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Developing Useful Spatially Explicit Habitat Models and Decision-Support Tools for Wildlife Management

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Introduction

In this chapter we advocate an applied approach to landscape ecological research and conservation planning through the use of spatial models, where the end goal is informing and guiding on-the-ground conservation actions. We do so to provide guidance for powerful approaches that agencies and organizations often adopt but that are, in our experience, frequently not used to their full potential. We assume throughout this chapter that explicit conservation actions, or treatments, are to be implemented. Our target audience includes conservation planners and managers who are contemplating the use or development of spatially explicit models to guide such actions. A substantial information gap regarding the use of models often exists between scientists and managers (Stauffer 2002, Noon et al. 2009), and we present real-world considerations for planners and managers who may be unsure of how to proceed given the sometimes bewildering array of spatial methods that are available (Elith et al. 2006, Thompson and Millspaugh 2009, Franklin 2013, Phillips et al. 2017).

Landscape ecology is important to wildlife management because location is important to conservation action. Landcover, vegetative communities, land use, landscape configuration, and land ownership vary across space (Forman 1995). Similarly, species distributions, densities, and demographics vary across space (Boyce and McDonald 1999), as do costs and opportunities for conservation action (Naidoo et al. 2006). In this chapter we present concepts and philosophies related to identifying the best place for each conservation treatment and the best conservation treatment for each place. Many of our examples come from migratory bird conservation efforts in the US Prairie Pothole Region (PPR), where spatial models guide annual expenditures of ~$70 million (USA) on acquisition of perpetual wetland and grassland easements held by the US Fish and Wildlife Service (USFWS) and also guide private lands management and programs such as the Farm Bill (Reynolds et al. 2006, Niemuth et al. 2008).

There are good reasons for adopting a landscape approach to conservation, and we assembled a list of the top 10 reasons we have seen advocated for use of spatial models (Table 12.1). Our top reason that agencies and organizations adopt a landscape approach to conservation (i.e., maps are pretty) is tongue-in-cheek, and we use it to underscore that although an attractive map is often the final product

Table 12.1. Top Ten Reasons for Adopting Use of Spatial Models in Wildlife Management

Number	Reason
10	Geographic information systems enable quick and efficient entry and processing of spatial data.
9	Theoretical advances in landscape ecology provide conceptual foundation for models.
8	Role of landscapes in sustaining wildlife populations is increasingly recognized.
7	Advances in statistical analysis enable development of models that accommodate complex data.
6	Increased computing power permits complex analyses.
5	Global positioning systems capacity permits easy acquisition of precise location data.
4	Remotely sensed data is more easily acquired and processed.
3	Many forms of response and predictor data are available for free over the Internet.
2	Programs promote use of spatial tools.
1	Maps are pretty.

of a landscape analysis, the myriad factors that went into producing the map may not be known or understood by some end users.

Maps are tools for conveying information and displaying patterns that would otherwise be difficult to comprehend (Monmonier 1996, Wiens 2002) and as such have strong appeal to managers, administrators, and conservation proponents. It is often difficult, however, if not impossible, to easily assess the quality of information in a map simply by looking at it, and the quality of data or inferences represented by the map may well be poor or misleading (Monmonier 1996, Wiens 2002, Anselin 2006). In addition, choices for displaying maps can be deceptive, as the selection of a value threshold or color palette for display can greatly influence what people perceive or focus on when viewing a map (Monmonier 1996, Krygier and Wood 2016, Morris et al. 2016). Consequently, a map runs the risk of being a shiny bauble that is used to draw attention to a program or effort but fails to provide useful direction for conservation. Worse, a poorly conceived and developed model can misdirect conservation actions to less efficient or effective locations. Our goal in this chapter is to provide guidance that will help users identify and avoid common pitfalls in developing spatial models and ensure that spatial analyses result in useful tools for conservation. The topics we present have repeatedly arisen in discussions with scientists, managers, and administrators as part of the targeting and delivery of conservation in the PPR, but the general concepts have broad applicability.

