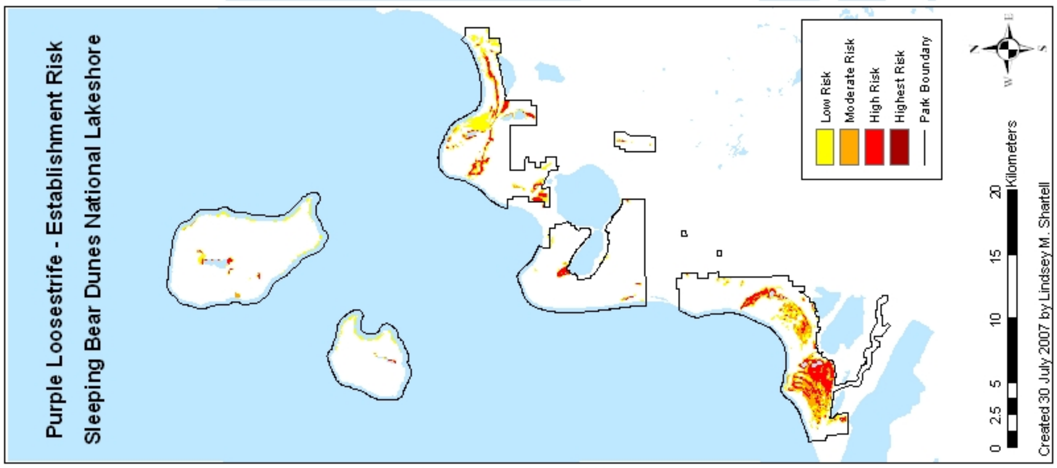


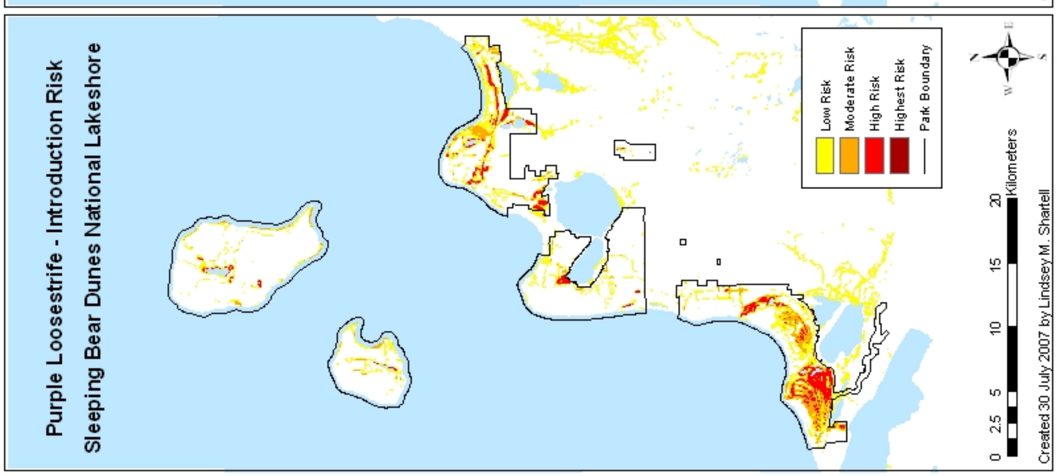
Appendix 3. Cont.



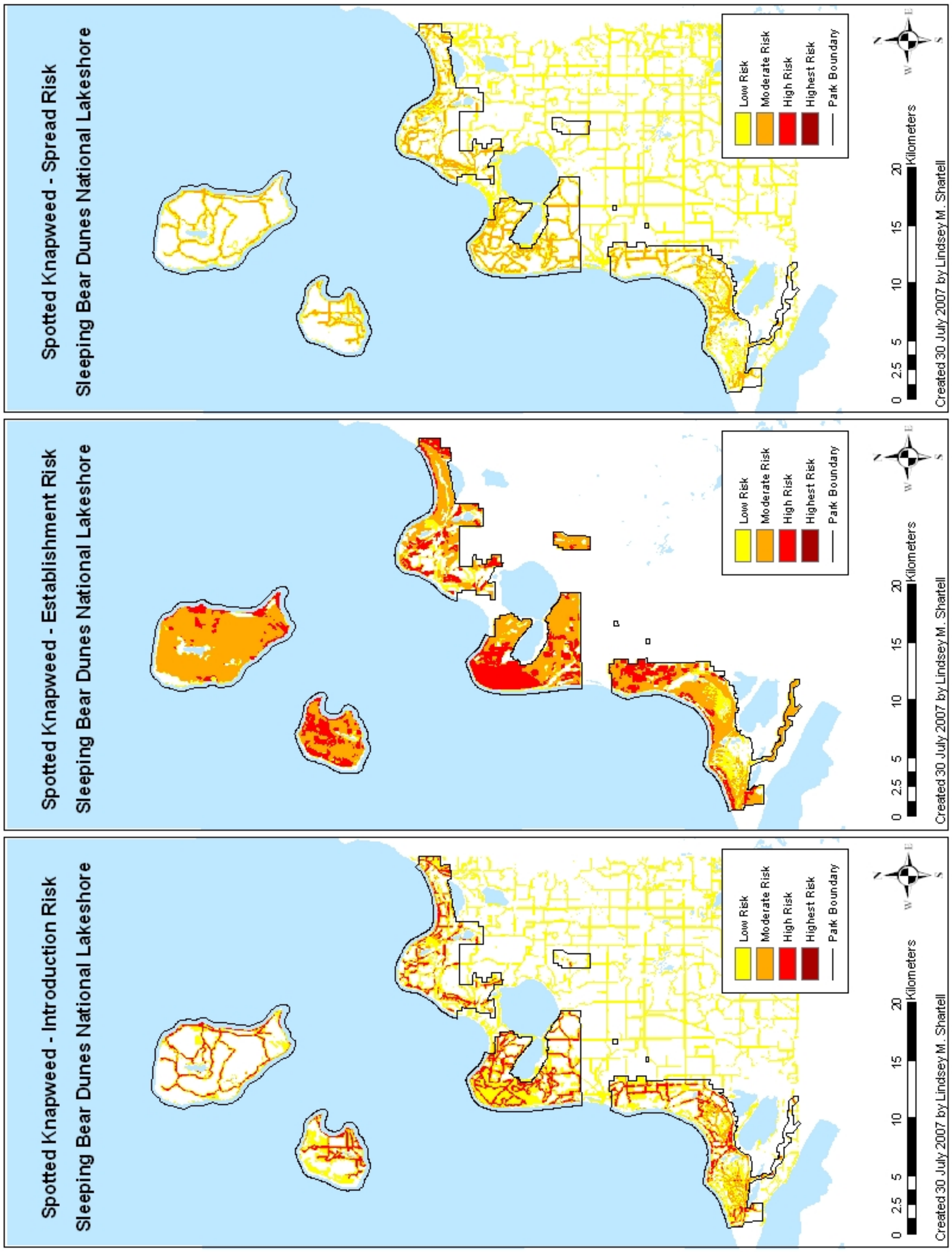
Appendix 3. Cont.



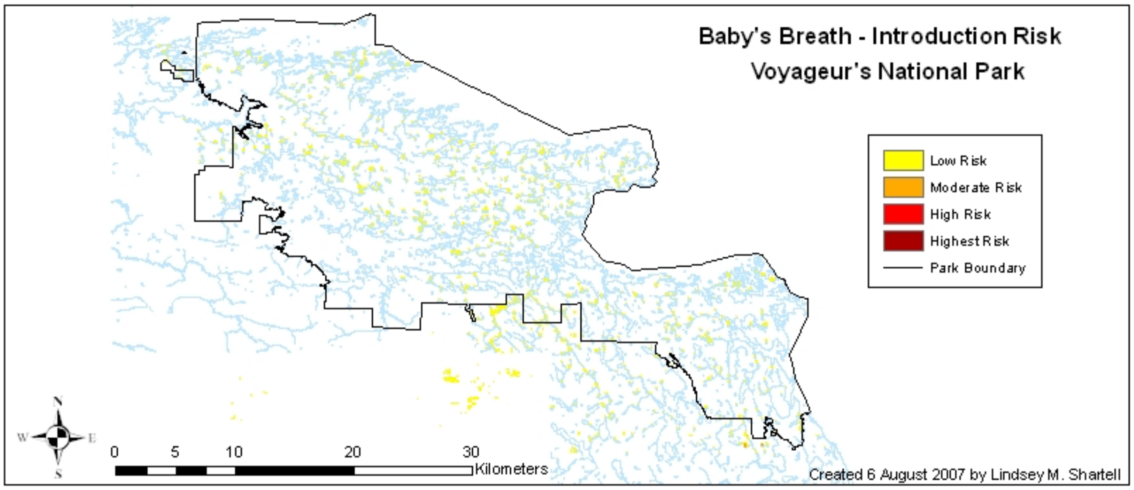


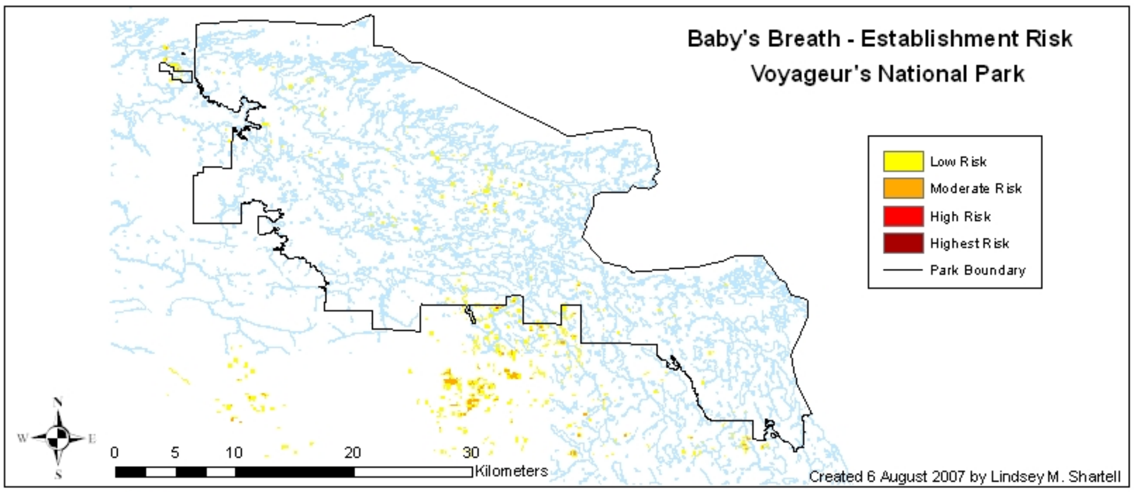


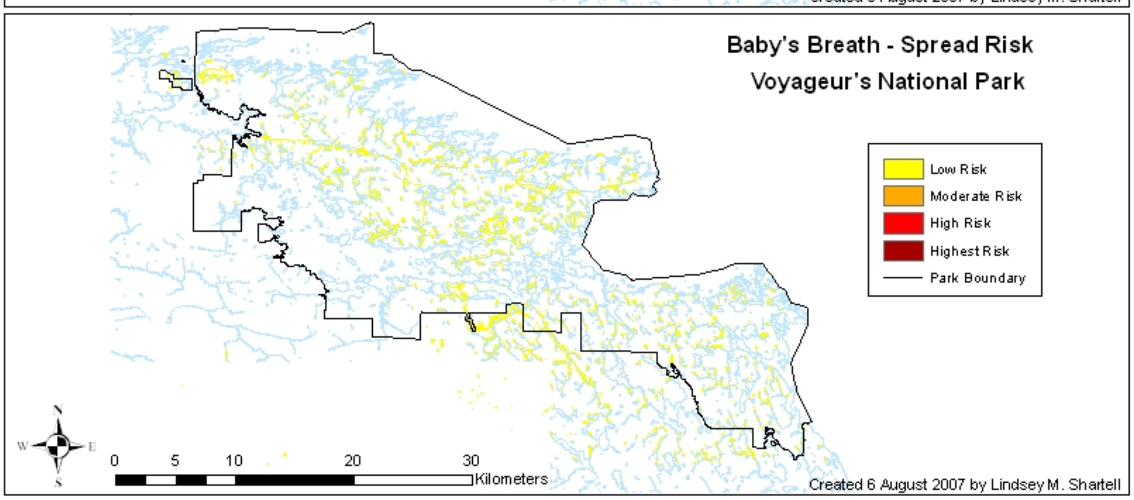
Appendix 3. Cont.



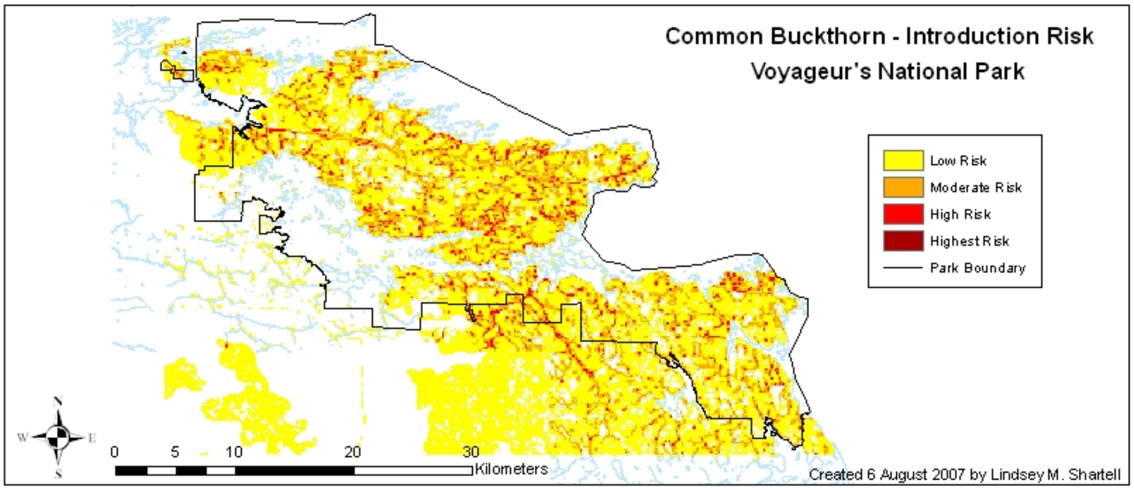
Appendix 3. Cont.

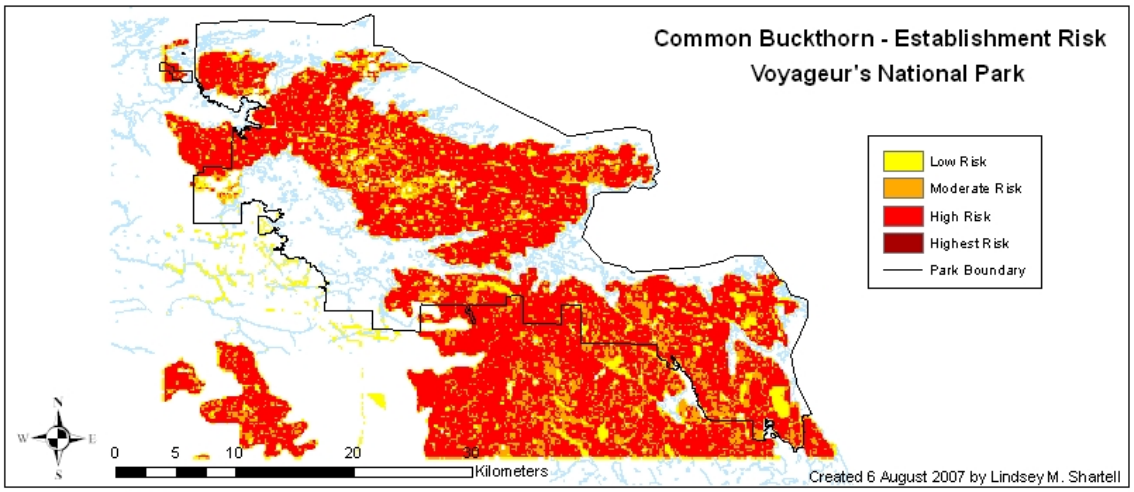


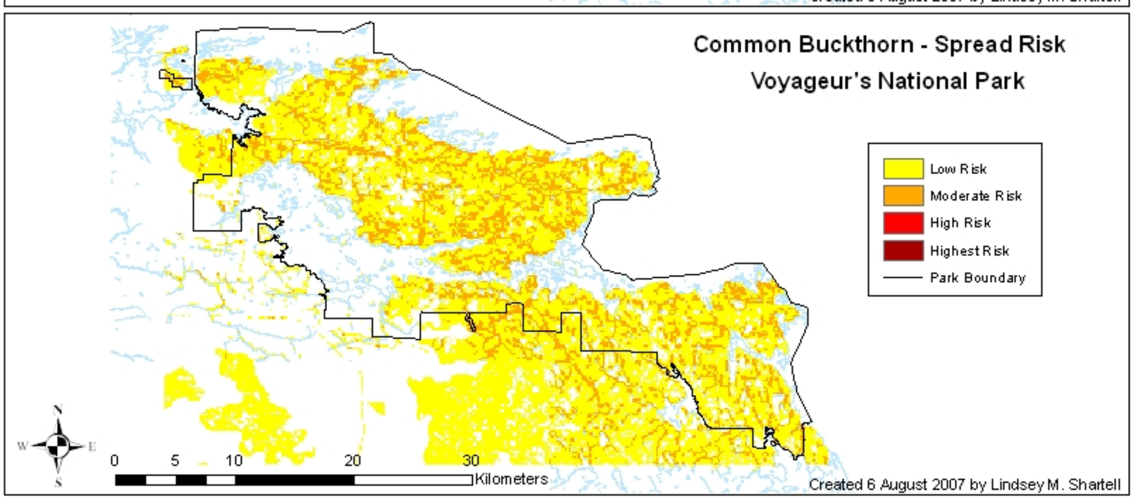




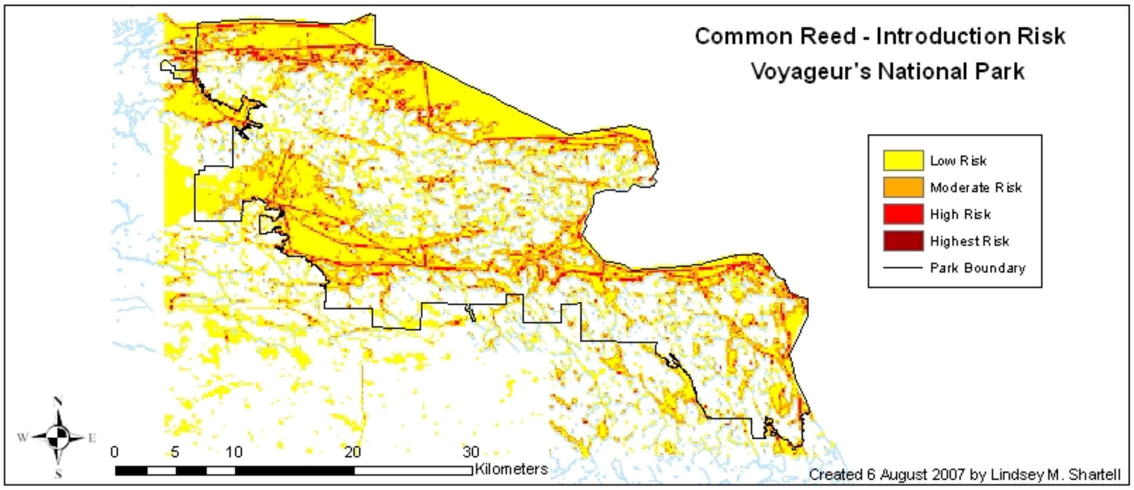
Appendix 3. Cont.