Definitions

Before proceeding, a few definitions are in order. We define a *model* as an articulation or representation of relationships. We use the terms *landscape model* and *spatial model* in a generic sense to describe models that are developed across broad spatial extents and then applied to geographic information system (GIS) data to create a map depicting the product of those relationships. These maps may depict a variety of biological responses, including species occurrence, density, survival, and reproductive success, as well as nonbiological responses.

Decision-support tools are derived from spatial models by integrating the spatial model with specific information about planned conservation actions and are used to determine the amount, type, or location of conservation treatments. Unlike spatial models, which are generally time consuming and expensive to develop, decision-support tools may be nothing more than a simple reclassification of a spatial model or models using treatment-specific thresholds. For example, the waterfowl thunderstorm map, so named because of its resemblance to a Doppler radar image of a thunderstorm crossing the Great Plains (Fig. 12.1A), is a simple decision-support tool used to guide upland conservation treatments for waterfowl. The thunderstorm map summarizes and establishes decision thresholds for output of regression and accessibility models for five species of upland-nesting ducks settling on >2 million wetlands of four water permanence classes over multiple years and states (Reynolds et al. 2006). The

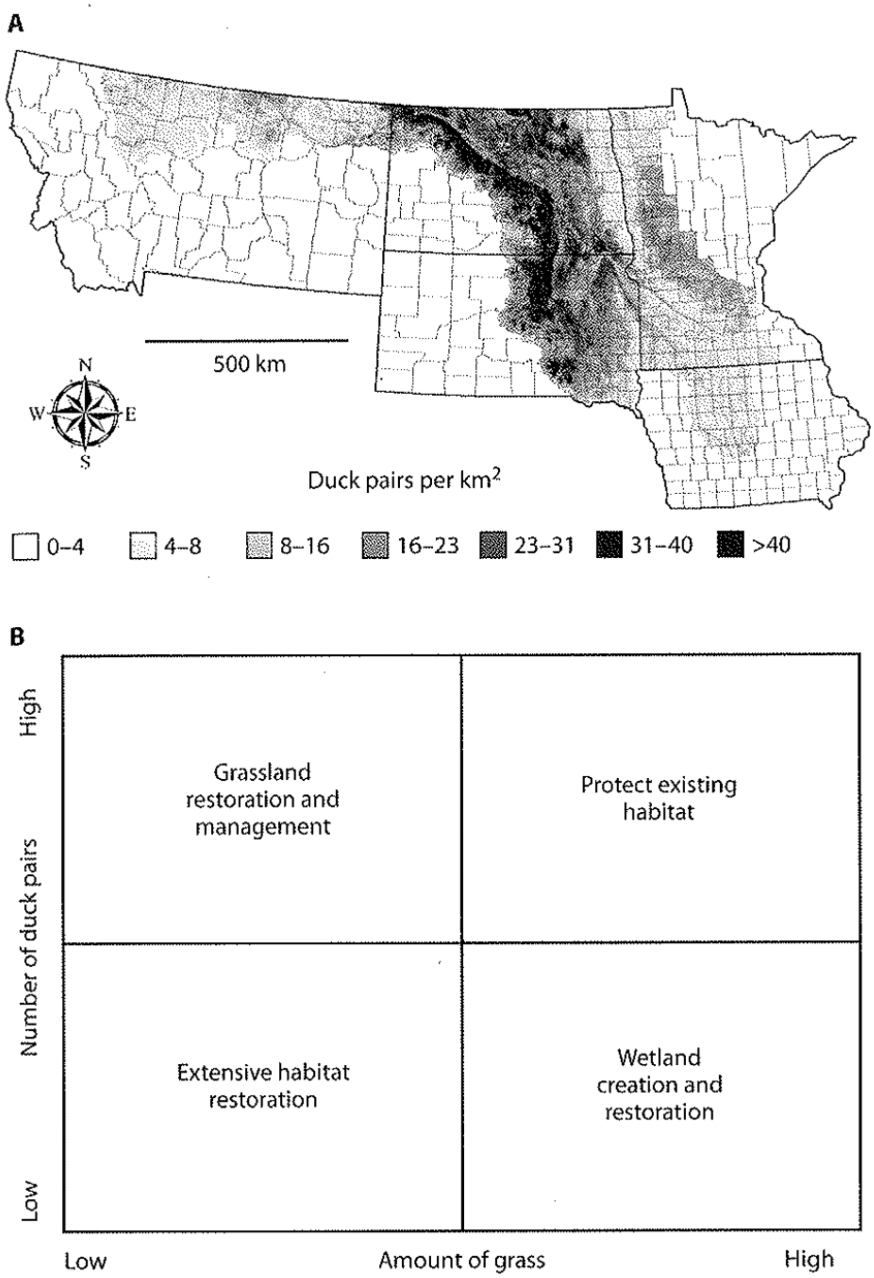

Figure 12.1. The waterfowl thunderstorm map (*A*) models the number of duck pairs with access to upland habitats (after Reynolds et al. 2006) in the Prairie Pothole Region of Iowa, Minnesota, Montana, North Dakota, and South Dakota, USA, 1987-2012. Used in conjunction with landcover data, a simple decision matrix (*B*) can be created to guide conservation actions. Thresholds for amounts of grass and number of duck pairs are determined by biology, funding, conservation goals, and availability of willing landowners.

thunderstorm map is the foundation of many upland conservation treatments in the region; when used in conjunction with landcover data, a simple decision matrix can be created to guide conservation actions (Fig. 12.1*B*). Options associated with this matrix are explored in "Identifying the Purpose," below.

Identifying the Purpose

Many spatial models have an unstated goal of identifying the best areas for conservation, but what these areas are best for is rarely specified. Land protection is often an implicit conservation treatment, but protection of existing habitat is not always an option and may be insufficient to conserve species whose habitat is limiting. Spatial models and decision-support tools will be more valuable to conservation if they are developed and applied in the context of specific, clearly articulated needs that explicitly consider intended conservation treatments, funding constraints, and species needs. For example, temperate grasslands are one of the most altered biomes on the planet, have the lowest rate of habitat protection of all major biomes (Hoekstra et al. 2005), and grassland loss in the PPR continues at a substantial rate (Stephens et al. 2008, Rashford et al. 2011, Lark et al. 2015). Grassland loss has major implications for waterfowl, as the majority of waterfowl in the PPR nest in grasslands (Batt et al. 1989). Grassland loss in the region also has major implications for grassland birds, which have their highest diversity in northern plains and have a larger proportion of species that are decreasing than any other bird group in North America (Askins 1993, Peterjohn and Sauer 1999). Consequently, grassland conservation is a focus of many conservation programs in the region, with the goal of providing benefits for upland-nesting waterfowl and many other species of grassland-dependent wildlife.

Given the size of the US PPR, which extends 1,800 km from northwestern Montana to central Iowa, and uneven distribution of remaining grasslands (Fig. 12.2A), spatial decision-support tools are especially useful for guiding conservation actions.