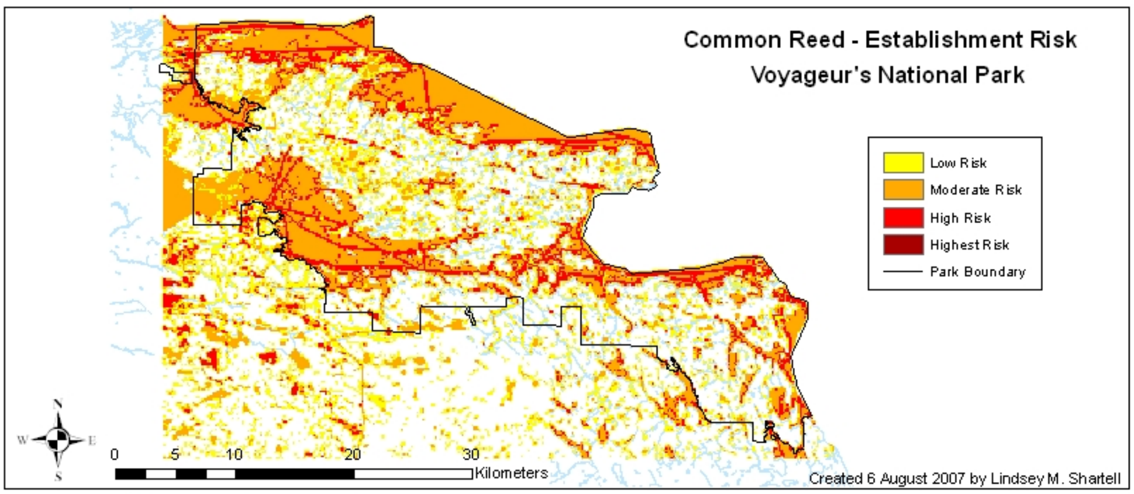


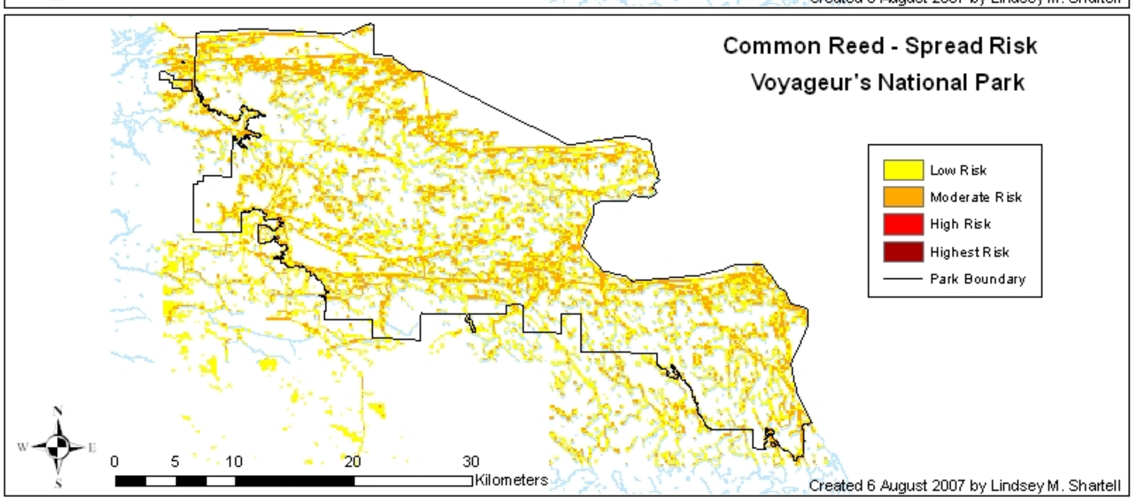




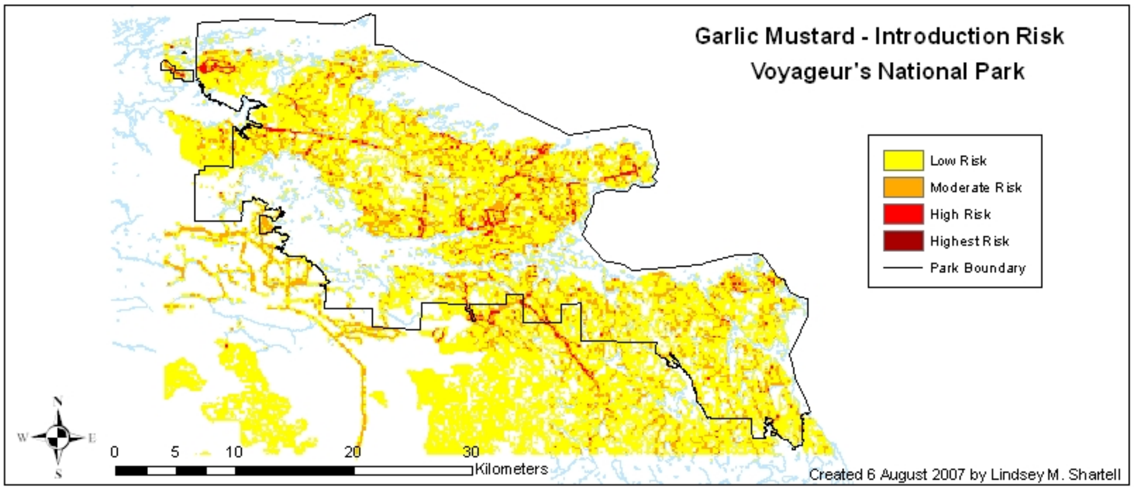
Appendix 3. Cont.

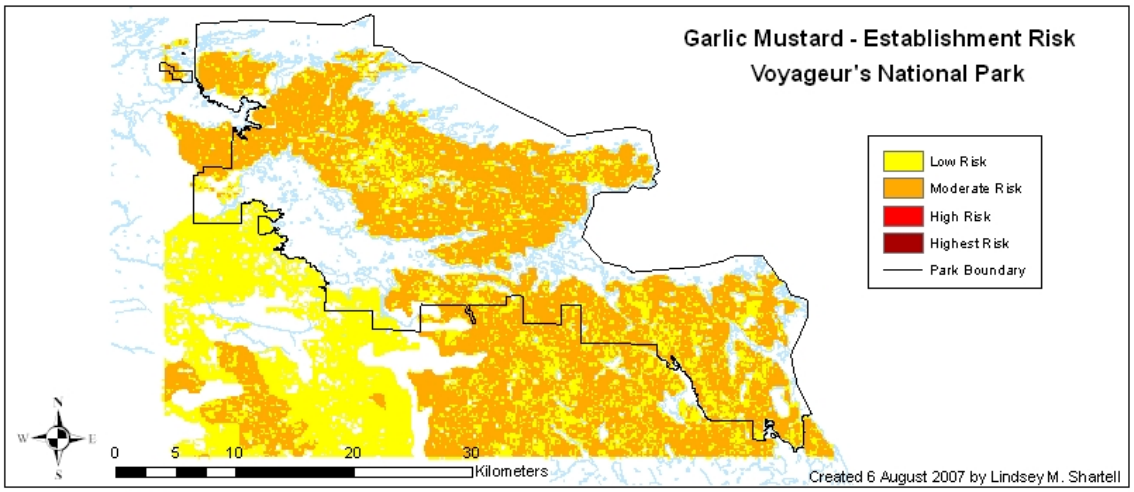


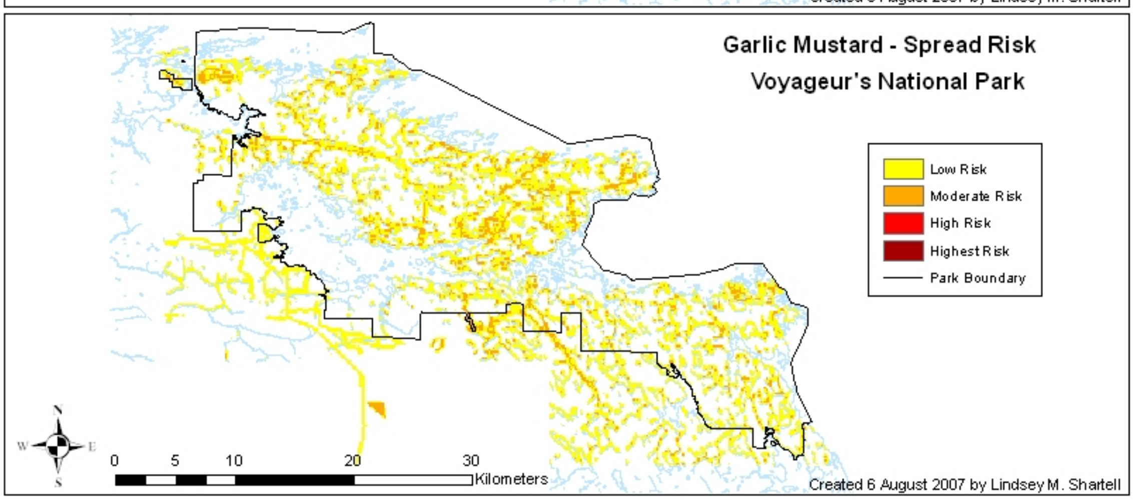




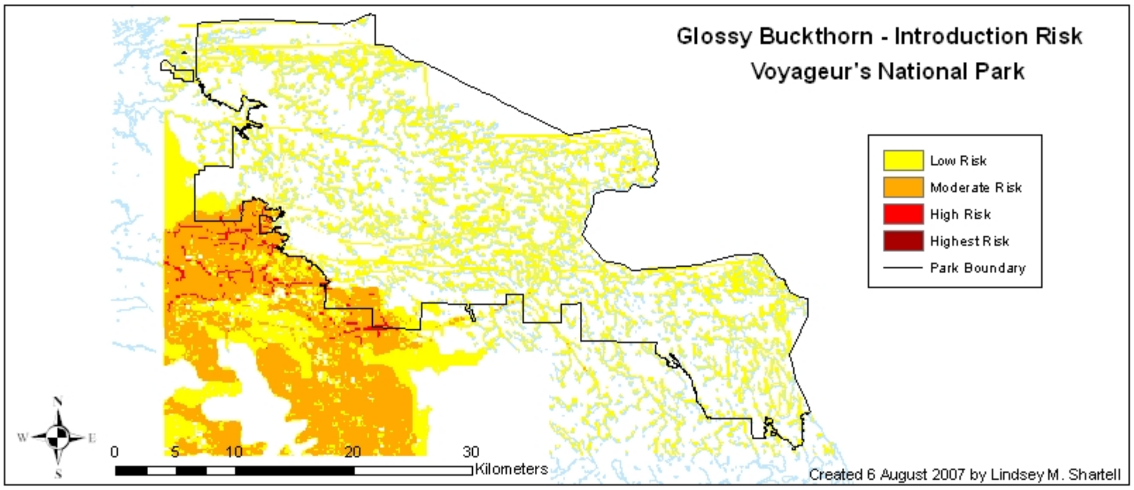
Appendix 3. Cont.

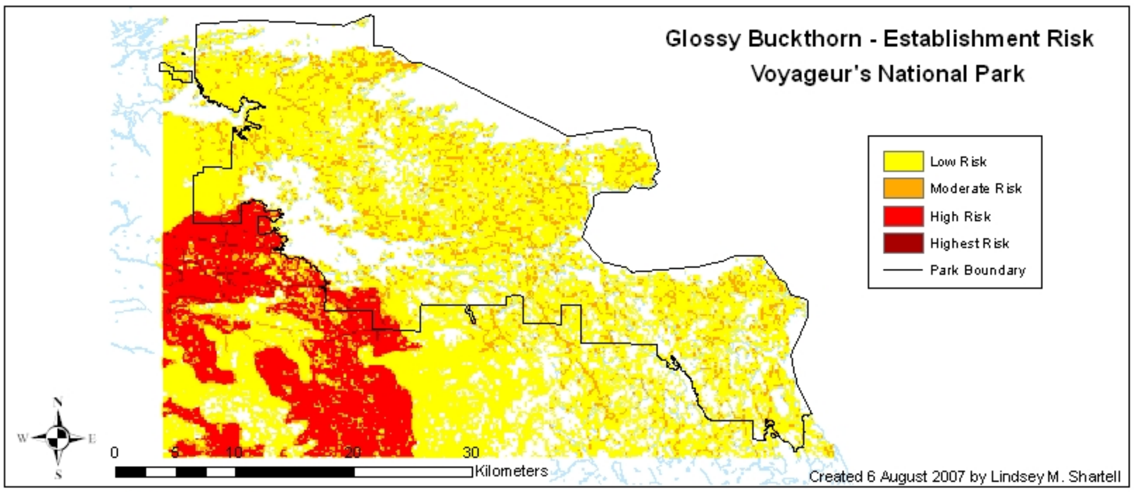


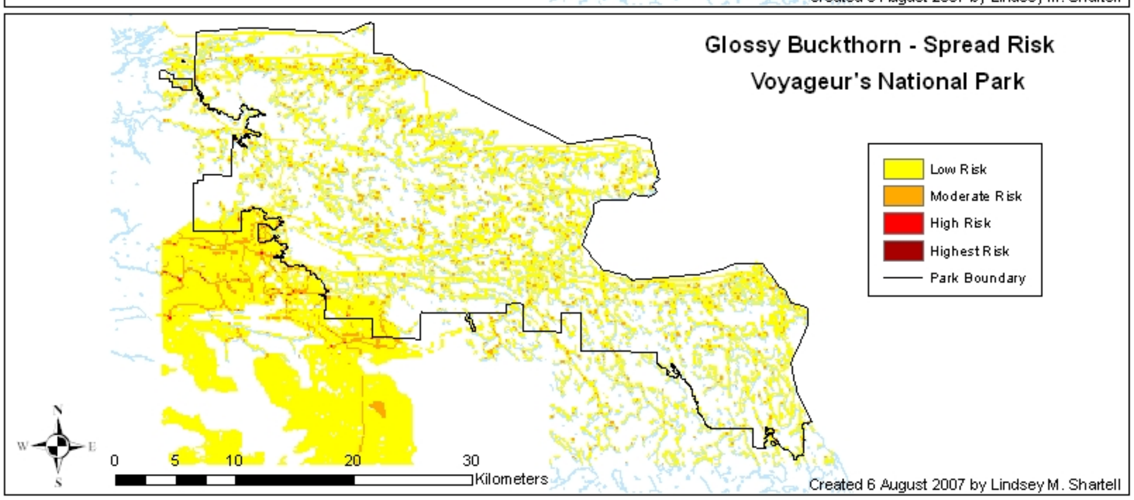




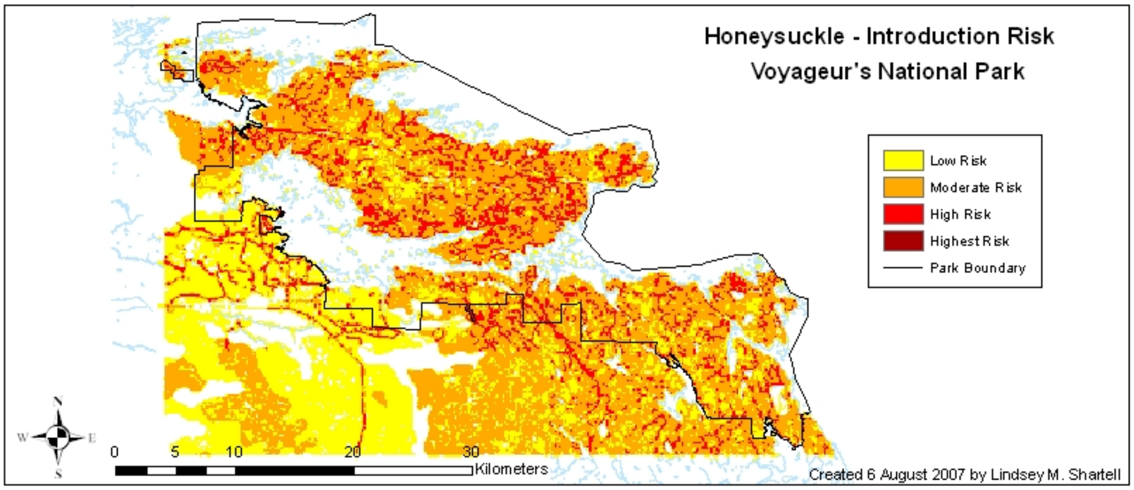
Appendix 3. Cont.