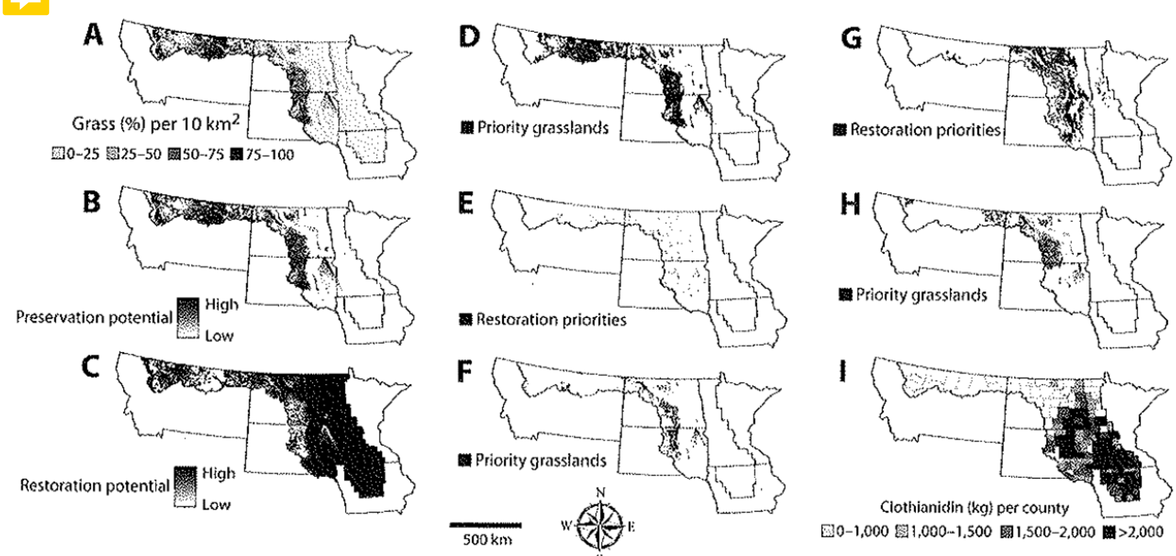

Figure 12.2. The amount of grassland in the Prairie Pothole Region (PPR) of Iowa, Minnesota, Montana, North Dakota, and South Dakota, USA, is generally lowest in the east and higher in the west (*A*), but Priority Areas for Conservation treatments differ greatly depending on intended treatments. Potential for grassland protection is highest where grass remains (*B*); potential for grassland restoration is highest where grass has been converted (*C*); areas with ≥35% grassland are located primarily in the western PPR (*D*); opportunities to restore grass to create areas with ≥35% grassland are more broadly distributed in the region (*E*); areas of existing grass with access to ≥16 pairs of upland-nesting ducks per square kilometer are located primarily in the western portion of the PPR in North Dakota and South Dakota (*F*); areas that are not grass but have access to ≥16 pairs of upland-nesting ducks per square kilometer are located primarily in the central and eastern portion of the PPR in North Dakota and South Dakota (*G*); areas of existing grass within the range of LeConte's sparrow (*Ammospiza leconteii*) are located primarily in the north-central PPR (*H*); and county-level application rates of clothianidin in 2012 are highest in the southeastern portion of the region, which will influence treatment options related to grasslands and insects (*I*). Landcover data from the National Land Cover Database (Homer et al. 2015); clothianidin data from Baker and Stone (2015).

The location of priority grasslands, and therefore the location of conservation actions, will be highly dependent on the specific conservation goal and other factors. For example, areas with highest potential for grassland preservation are where grass presently exists (Fig. 12.2*B*). But many funding opportunities and conservation programs focus on habitat restoration, and areas with highest potential for grassland restoration are where the least grass exists (Fig. 12.2*C*). The selection of priority areas becomes far more complex, however, when waterfowl benefits, land use, cost, risk, benefits to other species, and other factors are considered.

For example, waterfowl nesting success increases with the amount of grassland in the landscape (Greenwood et al. 1995, Reynolds et al. 2001, Stephens et al. 2005), and approximately 35% grass in a 10-km^2 landscape should, in general, provide a nesting success of 15%, which is required for population maintenance (Cowardin et al. 1985, Klett et al. 1988, Reynolds et al. 2001). Therefore grassland protection in areas with >35% grass (Fig. 12.2*D*) may be a waterfowl conservation priority. If a 35% grassland threshold is also a priority for grassland restoration, conservationists may want to target areas with 29% grass in a 10-km^2 landscape (Fig. 12.2*E*), as addition of 65 ha of grass, which is a common size for land ownership and management in the area, will bring the surrounding landscape over the 35% threshold. The previous two examples demonstrate how different treatments (protection versus restoration) substantially change the location of priority areas when considering the same biological relationship between waterfowl nest success and amount of grass in the surrounding landscape. Similarly, grassland protection and restoration may be targeted to include areas with potential to provide nesting habitat for high numbers of dabbling ducks (Figs. 12.2*F* and 12.2*G*, respectively), changing the distribution yet again.

Priorities might change further if non-waterfowl species are considered; grassland protection targeted to benefit species of interest with a limited distribution, in this case Le Conte's sparrow (*Ammodramus leconteii*), shifts priority grasslands to the north-central portion of the PPR (Fig. 12.2*H*). Grassland-dependent pollinators might be another consideration for planning grassland protection or restoration, as populations of honeybees, native bees, moths, and prairie butterflies are declining (Potts et al. 2010, Farhat et al. 2014, Koh et al. 2016). Neonicotinoid insecticides have been implicated in the decline of these species (Godfray et al. 2014, Pecenka and Lundgren 2015), and a potential response would be to avoid butterfly conservation efforts in areas of high neonicotinoid use, which, in the PPR, would shift conservation efforts west (Fig. 12.2*I*). But this action would move conservation actions away from areas with the highest densities of some pollinator species, which are found in the southeastern portion of the PPR (Wassenaar and Hobson 1998).

All of the previous examples of factors influencing the purpose and application of a decision-support tool focused on biology, primarily species distributions and demographics, but socioeconomic considerations also can be key factors in determining where to do conservation and how to do it in a cost-effective manner (Haight and Gobster 2009). For example, the mean price per hectare of farmland in 2015 across the PPR ranged from a high of $19,770 in Iowa to a low of $2,200 in Montana, with intermediate values of $4,740 in North Dakota, $5,730 in South Dakota, and $11,610 in Minnesota (National Agricultural Statistics Services 2015). Additional factors, including soil productivity and risk of grassland conversion, are important when prioritizing locations for action (Olimb and Robinson 2019), and in a region dominated by private lands, acceptance by landowners is necessary (Fields 2017). Factors such as these will vary among regions, but the key point is that the results of spatial planning tools and the areas that are prioritized are dependent on the intended conservation treatments, and using the wrong criteria or no criteria may diminish the value of conservation efforts (Abrahms et al. 2017, Fields 2017). Even if the correct criteria are used, the manner in which they are applied must be carefully considered, as questionable math, hidden value judgments, and arbitrary scores can greatly influence results (Game et al. 2013).

Theoretical and Conceptual Foundations of Models

A thorough understanding of the pertinent mechanisms and issues—whether biological, social, or economic—that affect the system under consideration is necessary to ensure the relevance and effectiveness of spatial tools. For example, the PPR contains millions of wetlands that range in size and water permanence from ephemeral wetlands that cover tens of square meters to lakes that cover hundreds of square kilometers (Kantrud et al. 1989). The PPR has substantial conservation programs because of the importance of the region to wetland-dependent migratory birds (North American Waterfowl Management Plan Committee 1986, Beyersbergen et al. 2004, Niemuth et al. 2008) and extensive and ongoing loss of wetlands (Dahl 1990, Oslund et al. 2010). Island biogeographic theory (MacArthur and Wilson 1967) is frequently advocated as a framework for targeting conservation of prairie wetlands (Whited et al. 2000, Bertassello et al. 2018). Occurrence of many wetland-dependent species is indeed higher on large wetlands relative to small wetlands (Kantrud and Stewart 1984, Naugle et al. 2001, Niemuth et al. 2006), and connectivity is important to some ecological processes in pothole wetlands (Galatowitsch and van der Valk 1996). But island biogeography, though intellectually appealing and appropriate in some situations, may be a poor choice for guiding conservation of wetland-dependent migratory birds in the PPR for several reasons.

Risk of loss is greater on small wetlands because, in addition to being smaller and shallower than large wetlands, they also are generally higher in the local topographic neighborhood and therefore easier to drain or consolidate than large wetlands (Anteau 2012). Small wetlands have higher densities of breeding dabbling ducks than large wetlands because territorial behavior limits the number of individuals of a species on a wetland. Therefore, ten 1-ha wetlands will have more pairs of breeding dabbling ducks than one 10-ha wetland (Cowardin et al. 1995).