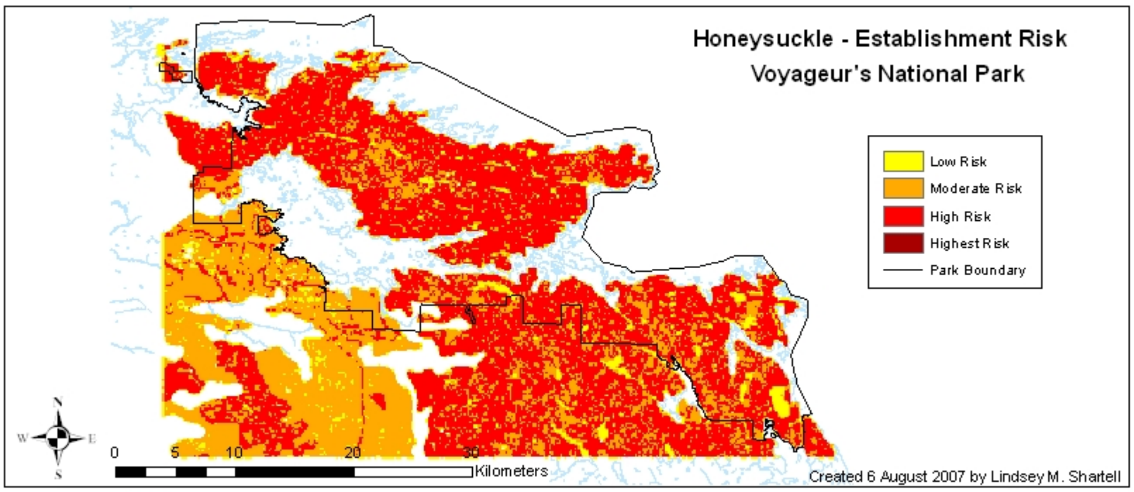


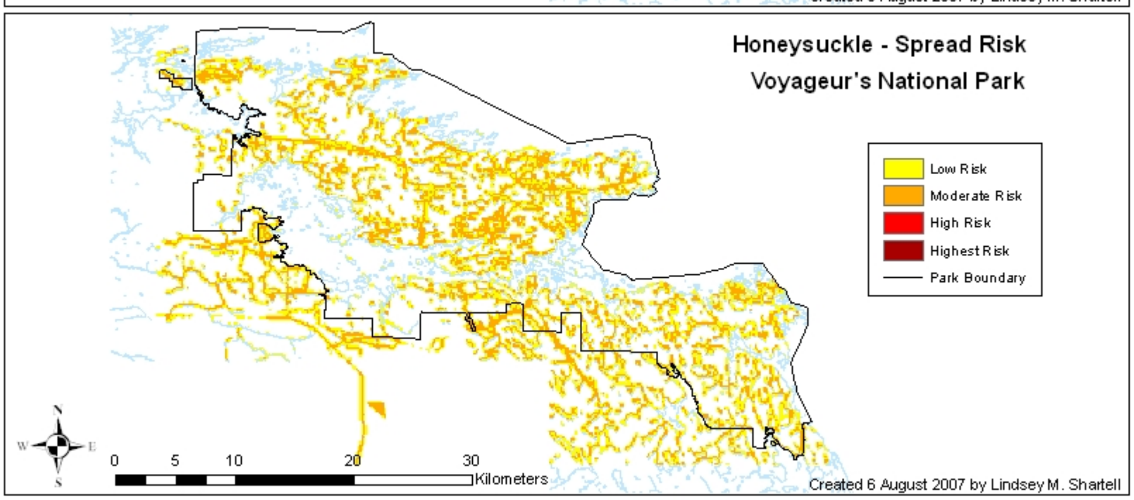




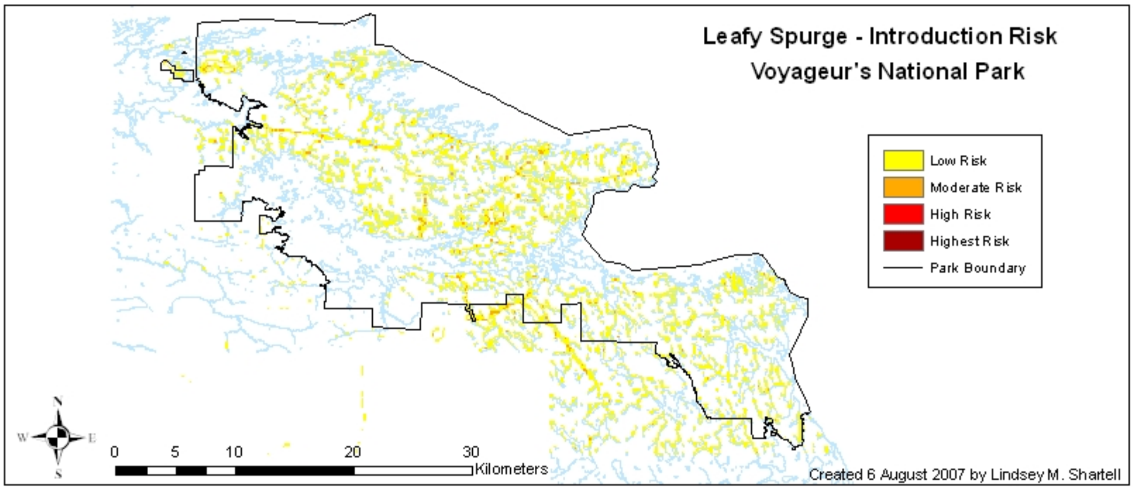
Appendix 3. Cont.

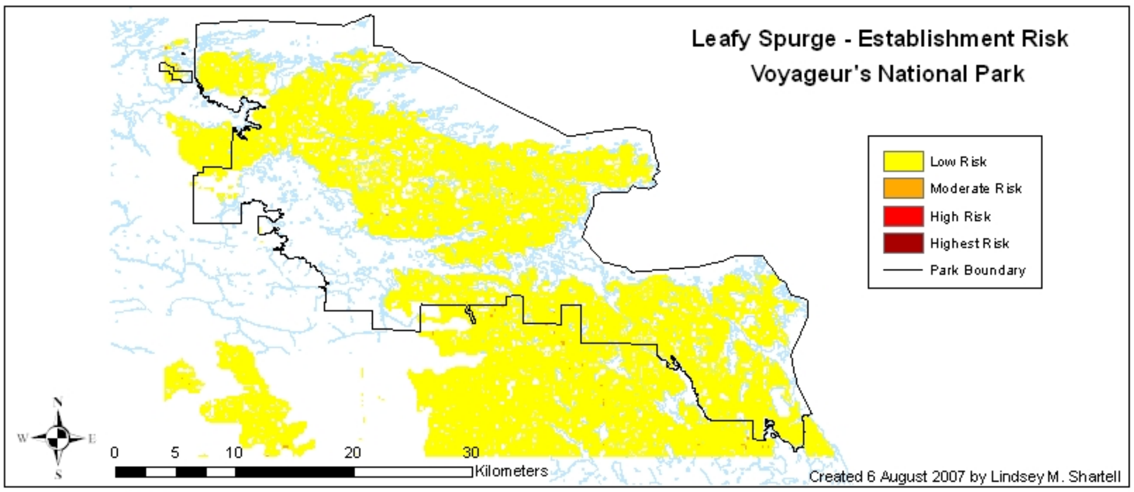


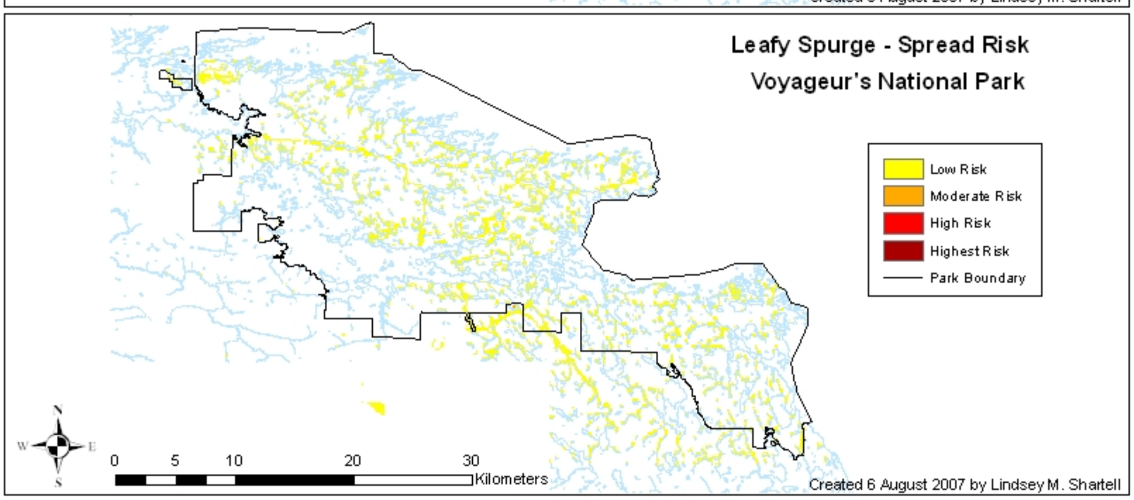




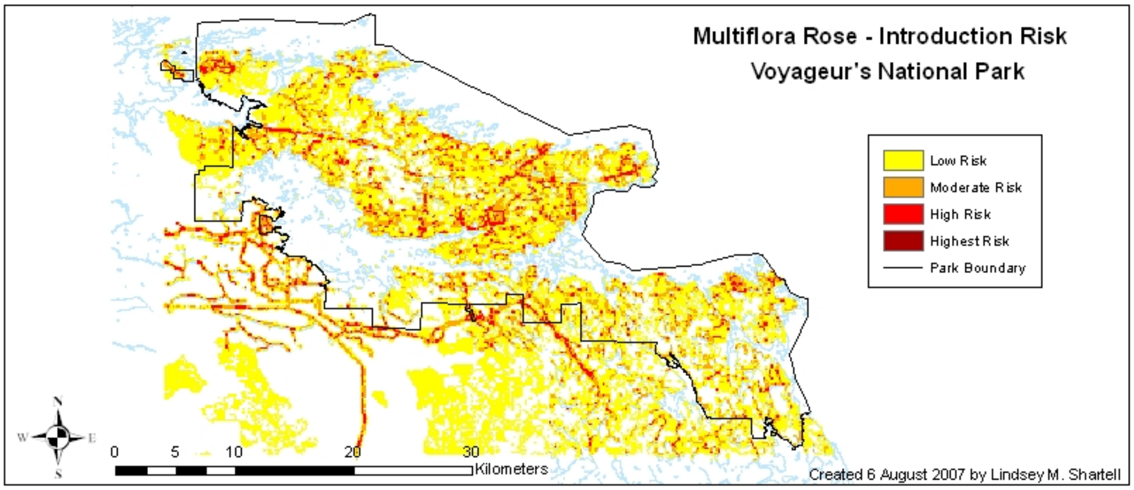
Appendix 3. Cont.

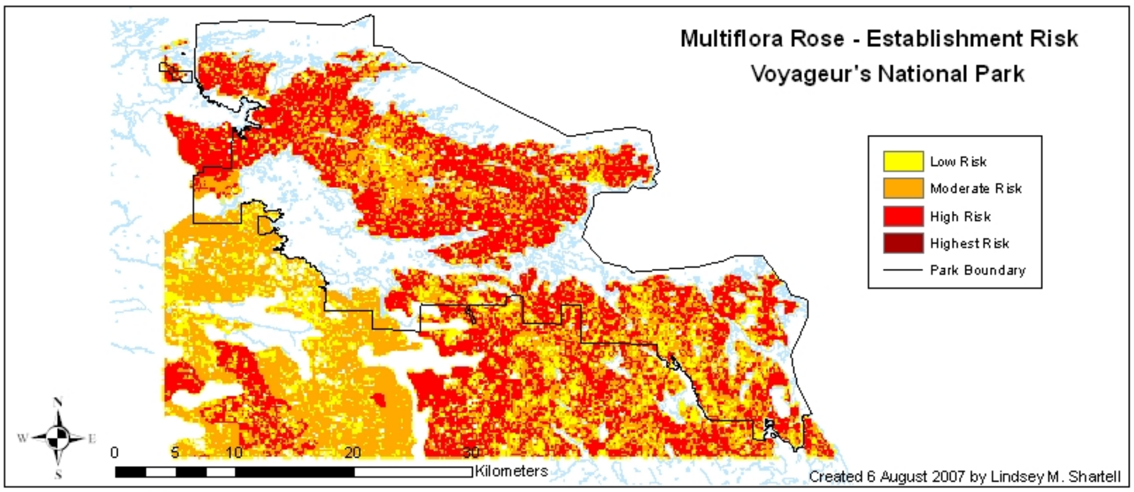


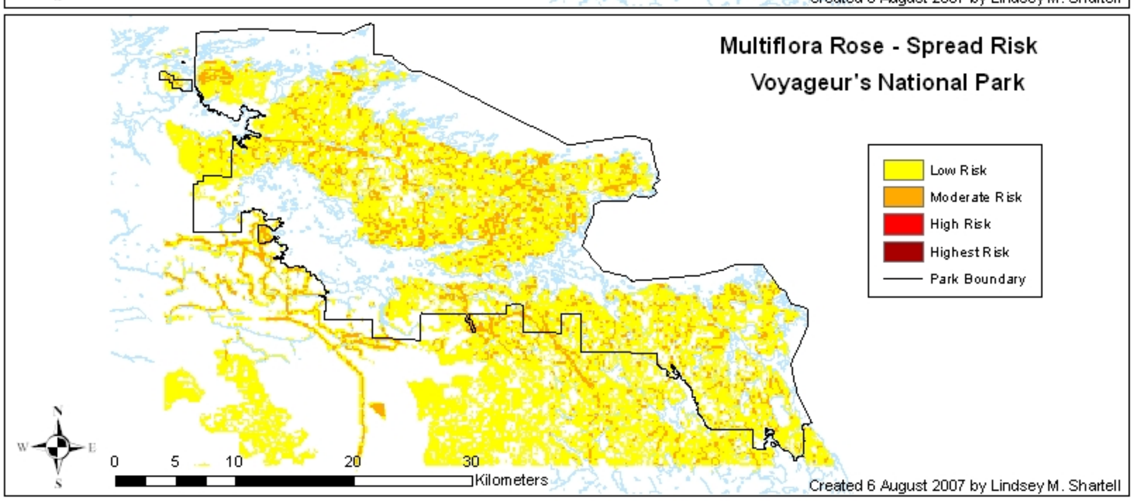




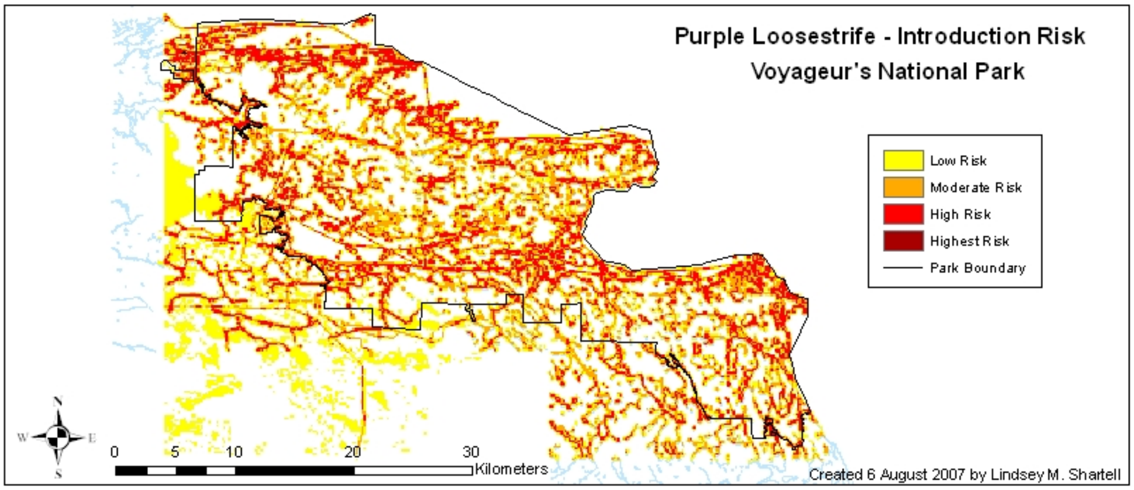
Appendix 3. Cont.

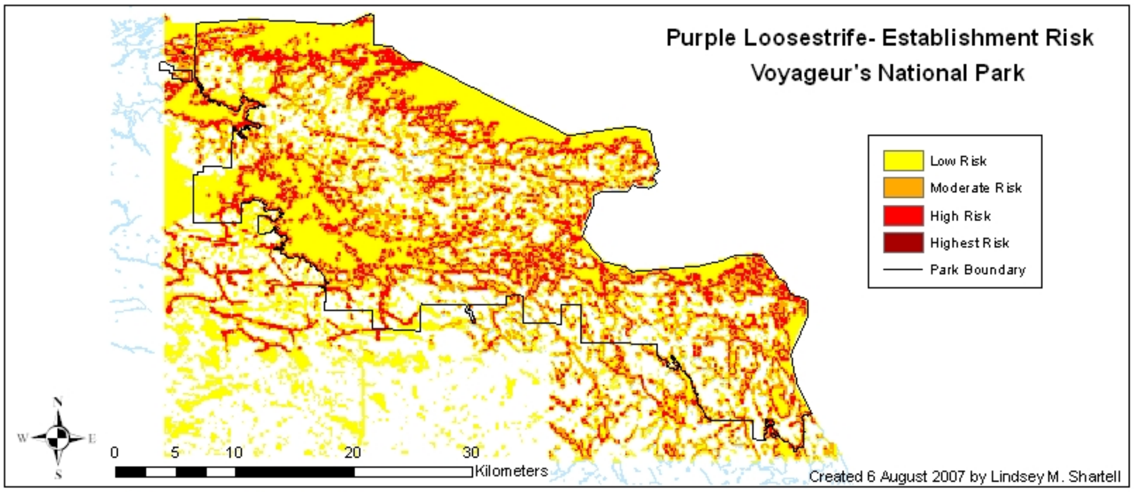


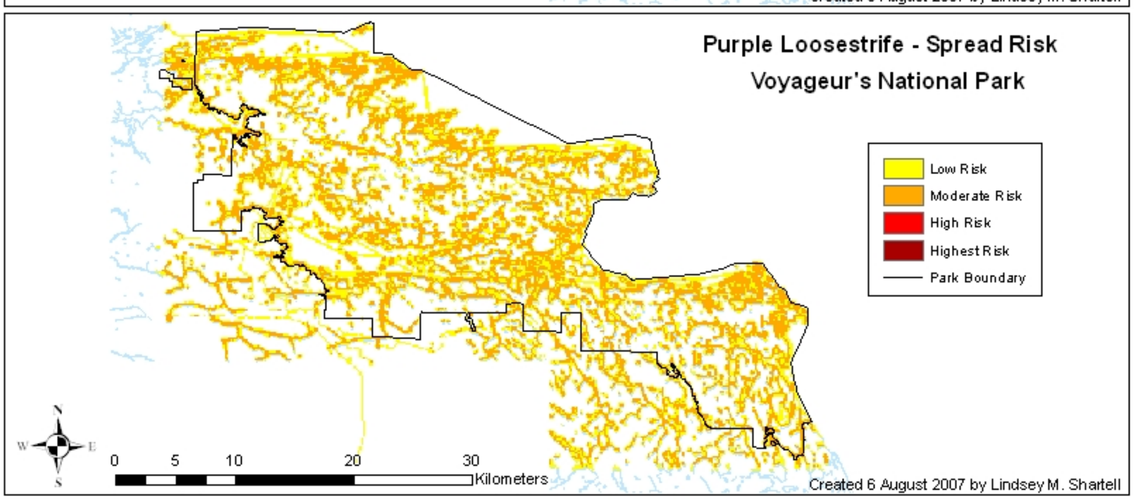




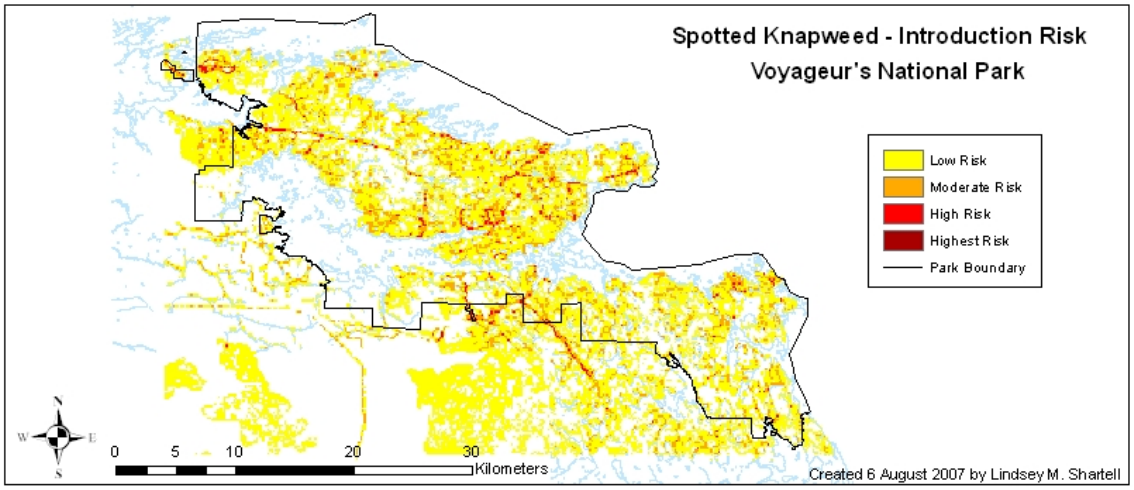
Appendix 3. Cont.