Many ecological processes are influenced by wetland size and depth, with the result that small wetlands provide better habitat for many species of breeding migratory birds, particularly waterfowl. Small wetlands are extremely productive because their shallow waters warm up early in spring, and their dynamic nature facilitates nutrient cycling and regeneration of vegetation (Harris and Marshall 1963, van der Valk and Davis 1978, Murkin et al. 1997). Small wetlands are more likely to support emergent vegetation communities that are absent from deeper wetlands (Kantrud et al. 1989). In addition, small wetlands often lack minnows, which forage on invertebrates and reduce invertebrate numbers, leading to reduced growth rates and survival of ducklings (Bouffard and Hanson 1997, Cox et al. 1998). Finally, proximity to other wetlands is not important for migratory birds to colonize a wetland, although wetland complexes may be important for providing foraging and brood-rearing opportunities throughout the breeding season. For all these reasons, a decision-support tool prioritizing large wetlands close to each other would be less efficient than tools based on pertinent issues and biology of the species under consideration, especially for conservation of upland-nesting dabbling ducks.

As always, conservation needs, purpose, and treatments must be considered. Water chemistry also varies with wetland size, as large, terminal wetlands generally have higher concentrations of salts and minerals than small wetlands (Kantrud et al. 1989). Resulting brackish water benefits some species, particularly shorebirds such as piping plover (*Charadrius melodus*) and American avocet (*Recurvirostra americana*), which breed on large or saline wetlands (Kantrud and Stewart 1984). For those species, large wetlands would be appropriate for conservation, assuming there was a threat to the wetlands, and the value of proximity to other wetlands would vary among species. Terrestrial applications of island biogeographic theory might be appropriate when considering resident species with limited mobility but would likely have unintended consequences and misdirect conservation actions for dabbling ducks given characteristics of PPR wetlands and many species of birds that use those wetlands.

Similarly, choice of response metric greatly affects how and where conservation will be implemented. Many spatial models focus on species richness, but unless the species and treatments are explicitly and well considered, richness metrics can have unintended consequences. First, the scale of species range or niche overlap and conservation treatment must be appropriate, as some species show congruent range at a broad scale but can actually be negatively correlated with each other at finer scales (Sherry and Holmes 1988), which is typically the scale at which management actions occur. Species richness models are often coarse grained, generally developed from occurrence data aggregated to large spatial units such as counties or summaries from broad surveys (Stohlgren et al. 2006, Distler et al. 2015). For these and other reasons, including spatially varying densities and annual rates of population change, overlays of species richness may be resistant to scaling up and scaling down (Conroy and Noon 1996). In addition, poorly considered richness measures often ignore rare species and promote generalists or weeds (Noss 1987, Rodda 1993) rather than the rarer and often declining specialists that are typically species of concern. Sites can have the same number of species but different species composition, which might require different conservation treatments or scales of treatment (Fischer et al. 2004). Species richness models can be sensitive to the species considered and their spatial distribution, which can cause a shift in the location of priorities (Van Horne 2002). Finally, areas of highest richness may not harbor every species under consideration; consequently, species not in the high-richness zones might not receive and benefit from conservation actions.

Targeting species richness can be problematic from the standpoint of implementing treatments. At a conceptual level, priority areas include the zone of overlap for multiple species, which may exclude the best areas for the individual species under consideration (Fig. 12.3). This restricted zone of geographic overlap may also limit opportunities for implementing conservation treatments, which may be particularly important in landscapes dominated by private lands, where conservation action only occurs with willing landowners. If species richness is estimated without consideration of species' life histories, limiting factors, and habitat requirements, conservation treatments intended to benefit one species may reduce habitat quality for other species. For example, bird species richness in the northern Great Plains is higher in areas with trees than in nearby grasslands (Dieni and Jones 2002, Van Horne 2002), but preservation or expansion of high richness sites containing forest species would be deleterious to grassland birds, which are declining more broadly and rapidly