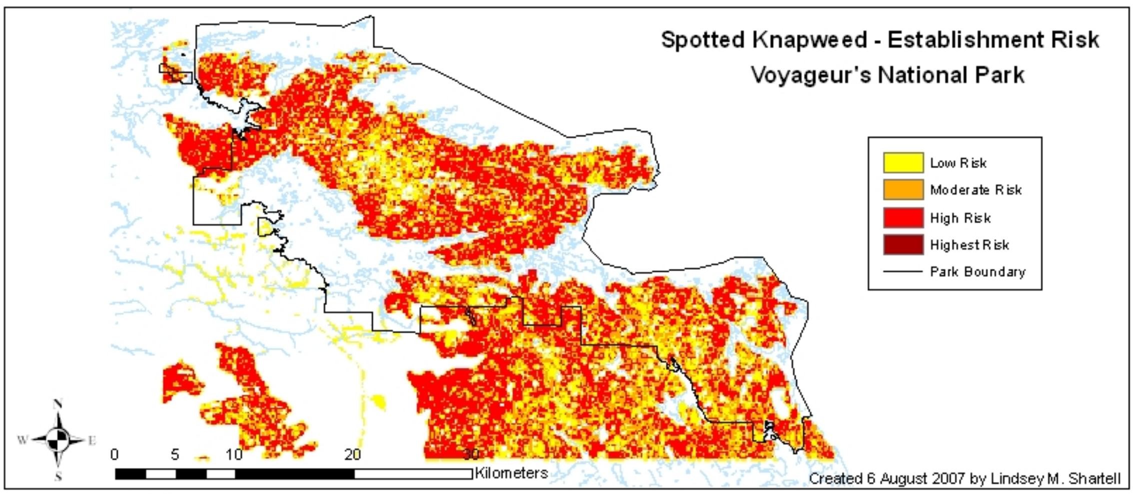


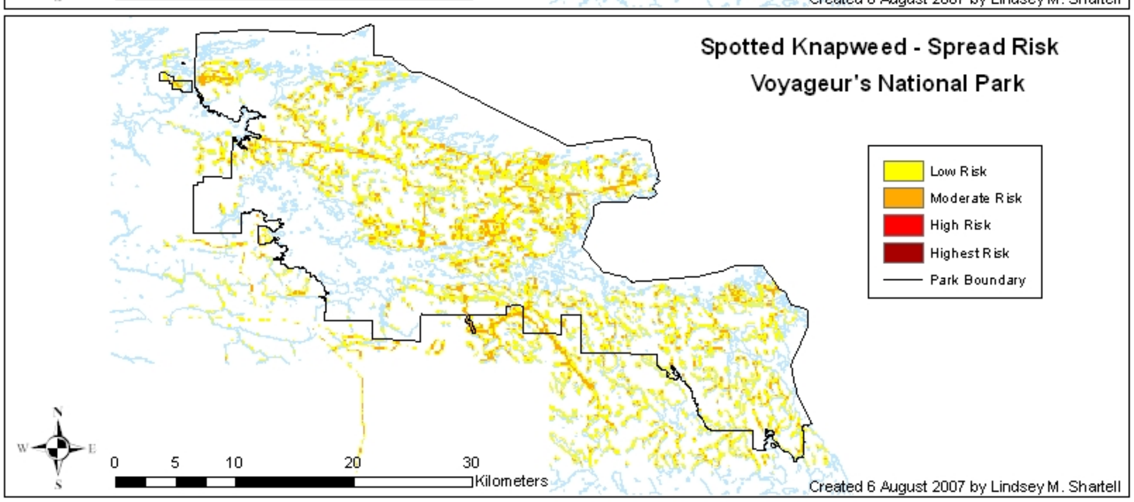




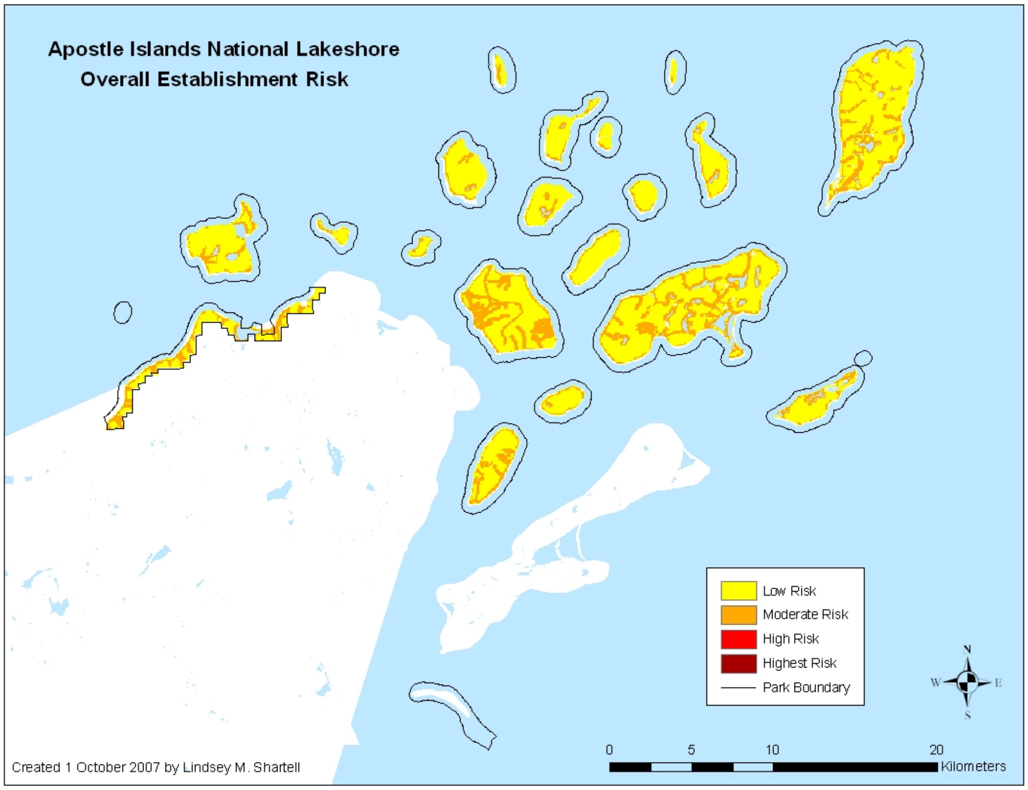
Appendix 3. Cont.



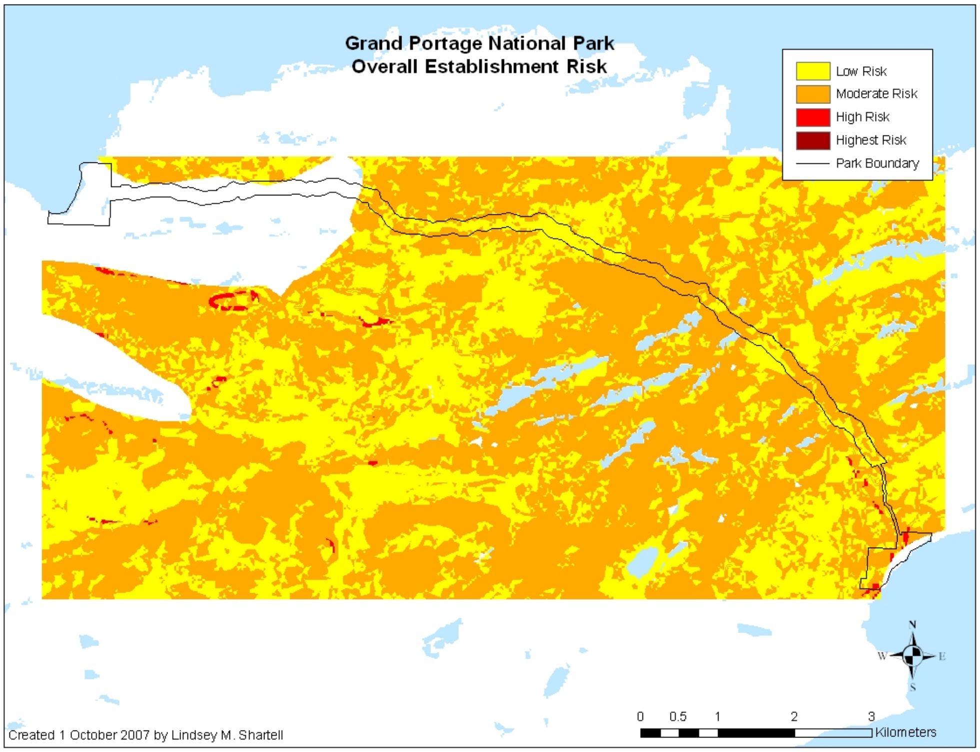


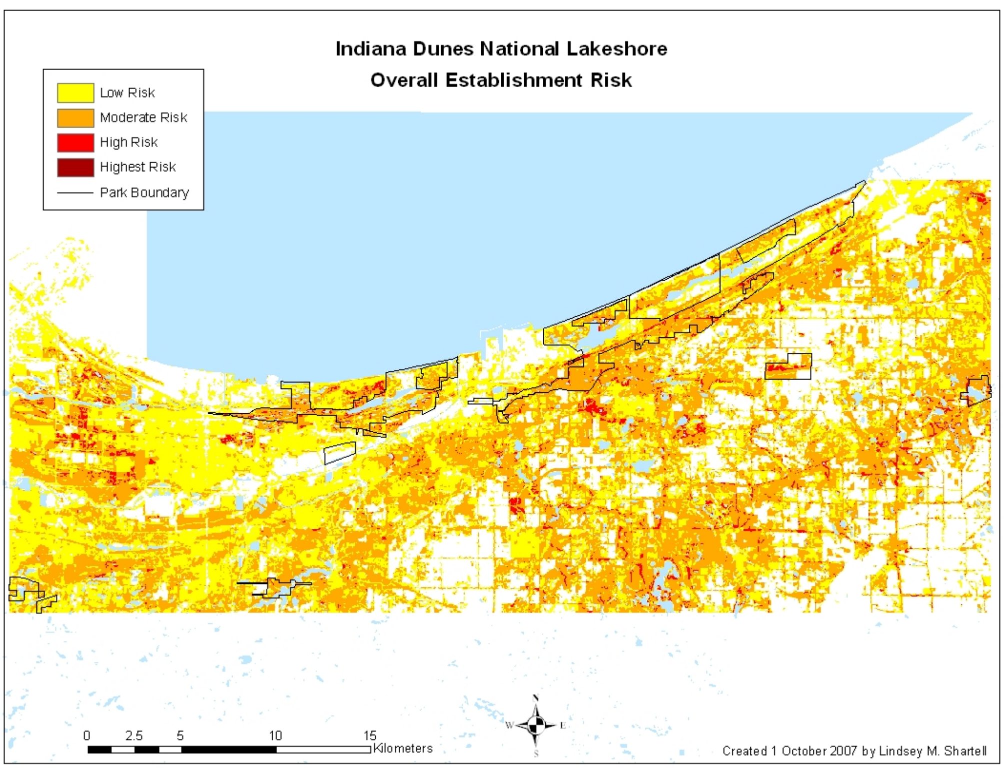


Appendix 4. Overall risk maps for the nine National Parks in the Great Lakes Network. Risk data was created by combining the establishment phase for all ten target invasive plants.

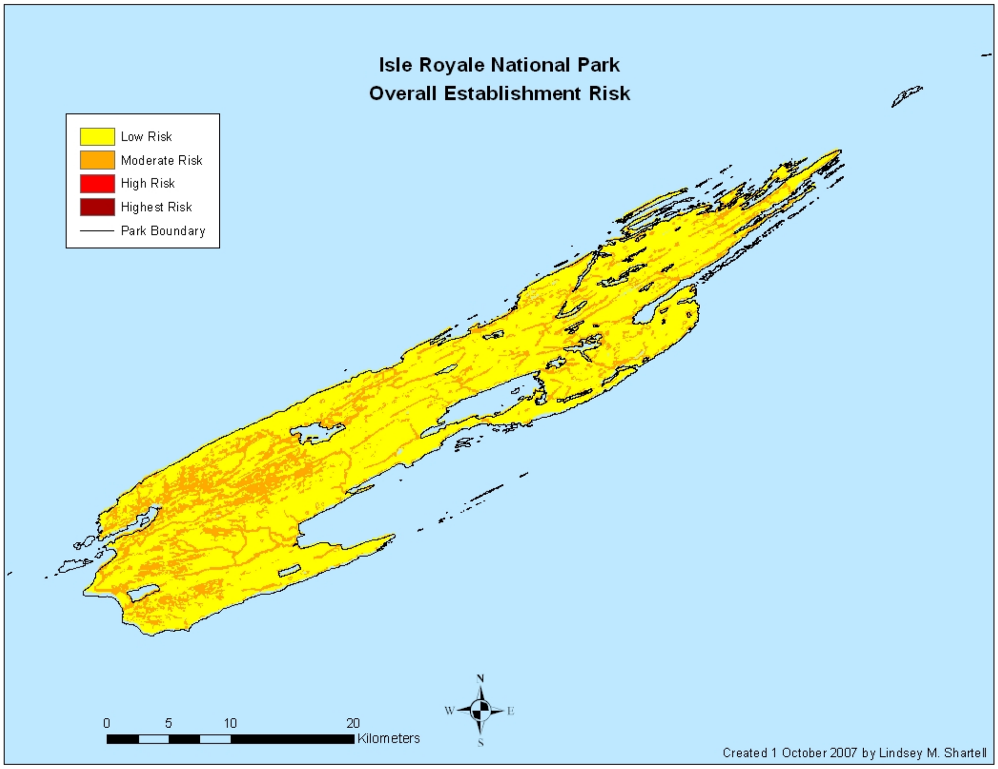


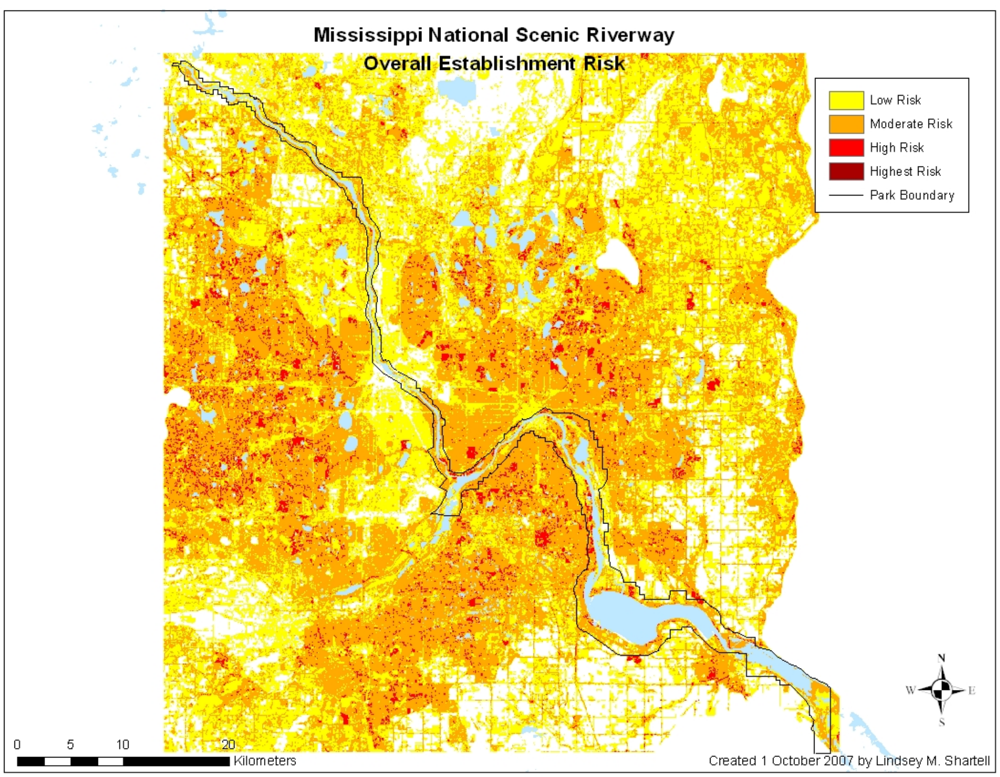
Appendix 4. Cont.



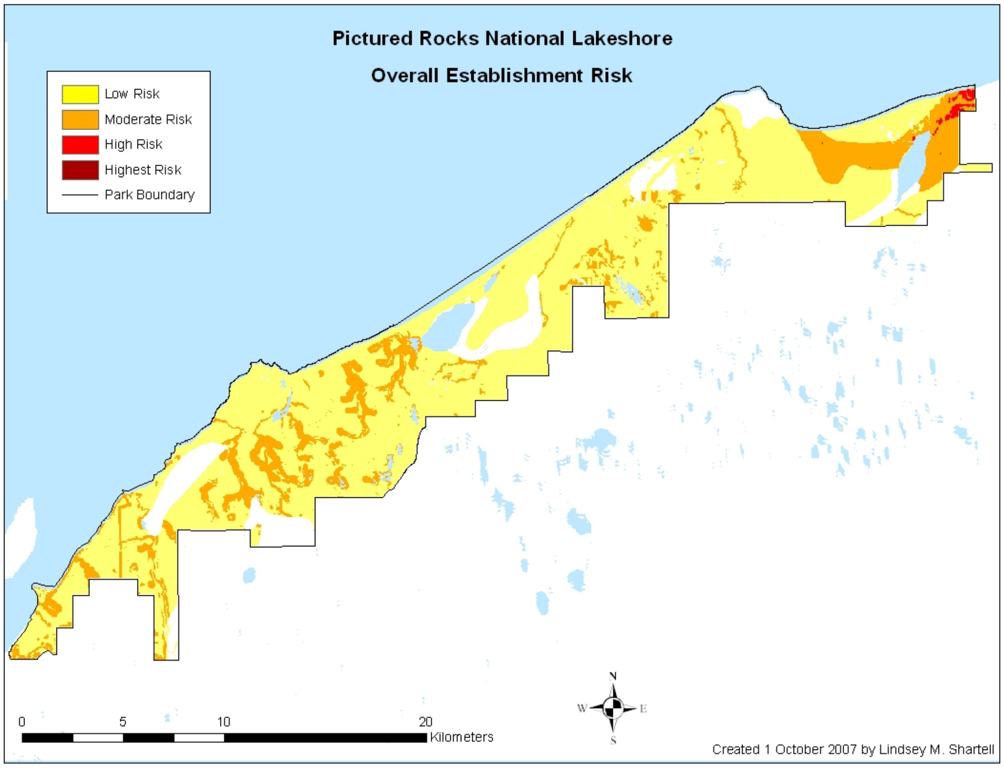


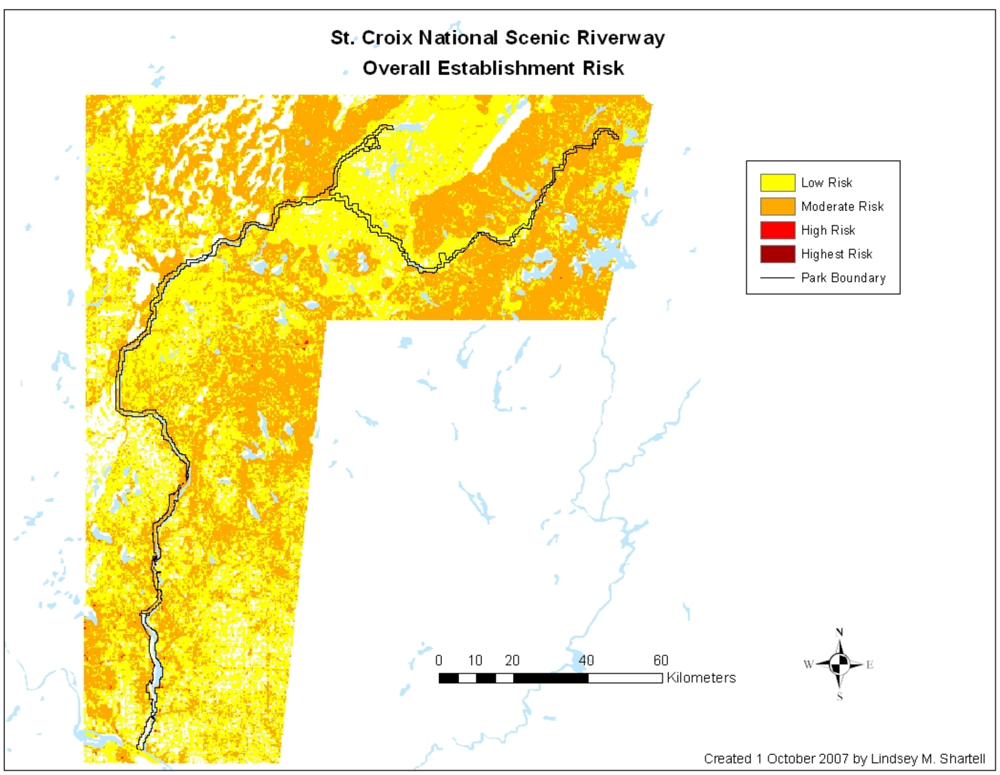
Appendix 4. Cont.



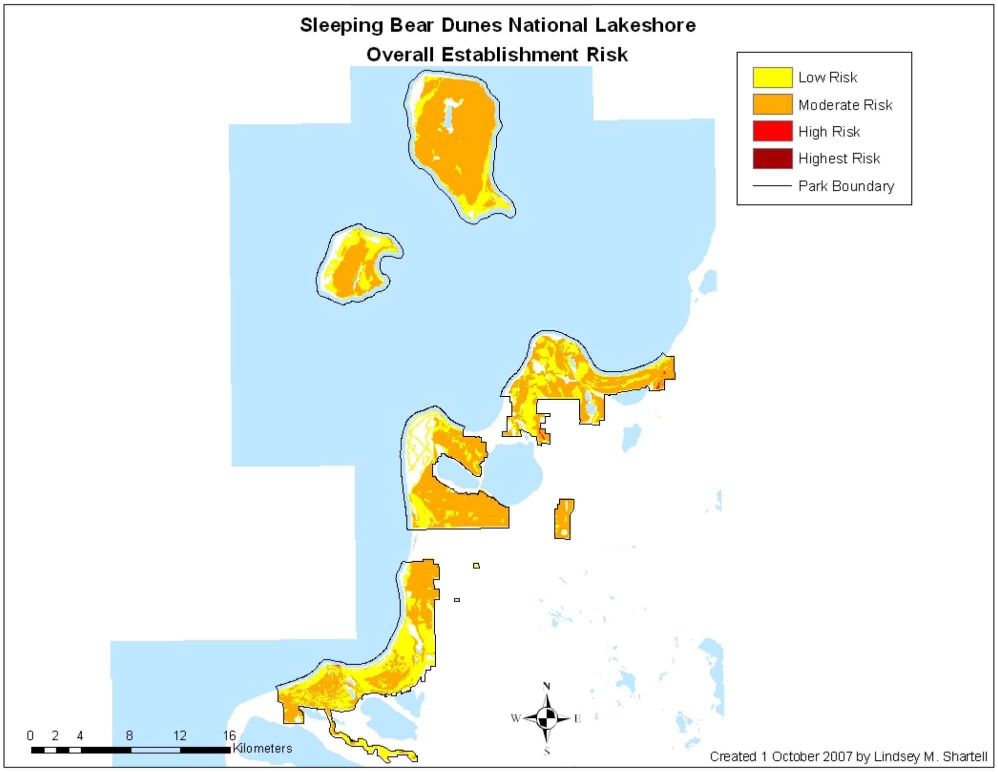


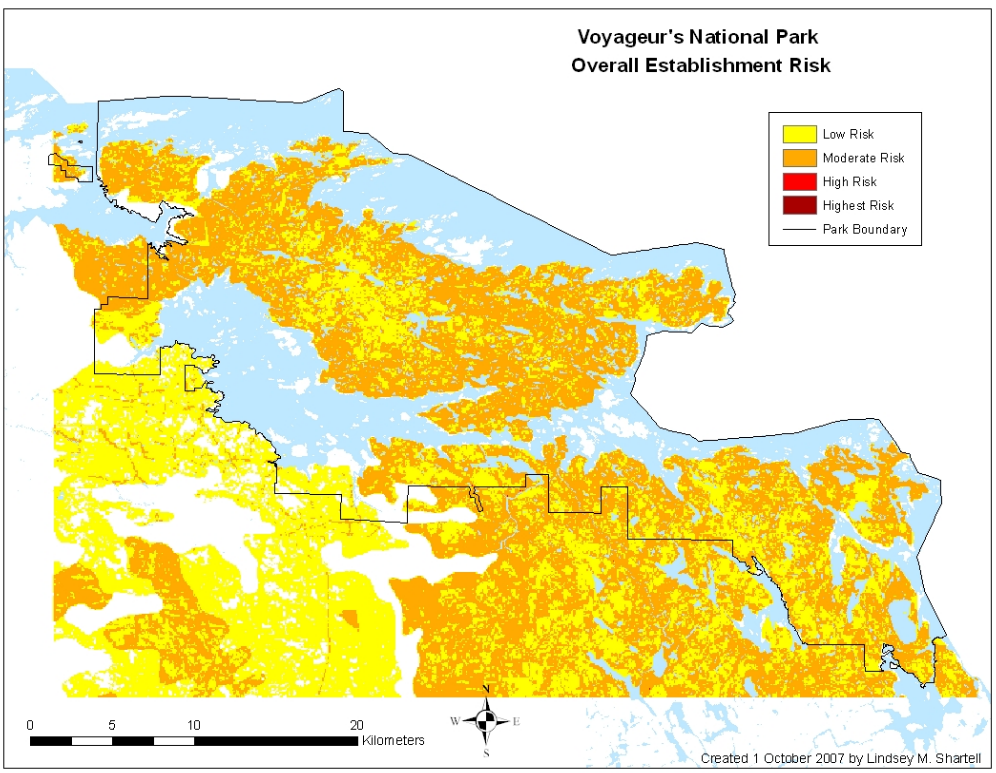
Appendix 4. Cont.





Appendix 4. Cont.





Chapter Three:

A Multi-Criteria Risk Model for Garlic Mustard (*Alliaria petiolata*) Invasion Across the Upper Peninsula of Michigan

Abstract

Throughout Midwestern forests, invasion of the exotic plant garlic mustard (*Alliaria petiolata* M. Bieb. [Cavara & Grande]) is an increasing problem. A multi-criteria risk model was developed for garlic mustard invasion in the Upper Peninsula of Michigan where the species is present but has not yet become widespread. The model utilizes geographic information system (GIS) data to predict the areas at highest risk for three phases of invasion: introduction, establishment, and spread. The model was run for the entire Upper Peninsula and for selected natural areas. The model predicted 13.0% of the Upper Peninsula to be at high risk, 32.8% at moderate risk, and 37.1% at low risk for the invasion of garlic mustard. The collection of distribution data for garlic mustard indicated that this species was present in five of the fifteen Upper Peninsula counties. However, field sampling of randomly generated sample points across the Upper Peninsula provided only two observations of garlic mustard presence. The low encounter rate during field sampling indicated that garlic mustard has not reached its full invasion potential in the region. By utilizing the models and risk maps as a guide, garlic mustard invasion can be managed more effectively through increased detection and more efficient control and containment.

Introduction

Garlic mustard (*Alliaria petiolata* M. Bieb. [Cavara & Grande], Figure 1) is one of the most pervasive exotic plants in Midwestern forests (Blossey *et al.* 2002). It is an obligate biennial herb in the mustard family (Brassicaceae), and can be identified by its heart-shaped, coarsely toothed leaves, white flowers, and seeds in slender pods. It is native to northern Europe, and was first documented in North America on the east coast in 1868 (Nuzzo 1993). Since then it has become widely established across eastern and central North America. Unlike many invasive plants, garlic mustard can invade and dominate the understory of forests, eliminating native vegetation and influencing ecosystem function. It decreases plant diversity by displacing native herbaceous plants and dominating the understory (Blossey *et al*. 2002). Garlic mustard has effects on many ecosystem components. Recent studies have found that garlic mustard disrupts mycorrhizal fungi associated with native tree seedlings, limiting their growth (Stinson *et al.* 2006). Garlic mustard also produces cyanide within the roots and aboveground tissues at levels that are toxic to humans and other vertebrates (Cipollini and Gruner 2007). This may have negative effects on the native butterfly *Pieris virginiensis* because of displacement of native host plants, also from the family Brassicaceae, by garlic mustard which may be toxic to the larvae (Porter 1994). The full consequences of garlic mustard in forests are not yet understood, but it is evident that the invasion of garlic mustard poses a significant threat to forested ecosystems.

Garlic mustard prefers sites dominated by mature deciduous forests (Meekins and McCarthy 1999, Myers and Anderson 2003), but it is also found in urban areas, disturbed areas, and along roads, railroads, and rivers (Nuzzo 1993). Garlic mustard can even survive in wetland habitats, moist woods, and swamp forests, despite being less competitive (Voss 1985). In Michigan, garlic mustard invades mainly deciduous woodlands, roadsides, and urban areas (Voss 1985). Invasion is promoted by disturbances, both human and natural (Anderson *et al*. 1996, Welk *et al*. 2002). Although most populations appear to be associated with some degree of disturbance (Byers and Quinn 1998), garlic mustard can invade relatively undisturbed forests with ease (Anderson *et al*. 1996). Studies indicate that garlic mustard is more likely to invade species-rich sites rather than species-poor sites (Blossey *et al*. 2002). Garlic mustard commonly inhabits mesic shaded areas, but can survive in well-drained sunny sites as well (Meekins and McCarthy 2002). In its native range it grows best on base-rich soils (Cavers *et al*. 1979). This association is also observed within its invaded range, but with a more noticeable absence from acidic soils (Nuzzo 1991). Prescribed burning has been tested as a control method for garlic mustard (Nuzzo 1991). Although high-intensity fires will kill rosettes and adult plants, the seeds are still viable and the disturbance may lead to a release from competition allowing further spread of garlic mustard (Luken and Shea 2000). Prescribed burning may also increase the presence of garlic mustard and enhance the growth of seedlings by removing the litter layer creating a suitable seedbed (Blossey *et al*. 2001).

The successful spread of garlic mustard is due to its ability to reproduce and disperse effectively. Garlic mustard has a biennial life cycle, which facilitates rapid invasion. The seeds germinate in the spring, forming rosettes that over-winter and emerge as flowering adults the following spring. The adults grow rapidly in early spring when most native plants are still dormant (Anderson *et al*. 1996). The flowers are usually pollinated by insects, but the plants have the ability to self-pollinate as well (Cruden *et al*. 1996). The seeds are held in siliques, and a single, robust plant can have as many as 7,900 seeds (Nuzzo 1993). Garlic mustard seeds are expelled up to two meters from the parent plant (Nuzzo 1999). Most seeds land near the parent plant, and germinate within one meter creating thick patches of garlic mustard that crowd out native plants (Drayton and Primack 1999). Long-distance dispersal of garlic mustard seedlings occurs by humans, animals, and water (Cavers *et al*. 1979). Roadways, trails, waterways, irrigation systems, and lakeshores offer pathways for human and natural dispersal. The use of heavy equipment in the construction of roads, dams, and bridges aids in the spread of seedlings between construction sites. The seedlings can also be transported over long distances by floodwaters (Nuzzo 1999). Seed dispersal occurs via deer, mice, and other small mammals (Blossey *et al*. 2001). Nuzzo (1999) calculated an average rate of spread of 5.4 m per year in a "high quality, relatively undisturbed forest", and noticed that spread was rapid into suitable microsites and slower into less suitable sites. The rate of spread of garlic mustard can be dramatically increased by a single disturbance event, and repeated disturbances can promote an even greater rate of spread as well as increased cover (Nuzzo 1999).

Predictive modeling of the potential distribution of garlic mustard has been attempted, producing alarming results. Using a bioclimatic model, Welk *et al.* (2002) found that garlic mustard could invade a large portion of North America ranging from the Rocky Mountains to the eastern coast. For Michigan, Welk *et al.* (2002) predicted a potential distribution across the whole state. Peterson *et al.* (2003) used ecological niche modeling to predict similar results, indicating that a majority of North America could potentially be invaded. Both methods were based solely on climate-related variables and predicted distribution on a national scale. Although large areas of potential distribution were predicted, invasion would be limited only to suitable sites across these areas, which was not illustrated in detail. Consequently, similar methods may not be appropriate for predicting garlic mustard invasion at the state level. A smaller scale model, however, would greatly improve predictions of potential habitat and probable invasion patterns. This approach would also permit the use of additional parameters and any available Geographic Information System (GIS) data.

Garlic mustard is not yet widely distributed in the Upper Peninsula of Michigan, however it has the potential to be a serious, wide-ranging problem (Voss 1985). Voss (1985) indicated that garlic mustard was present in only two of the 15 Upper Peninsula counties. This is consistent with the distribution data found on the USDA PLANTS Database (accessed June 2007). As expected, further garlic mustard invasions have been identified in additional counties throughout the Upper Peninsula. An updated distribution map is needed to determine the full extent of the current garlic mustard invasion. Furthermore, a multi-criteria risk model and detailed risk maps would assist with the monitoring and management of garlic mustard in the Upper Peninsula.

This project aimed to update the distribution map and create a multi-criteria risk model for garlic mustard in the Upper Peninsula of Michigan. The specific objectives of this study were to: (1) create an updated distribution map for garlic mustard in the Upper Peninsula of Michigan and surrounding areas, (2) develop and test a multi-criteria risk model for garlic mustard invasion risk, and (3) utilize the model to create risk maps for the Upper Peninsula as a whole and for smaller selected natural areas throughout the Upper Peninsula.

Methods

Study Area

The Upper Peninsula contains 4,261,048 hectares, which is about one-third of the land area in Michigan. However, the region has only three percent of the human population of Michigan. The Upper Peninsula also includes only 15 of the 83 counties in Michigan. The landscape is comprised mainly of deciduous, coniferous, and mixed forests and wetlands. Much of the forest is found within the Ottawa and Hiawatha National Forests and State forest lands. Other federally-managed natural areas include Pictured Rocks National Lakeshore, Isle Royale National Park, and Seney National Wildlife Refuge. Many smaller natural areas are distributed among these contiguous forested areas.