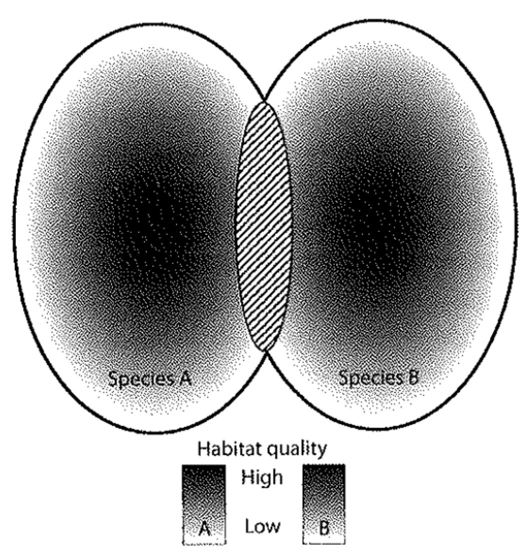

Figure 12.3. The zone of geographic overlap in the range of two hypothetical species excludes the best areas for the species under consideration and limits the area in which conservation treatments can be applied.

than any other bird group in North America (Knopf 1994, Peterjohn and Sauer 1999). One species richness decision-support tool that we reviewed included vertebrate, invertebrate, terrestrial, aquatic, rare, abundant, migratory, resident, generalist, specialist, breeding, nonbreeding, predator, and prey animals, as well as plants, in a variety of ecosystems under varying levels of human disturbance. Identifying a conservation treatment to improve the status of all species included in that model would be nearly impossible, with the possible exception, as a reviewer wryly noted, of broadscale human exclusion, as demonstrated by the response of wildlife in the Korean demilitarized zone (Kim 1997) and the Chernobyl exclusion zone (Deryabina et al. 2015).

Many of the shortcomings of species richness approaches can be avoided by targeting overlap of best areas rather than simple occurrence, or by addressing priority species individually or in small aggregates. If that is not possible, shortcomings can be circumvented by careful consideration of species biology and intended treatments and focusing on benefits for few similar species rather than richness of many disparate species (Fleishman et al. 2006). For example, many species of birds are area sensitive, where their occurrence, density, survival, or reproductive success is greater in large habitat patches than small habitat patches (Fig. 12.4*A*; Robbins 1979, Ribic et al. 2009). By meeting the requirements of the most area-sensitive species under consideration, requirements of less area-sensitive species also will be met, assuming that appropriate fine-grained features are also present (Fig. 12.4*B*). This approach uses the most area-sensitive species as a surrogate or umbrella for the less area-sensitive species; however, it has the consequence that smaller patches important to less area-sensitive species are dismissed, which limits management options and may prevent attainment of population objectives for those species.

The use of surrogate species for prioritizing areas for conservation poses similar problems in that biological outcomes in response to conservation practices are often unknown and may differ from what was intended (Carlisle et al. 2018). As mentioned above, protection of grassland and wetland complexes for waterfowl is a common conservation treatment in the PPR that has provided substantial benefits for other species (Niemuth et al. 2018). Waterfowl may be reasonable surrogates for those species of grassland birds in the PPR that respond positively to small wetlands with emergent vegetation; however, waterfowl will be poor surrogates for those species that prefer xeric sites and have different spatial distributions (Niemuth et al. 2017). The use of surrogates might be justified by a commitment to investigate assumptions of the approach, but by the time sufficient information is gathered to determine how well the surrogate tracks the target species, one could just as well work directly with the target species (i.e., if adequate information exists to justify the use of surrogates, then there is likely no need to use surrogates). If surrogate species are used, it is essential to explicitly state the surrogate approach being used and acknowledge that the use of surrogate species is a shortcut that will fail at some level, as species differ in distribution, ecological niches, and limiting factors (Che-Castaldo and Neel 2012, AMEC Environment and Infrastructure 2014, Carlisle et al. 2018). The bottom line is that decision-support tools based on richness or surrogate species will likely lack explicit, clearly articulated relationships that can be used to identify and guide specific conservation treatments to benefit recognized priority species.