Distribution Mapping

Throughout the Upper Peninsula the known locations of garlic mustard were collected in the first step to create an updated distribution map. Invasion data were provided by the U.S. Forest Service, U.S. Fish and Wildlife Service, and the Michigan Department of Natural Resources. The distribution map from the USDA Plants database was used to create an initial county map indicating presence or absence of garlic mustard. Known occurrences and collected data were added to this map to create a current county-wide distribution map. The spatial GIS data were also used to create a detailed map of known invasions in the Upper Peninsula and surrounding areas. When available, ArcGIS shapefiles were utilized to obtain or create spatial data. All other data were converted into point or polygon ArcGIS shapefiles. The Michigan GeoRef projection and NAD 1983 were used as the standard spatial projection and datum.

Model Development

A multi-criteria risk model for garlic mustard invasion in the Upper Peninsula of Michigan was developed within ArcGIS 9.2 using ModelBuilder (ESRI 2006). The model utilizes GIS data to predict the areas at highest risk for the three phases of invasion: introduction, establishment, and spread. Two versions of the model were created, with only one difference. The first version was developed for the entire Upper Peninsula and converted all parameter data into 30 m x 30 m raster grids. The second model was developed to predict risk in selected natural areas. This model converted all data into smaller 10 m x 10 m raster grids to account for the higher-quality, finer-scale data that were available. The natural areas selected to be modeled were the western and eastern portions of the Hiawatha National Forest (WHNF and EHNF), Pictured Rocks National Lakeshore (PIRO), Ottawa National Forest (ONF), and Seney National Wildlife Refuge (SNWR, Figure 2). A literature review was done to determine the parameters for the model (see Appendix 1 for sources consulted). A list of factors affecting invasion risk was created for each phase of invasion. From this list, the parameters found to be important for predicting invasion, and that had obtainable spatial data, were: distance to roads, railroads, trails, distance to water, vegetation type, soil moisture, soil pH, and known invasion. A parameter called “other features” was also included in the model.

The data utilized for this parameter varied by the area being modeled and the data available. It represented points of disturbance, such as campgrounds or other features that have a known effect on invasion. Multiple point shapefiles could be entered as parameters.

For each model parameter, GIS data was obtained or created. Transportation and hydrology layers were obtained from Michigan Geographic Framework data, which were available from the Michigan Geographic Data Library (State of Michigan 2007). For the Upper Peninsula, the 2001 National Land Cover Dataset (NLCD) was used for vegetation. Finer-scale vegetation data were available for PIRO and SNWR. Soil data were obtained from the Natural Resource Conservation Service (NRCS), and used to create the parameters of soil drainage and soil pH. State Soil Geographic (STATSGO) data, which had a scale of 1:250,000, were used for the Upper Peninsula. When available, Soil Survey Geographic (SSURGO) data, which were collected by county at a scale of 1:24,000, were used to run the model at smaller selected natural areas. For the parameter “other features” any accessible and applicable GIS data were utilized, in particular the location of campgrounds and known disturbance sites. One significant source of disturbance and an area frequently observed as an initial invasion point was campgrounds. However, there were no current GIS shapefiles found for campgrounds. Thus, a point shapefile was created of all State Forest campgrounds using a Michigan atlas and location details from a state forest campground brochure (MI DNR 2007). Campground locations were also collected from the National Park Service and the US Forest Service. There are no campgrounds within the boundaries of SNWR. The shapefiles created for the distribution of garlic mustard were used for the known invasion parameter.

Based on the literature review, risk ratings from zero to five were assigned to each level of each parameter based on its relationship to the invasion and survival of garlic mustard. A risk of five was assigned to conditions that were very suitable and frequently associated with the presence of garlic mustard. A risk of four indicated moderately suitable conditions, three indicated somewhat suitable conditions, two indicated rarely suitable conditions, one indicated very rarely suitable conditions, and zero indicated conditions that were completely unsuitable. When possible, a standard curve was applied to continuous data using information on when risk begins, peaks, and ends. For categorical data (i.e. vegetation), risk values were assigned by considering the suitability of each of the vegetation types to invasion and survival. Different risks were assigned to the same parameter for each of the three phases of invasion, when applicable. Each parameter in the model was assigned a rank and a confidence level. From these calculations, the influence value of the parameter was determined. The parameters were combined by means of a weighted overlay assigning the influence as the percent weight.

Introduction, establishment, and spread of garlic mustard were each based on a different set of parameters and influences (Table 1, Figure 3). Introduction was based on a weighted overlay of dispersal, vegetation, soil drainage, soil pH, and other features. For dispersal, the distance from each pixel to a road, trail, or railroad, and distance to water, was calculated. The distances were also assigned a risk value from zero to five. Each vegetation type was assigned a risk value from zero to five based on suitability for garlic mustard seedling invasion and survival. The drainage rating and average pH of each soil type was used to assign a risk value based on the preference of garlic mustard. Any information on other features was combined and areas with disturbance present received a value of five indicating high risk. Establishment was based on a weighted overlay of introduction risk, vegetation, soil moisture, and soil pH. The introduction risk was the value assigned by the weighted overlay of factors affecting introduction. Spread was based on a weighted overlay of establishment risk, dispersal, other features, and known garlic mustard invasion. The distance to known garlic mustard locations was calculated, and a risk value was assigned to these distances.

The models were run for the entire Upper Peninsula, as well as for the selected natural areas. The appropriate GIS data were selected for each parameter on the model dialog screen. The GIS data was processed through the steps of the model creating the three output files, one for each phase of invasion. Each pixel in the output received a risk rating between zero and five. The output files were used to create maps highlighting the areas at risk for invasion. The areas at highest risk, a rating of five, were shown in red. Areas of moderate risk, a rating of four, were shown in orange. Low risk areas, a rating of three, were shown in yellow. Areas with little to no risk for invasion (ratings two to zero) were shown in gray.

Accuracy Assessment

The accuracy of the model was assessed using two methods. First, the known garlic mustard invasion data were compared to the model output. The percentage of pixels with known garlic mustard that received a high risk of invasion was calculated for each phase. This was done for the Upper Peninsula as well as the selected natural areas.

Following analysis, the models were slightly adjusted by altering the weights of each parameter to obtain best accuracy, but being careful to avoid over-fitting. This was done by adjusting values individually, while monitoring the percent of the area at risk for large increases.

Model outputs were also ground-truthed. This method of evaluation tested the performance of the adjusted model and also tested the utility of the model in the field. Two of the natural areas, ONF and SNWR, were selected for testing the model. At each area, 25 random points were generated using the program Random Point Generator (Sawada 2002). An additional 40 points were generated along selected Upper Peninsula roads and at campgrounds occurring along these roads. The GPS coordinates of each point were downloaded onto a Garmin GPS Map 76 unit. Using maps of the random points, a field crew navigated to within 15 m of each point. The exact coordinates were recorded at each point, as well as the accuracy of the GPS unit. The points were later adjusted to the exact coordinates recorded in the field. At each random point the presence and abundance of garlic mustard was recorded within a 30 m x 30 m plot. The random point served as the center of the plot, and a compass was used to align the plot with the cardinal directions. Abundance was based on a visual assessment of percent cover. The presence data were used to calculate the percent of pixels with garlic mustard and a high risk of invasion using the predicted risks for each phase. The absence data were also assessed, but with the understanding that garlic mustard has not yet reached its full invasion potential. The abundance data were used to qualitatively assess the differences in risk among the three phases of invasion.

Results

Garlic Mustard Distribution

Land managers reported known garlic mustard invasions in five of the 15 Upper Peninsula counties (Figure 4). Counties shown to have this species were Alger, Gogebic, Mackinac, Marquette, and Ontonagon. Surrounding invasions were reported in the Lower Peninsula of Michigan and in Wisconsin. These populations of garlic mustard were located at numerous sites across the Upper Peninsula. Forty-five points with garlic mustard were mapped in and around the ONF. This included three points within the Porcupine Mountains Wilderness State Park, and 27 points within surrounding forests in Wisconsin. Garlic mustard was reported at six points in and around the WHNF, including a large invasion just outside the National Forest boundary along the AuTrain Basin. Invasions were located in Marquette County at nine points. In Mackinac County, garlic mustard was documented near the Cut River Bridge.

Model Accuracy Assessment

Assessment of model accuracy using the known garlic mustard invasion data resulted in an average across phases of 99.0% ($\pm$ 1.4%) of pixels with garlic mustard correctly predicted at risk for garlic mustard (rating of $\geq$ three). This was based on 494 points for which the presence of garlic mustard was known. Of these same 494 locations, 88.8% ($\pm$ 4.2%) were assigned a moderate or high risk (rating of four or five). These percentages were an average for the three phases of invasion. Broken down by phase the locations of known invasions predicted to have moderate or high risk were 87.3% for introduction, 93.5% for establishment, and 85.6% for spread. Two of the selected natural areas, the WHNF and the ONF, had known garlic mustard invasions, thus the model outputs for these areas were also assessed. Garlic mustard was found at six sites making up 320 points in the WHNF. Of these locations, 100.0% were at risk for garlic mustard, and 95.9% (± 1.9%) had a moderate or high risk. There were 18 known invasion points within the ONF, of which 100.0% were at risk for garlic mustard. Of these, 57.4% (± 11.6%) had a moderate or high risk.

During the field sampling period, garlic mustard was only found at two of the 90 points sampled. The first location was at South Manistique Lake State Forest Campground, and the second was near a known invasion site at Forest Lake State Campground. The risks predicted at these points were high risk of establishment at South Manistique and moderate risk at Forest Lake.

Model Results

The model predicted a range of risks across the Upper Peninsula, which varied slightly by phase of invasion (Figure 5). The model predicted 13.0% of the Upper Peninsula to be at high risk (rating of five) for establishment of garlic mustard invasion. An additional 32.8% was at moderate risk (risk rating of four) and 37.1% was at low risk (risk rating of three) for establishment. For the selected natural areas, the ONF showed the most extensive area at risk for garlic mustard. The WHNF and EHNF also showed much of the forested land at risk. For both Forests, roughly three-quarters of the area were at risk for garlic mustard invasion. The models predicted the western portion to have slightly higher risk, with 38.2% of the WHNF and surrounding areas at moderate or high risk for establishment, compared to 27.0% of the eastern portion. PIRO showed areas of establishment risk, but this was limited to 38.7% of the Park. There were only small areas at risk for garlic mustard establishment at SNWR, which made up only 9.2% of the Refuge.

Discussion

The multi-criteria risk model created for garlic mustard correctly identified invaded areas as high risk. The risk maps created from the model output provided a unique view of the model results. Garlic mustard invasion risk was high in many areas of the Upper Peninsula. These results were consistent with the findings of Welk *et al.* (2002) and Peterson *et al.* (2003), who both used models to predict that the Upper Peninsula of Michigan would be suitable habitat for invasion. The risk map for the spread phase showed few areas at high risk. This was due in part to the known invasion parameter. Because the known invasions were point data, which when converted to pixels made up only 0.001% of the area of the Upper Peninsula, they lowered the predicted risk across un-invaded areas to moderate rather than high. This factor is, however, important to include when utilizing the model to assist with planning control efforts. This problem can be avoided by modeling the spread phase over smaller areas, with more in-depth data on known invasions.

The low encounter rate of garlic mustard during field sampling and distribution data collection indicate that garlic mustard has not reached its full invasion potential in the Upper Peninsula. This presents an ideal setting for the application of the predictive model and risk maps as resources for monitoring introduction, establishment, and spread of garlic mustard, as well as prioritizing known invasion for management.

The risk predictions for selected natural areas corresponded with known garlic mustard invasions. ONF had several known garlic mustard invasions and showed many areas at high risk. The WHNF also had known invasions and showed many areas at risk for garlic mustard. PIRO showed less risk and had no known invasions. SNWR, which is made up primarily of wetlands and xeric, conifer-dominated forests, showed very little risk for garlic mustard, which prefers hardwood-dominated sites. Consequently, even though this area is located between established populations in the Eastern Upper Peninsula, garlic mustard has not yet been observed at SNWR.

Garlic mustard was only found at two of the 90 field sampling points. For the two natural areas selected for sampling, SNWR was not expected to have garlic mustard present because of the lack of suitable habitat, but ONF has known invasions and was expected to provide presence data points. The reason no garlic mustard was found at ONF may be due in part to recent control efforts to remove and treat garlic mustard within the Forest. The lack of presence data made the analysis of the field data difficult. For this reason the assessment of the model accuracy was based on the comparison of predicted risks and known garlic mustard invasions. Further field sampling may be necessary to fully validate the model.

Conclusion

The multi-criteria risk model developed here determines the level of risk based on biological, environmental, and human-induced factors that are significant in the Upper Peninsula of Michigan. The model in its current form is not applicable to sites outside this region. However, for other similar areas, models could be produced and further developed to include additional parameters such as land use, elevation, and climate. Many factors affecting the invasion of garlic mustard cannot be mapped spatially or applied in models. Stochastic events may play a large role in the location of initial introductions. As a result, monitoring for garlic mustard should not be based solely on the results of predictive models, but guided by them.

By utilizing the risk maps as a guide, the time and resources required for monitoring and management of garlic mustard can be reduced. Monitoring efforts can focus on areas at high risk for introduction and establishment, and control efforts can be prioritized to areas with a high risk of establishment and spread. The insight gained from the model and risk maps can increase the success of monitoring and control efforts for garlic mustard.

Literature Cited

Anderson, R. C., S. S. Dhillion, and T. M. Kelley. 1996. Aspects of the ecology of an invasive plant, garlic mustard (*Alliaria petiolata*), in central Illinois. Restor Ecol 4:181-191.

Blossey, B., V. Nuzzo, H. Hinz, and E. Gerber. 2001. Developing biological control of *Alliaria petiolata* (M. Bieb.) Cavara and Grande (garlic mustard). Nat Areas J 21:357-367.

Blossey, B., V. A. Nuzzo, H. L. Hinz, and E. Gerber. 2002. Garlic Mustard *In:* Van Driesche, R., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 365-372. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Byers, D. L. and J. A. Quinn. 1998. Demographic variation in *Alliaria petiolata* (Brassicaceae) in four contrasting habitats. J Torrey Bot Soc 125:138-149.

Cavers, P. B., M. I. Heagy, and R. F. Kokron. 1979. The biology of Canadian weeds. 35. *Alliaria petiolata* (M. Bieb.) Cavara and Grande. Can J Plant Sci 59:217-229.

Cipollini, D. and B. Gruner. 2007. Cyanide in the chemical arsenal of garlic mustard, *Alliaria petiolata.* J Chem Ecol 33:85-94.

Cruden, R. W., A. M. McClain, and G. P. Shrivastava. 1996. Pollination and breeding system of *Alliaria petiolata* (Brassicaceae). Bull Torrey Bot Club 123:273-280.

Drayton, B. and R. B. Primack. 1999. Experimental extinction of garlic mustard (*Alliaria petiolata*) populations: implications for weed science and conservation biology. Biol Invasions 1:159-167.

ESRI. 2006. ArcGIS 9.2. Badlands, CA.

Luken, J. O. and M. Shea. 2000. Repeated prescribed burning at Dinsmore Woods State Nature Preserve (Kentucky, USA): Responses of the understory community. Nat Areas J 20:150-158.

Meekins, J. F. and B. C. McCarthy. 1999. Competitive ability of *Alliaria petiolata* (garlic mustard, Brassicaceae), an invasive, nonindigenous forest herb. Int J Plant Sci 160:743-752.

Meekins, J. F. and B. C. McCarthy. 2002. Effect of population density on the demography of an invasive plant (*Alliaria petiolata*, Brassicaceae) population in a southeastern Ohio forest. Am Midl Nat 147:256-278.

MI DNR. 2007. Michigan State Forest Campgrounds and Pathways: Upper Peninsula. Michigan Department of Natural Resources.

Myers, C. V. and R. C. Anderson. 2003. Seasonal variation in photosynthetic rates influences success of an invasive plant, garlic mustard (*Alliaria petiolata*). Am Midl Nat 150:231-245.

Nuzzo, V. A. 1991. Experimental control of garlic mustard [*Alliaria petiolata* (Bieb.) Cavara and Grande] in northern Illinois using fire, herbicide, and cutting. Nat Areas J 11:158-167.

Nuzzo, V. A. 1993. Distribution and spread of the invasive biennial *Alliaria petiolata* (garlic mustard) in North America *In:* McKnight, B. N. (ed.). *Biological Pollution: The Control and Impact of Invasive Exotic Species*. pp 137-146. Indiana Academy of Science, Indianapolis.

Nuzzo, V. A. 1999. Invasion pattern of the herb garlic mustard (*Alliaria petiolata*) in high quality forests. Biol Invasions 1:169-179.

Peterson, A. T., M. Papes, and D. A. Kluza. 2003. Predicting the potential invasive distributions of four alien plant species in North America. Weed Sci 51:863-868.

Porter, A. 1994. Implications of introduced garlic mustard (*Alliaria petiolata*) in the habitat of *Pieris virginiensis* (Pieridae). J Lepid Soc 48:171-172.

Sawada, M. 2004. Random Point Generator. Available from ESRI ArcScripts (http://arcscripts.esri.com/details.asp?dbid=12098).

State of Michigan. 2007. Michigan Geographic Data Library (http://mcgi.state.mi.us/mgdl/, 1 January 2007). Center for Geographic Information, Department of Information Technology.

Stinson, K. A., S. A. Campbell, J. R. Powell, B. E. Wolfe, R. M. Callaway, G. C. Thelen, S. G. Hallett, D. Prati, and J. N. Klironomos. 2006. Invasive plant suppresses the growth of native tree seedlings by disrupting belowground mutualisms. PLoS Biol 4(5): e140.

USDA, NRCS. 2007. The PLANTS Database (http://plants.usda.gov, 16 May 2007). National Plant Data Center, Baton Rouge, LA 70874-4490 USA.

Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Welk, E., K. Schubert, and M. H. Hoffmann. 2002. Present and potential distribution of invasive garlic mustard (*Alliaria petiolata*) in North America. Divers Distrib 8:219-233.

Table 1. The influence values (percent weights) assigned to the parameters used to determine risk for each phase of invasion in the multi-criteria risk model.

Parameter	Percent Weight
Introduction	
Dispersal	0.43
Vegetation	0.22
Soil Drainage	0.14
Soil pH	0.14
Other Features	0.07
Establishment	
Introduction Risk	0.11
Vegetation	0.45
Soil Drainage	0.22
Soil pH	0.22
Spread	
Establishment Risk	0.53
Dispersal	0.26
Other Features	0.13
Known Invasion	0.08

Figure 1. An adult garlic mustard (*Alliaria petiolata*) plant.

Photo credit: Lindsey Shartell.

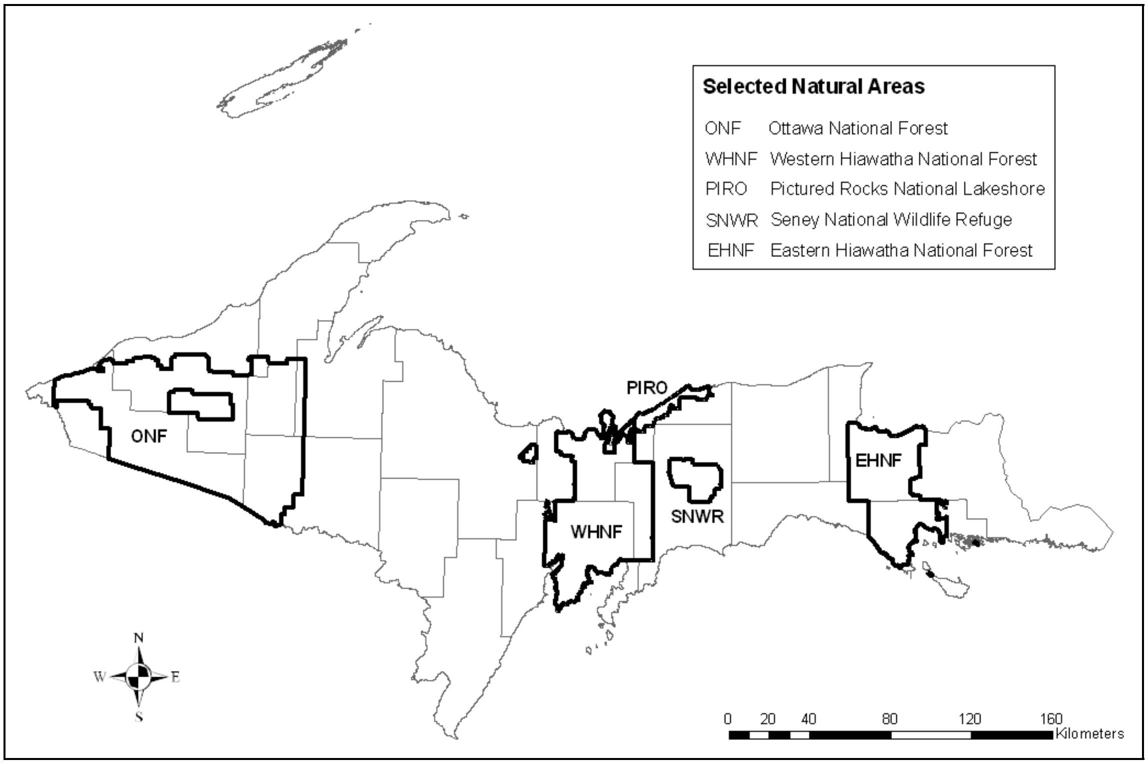


Figure 2. Selected natural areas for which the multi-criteria model was run and tested.

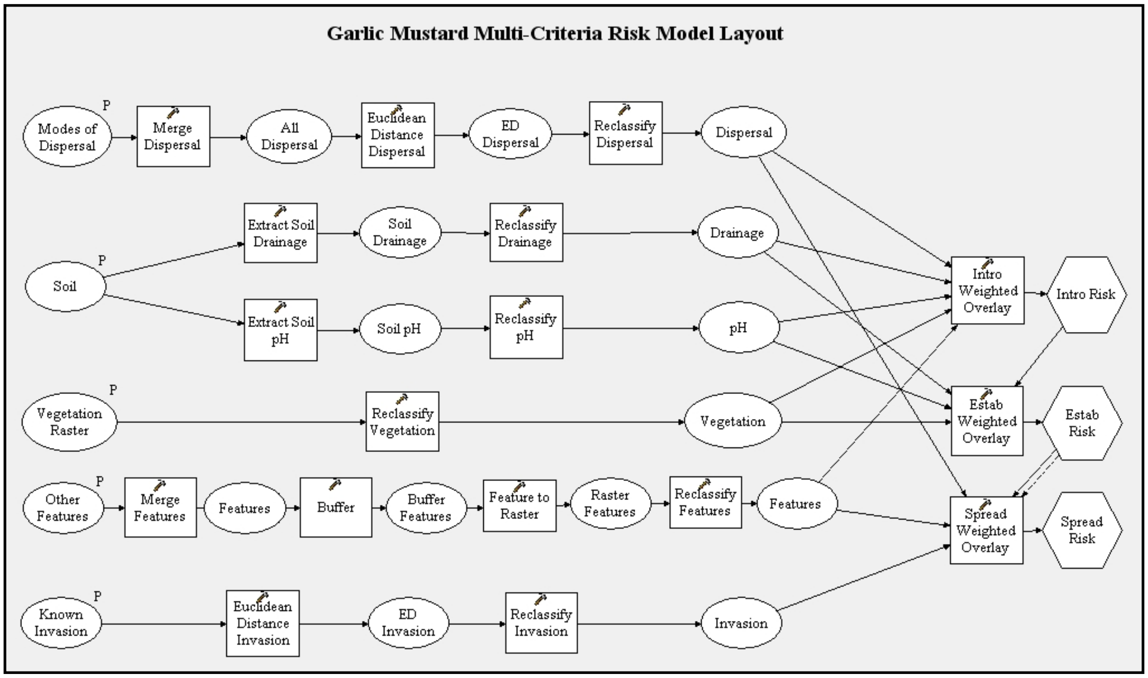


Figure 3. The multi-criteria risk model layout showing the parameters used by each phase of invasion (P = user defined parameter).

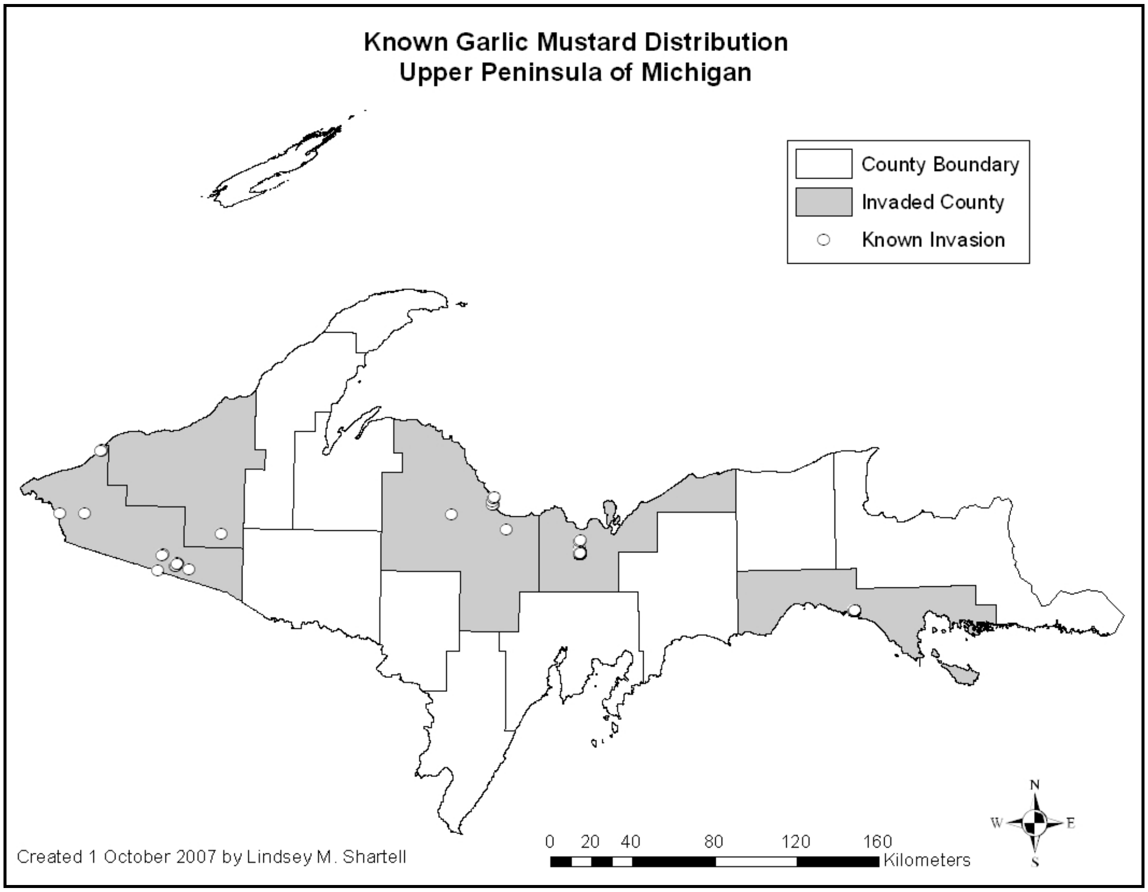


Figure 4. Known distribution map for garlic mustard in Upper Peninsula counties.

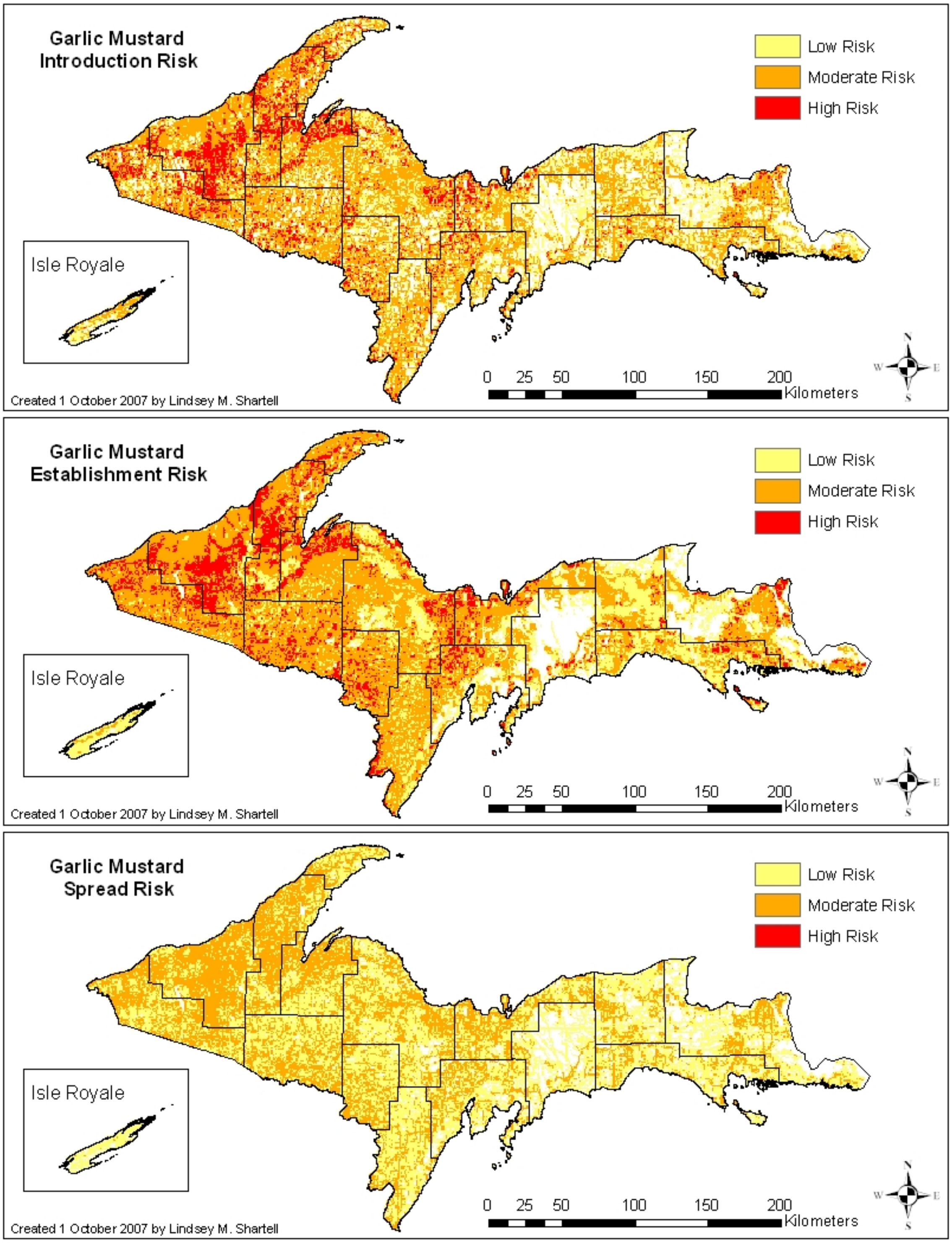


Figure 5. Garlic mustard invasion risk at three phases of invasion for the Upper Peninsula of Michigan.

Appendix 1. Bibliography used for the garlic mustard literature review to assist with model development.

Anderson, R. C., S. S. Dhillion, and T. M. Kelley. 1996. Aspects of the ecology of an invasive plant, garlic mustard (*Alliaria petiolata*), in central Illinois. Restor Ecol 4:181-191.

Baskin, J. M. and C. C. Baskin. 1992. Seed germination biology of the weedy biennial *Alliaria petiolata*. Nat Areas J 12:191-197.

Blossey, B. 1999. Before, during and after: the need for long-term monitoring in invasive plant management. Biol Invasions 1:301-311.

Blossey, B., V. Nuzzo, H. Hinz, and E. Gerber. 2001. Developing biological control of *Alliaria petiolata* (M. Bieb.) Cavara and Grande (garlic mustard). Nat Areas J 21:357-367.

---------- 2002. Garlic Mustard *In:* Van Driesche, R., B. Blossey, M. Hoddle, S. Lyon, and R. Reardon (eds.). *Biological Control of Invasive Plants in the Eastern United States*. pp 365-372. U. S. Department of Agriculture, Forest Service, FHTET-2002-04.

Bossdorf, O., S. Schroder, D. Prati, and H. Auge. 2004. Palatability and tolerance to simulated herbivory in native and introduced populations of *Alliaria petiolata* (Brassicaceae). Am J Bot 91:856-862.

Brown, W. T., M. E. Krasny, and N. Schoch. 2001. Volunteer monitoring of nonindigenous invasive plant species in the Adirondack Park, New York, USA. Nat Areas J 21:189-196.

Byers, D. L. and J. A. Quinn. 1998. Demographic variation in *Alliaria petiolata* (Brassicaceae) in four contrasting habitats. J Torrey Bot Soc 125:138-149.

Cavers, P. B., M. I. Heagy, and R. F. Kokron. 1979. The biology of Canadian weeds. 35. *Alliaria petiolata* (M. Bieb.) Cavara and Grande. Can J Plant Sci 59:217-229.

Cruden, R. W., A. M. McClain, and G. P. Shrivastava. 1996. Pollination and breeding system of *Alliaria petiolata* (Brassicaceae). Bull Torrey Bot Club 123:273-280.

Drayton, B. and R. B. Primack. 1999. Experimental extinction of garlic mustard (*Alliaria petiolata*) populations: implications for weed science and conservation biology. Biol Invasions 1:159-167.

Luken, J. O. and M. Shea. 2000. Repeated prescribed burning at Dinsmore Woods State Nature Preserve (Kentucky, USA): Responses of the understory community. Nat Areas J 20:150-158.

Mackenzie, S. J. B. 1995. Response of garlic mustard (*Alliaria petiolata* (M. Bieb.) Cavara and Grande) seeds and first year plants to cold, heat, and drought. M.S. Thesis, Wright State University.

Meekins, J. F. and B. C. McCarthy. 1999. Competitive ability of *Alliaria petiolata* (garlic mustard, Brassicaceae), an invasive, nonindigenous forest herb. Int J Plant Sci 160:743-752.

---------- 2000. Responses of the biennial forest herb *Alliaria petiolata* to variation in population density, nutrient additions, and light availability. J Ecol 88:447-463.

Appendix 1. Cont.

---------- 2002. Effect of population density on the demography of an invasive plant (*Alliaria petiolata*, Brassicaceae) population in a southeastern Ohio forest. Am Midl Nat 147:256-278.

Myers, C. V. and R. C. Anderson. 2003. Seasonal variation in photosynthetic rates influences success of an invasive plant, garlic mustard (*Alliaria petiolata*). Am Midl Nat 150:231-245.

Myers, C. V., R. C. Anderson, and D. L. Byers. 2005. Influence of shading on the growth and leaf photosynthesis of the invasive non-indigenous plant garlic mustard [*Alliaria petiolata* (M. Bieb) Cavara and Grande] grown under simulated late-winter to mid-spring conditions. J Torrey Bot Soc 132:1-10.

Nuzzo, V. A. 1991. Experimental control of garlic mustard [*Alliaria petiolata* (Bieb.) Cavara and Grande] in northern Illinois using fire, herbicide, and cutting. Nat Areas J 11:158-167.

---------- 1993. Distribution and spread of the invasive biennial *Alliaria petiolata* (garlic mustard) in North America *In:* McKnight, B. N. (ed.). *Biological Pollution: The Control and Impact of Invasive Exotic Species*. pp 137-146. Indiana Academy of Science, Indianapolis.

---------- 1994. Response of garlic mustard (*Alliaria petiolata* Bieb. [Cavara and Grande]) to summer herbicide treatment. Nat Areas J 14:309-310.

---------- 1999. Invasion pattern of the herb garlic mustard (*Alliaria petiolata*) in high quality forests. Biol Invasions 1:169-179.

Peterson, A. T., M. Papes, and D. A. Kluza. 2003. Predicting the potential invasive distribution of four alien plant species in North America. Weed Sci 51:863-868.

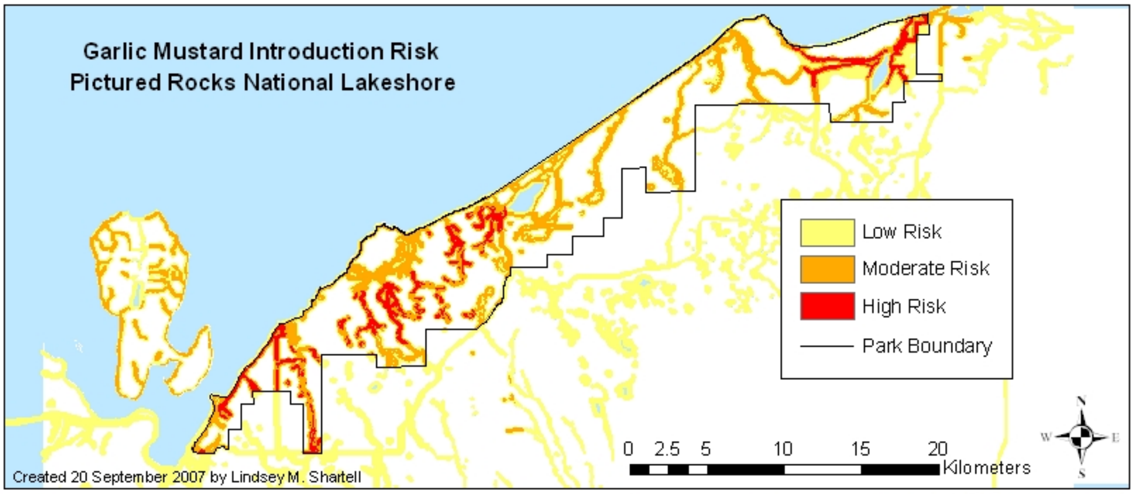
Porter, A. 1994. Implications of introduced garlic mustard (*Alliaria petiolata*) in the habitat of *Pieris virginiensis* (Pieridae). J Lepid Soc 48:171-172.

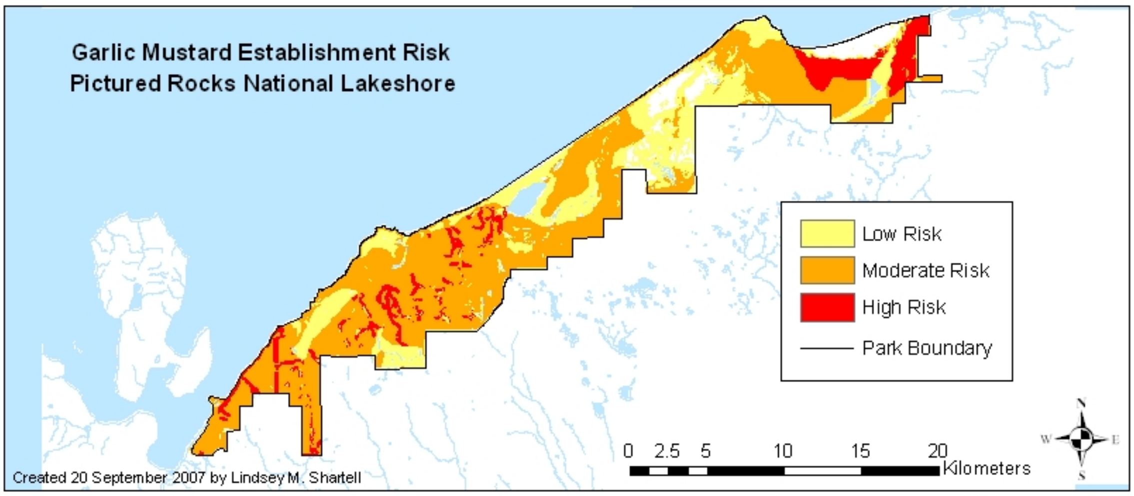
Roberts, H. A. and J. E. Boddrell. 1983. Seed survival and periodicity of seedling emergence in eight species of Cruciferae. Ann Appl Biol 103:301-304.

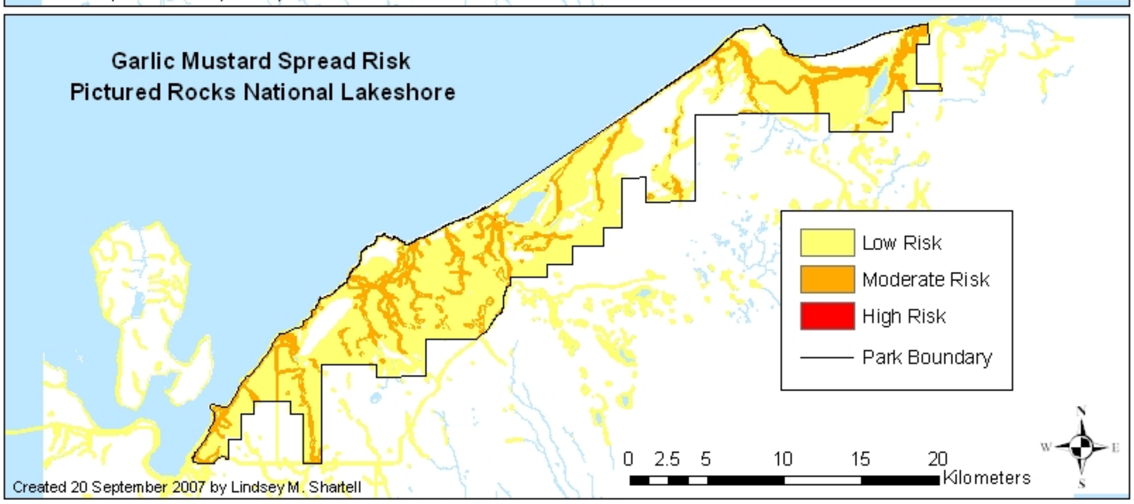
Voss, E. G. 1985. Michigan Flora. Part II Dicots. Cranbrook Institute of Science and University of Michigan Herbarium.

Welk, E., K. Schubert, and M. H. Hoffmann. 2002. Present and potential distribution of invasive garlic mustard (*Alliaria petiolata*) in North America. Divers Distrib 8:219-233.

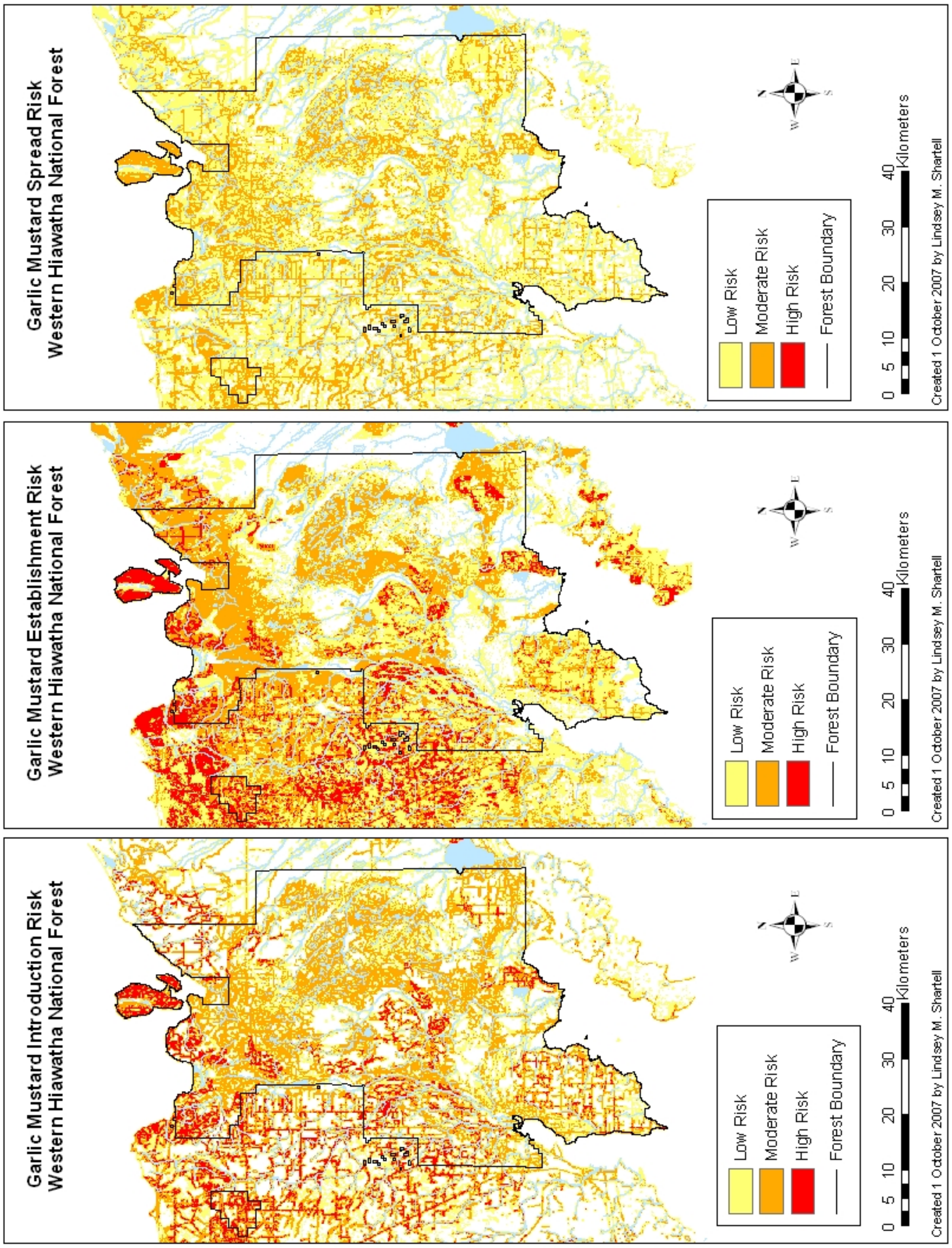
Appendix 2. Garlic mustard risk maps at three phases of invasion for the selected natural areas in the Upper Peninsula of Michigan.



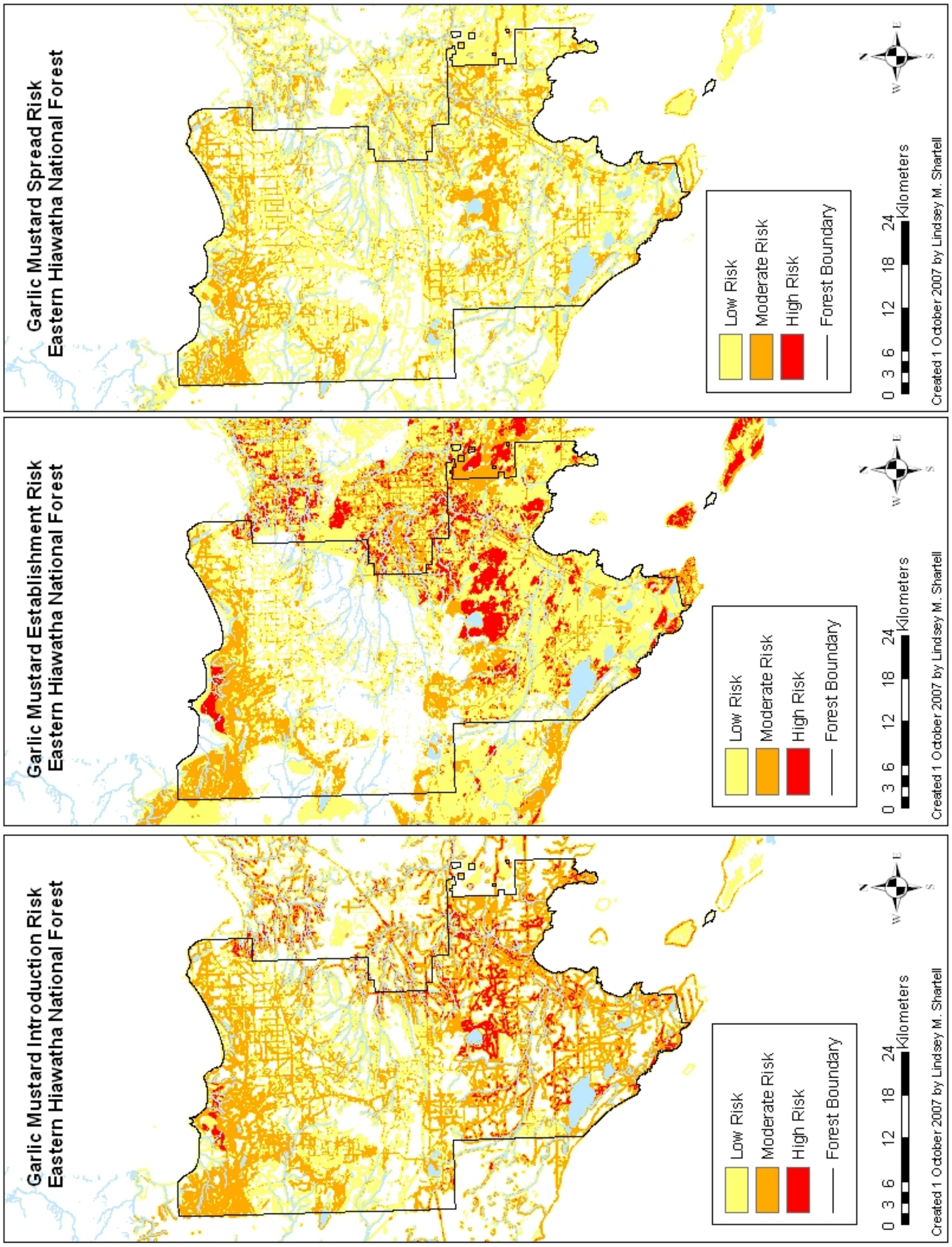




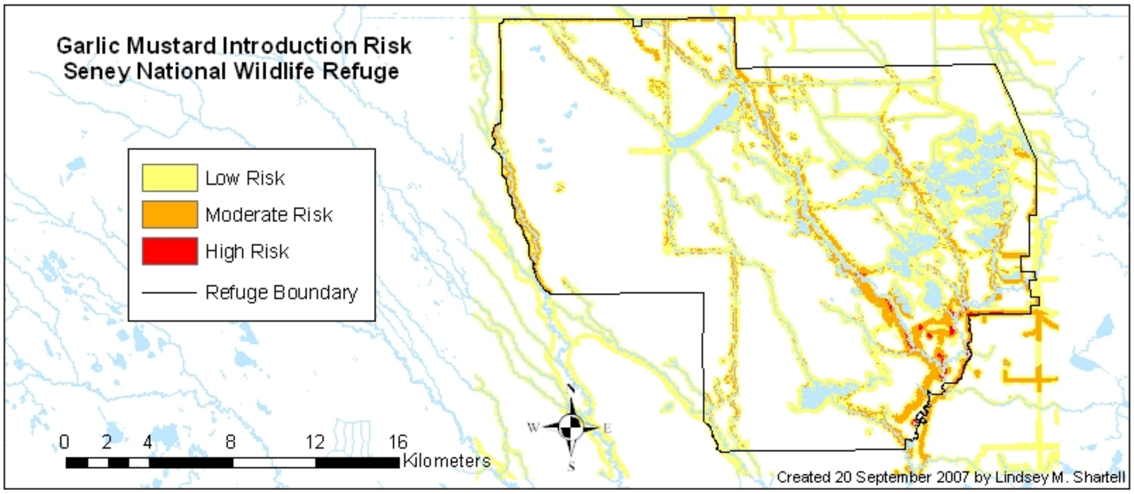
Appendix 2. Cont.

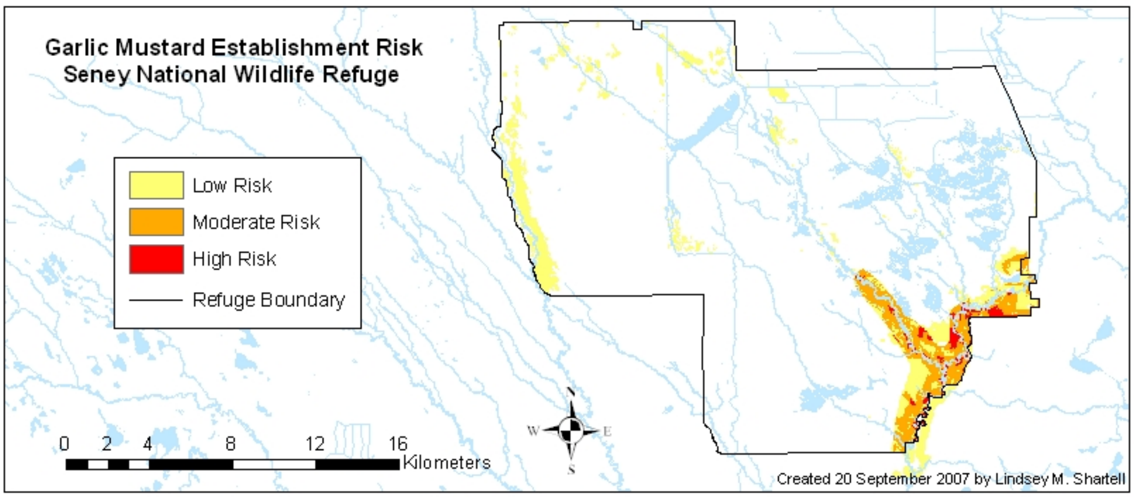


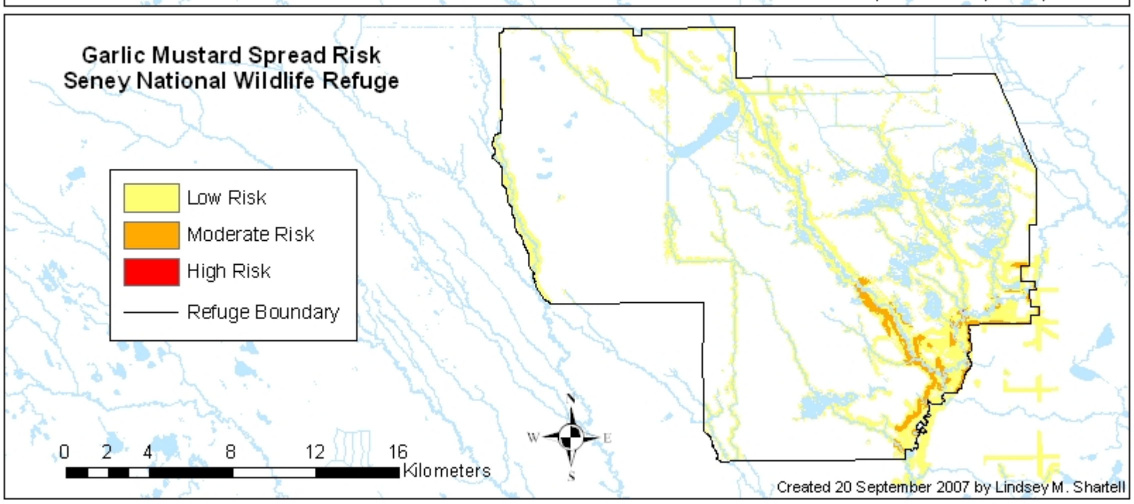
Appendix 2. Cont.



Appendix 2. Cont.







Appendix 2. Cont.