Data Characteristics and Quality

Statisticians tend to think of data as information from which one can make inferences. With the advent of computers, the word "data" has taken on a more encompassing definition along the lines of information that is stored, regardless of its quality or ability to support inferences. Confusing the two meanings is easy in landscape-level investigations where large amounts of data (sometimes of the second meaning) are amassed, processed, and analyzed

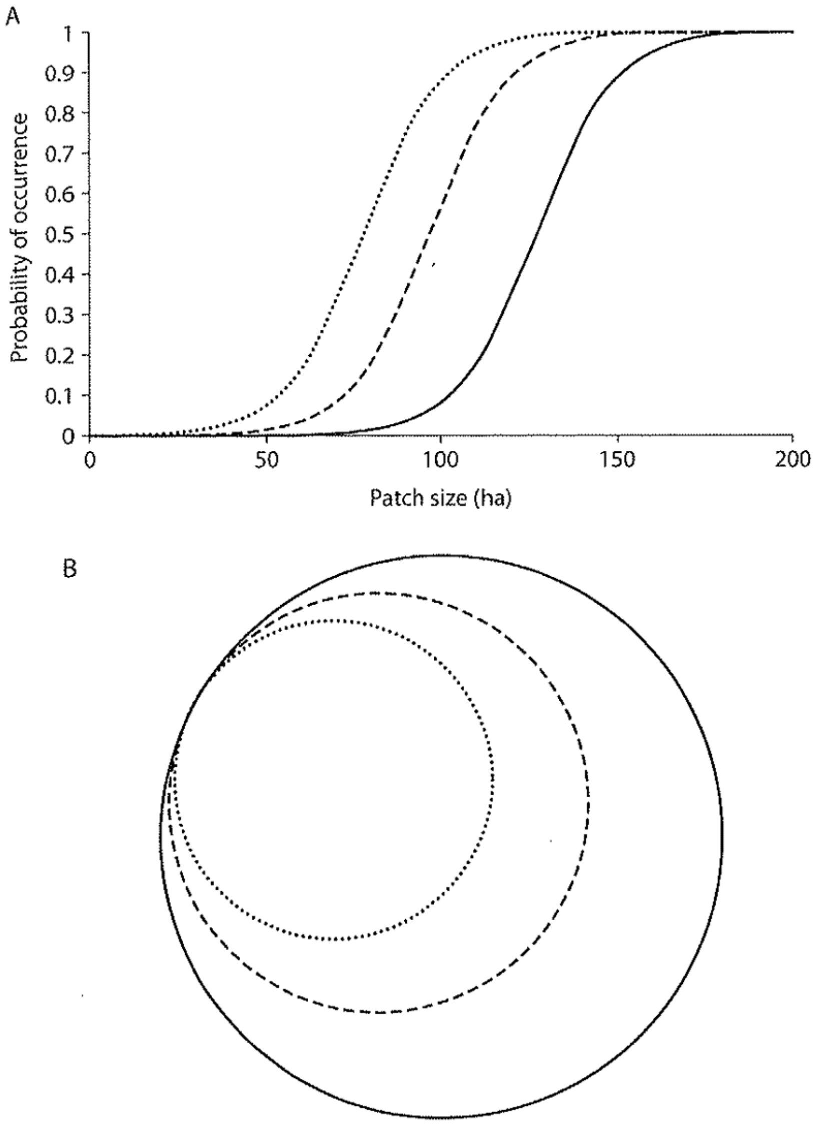

Figure 12.4. Occurrence of three hypothetical species (*A*) is positively related to patch size and varies among species. When species are properly selected, occurrence can be nested, rather than overlapping (*B*), as in Figure 12.3. Benefits for multiple species can be provided by protecting or restoring a habitat patch that is sufficiently large to harbor the most area-sensitive of the three species (solid line), thereby also benefitting less area-sensitive species. Dotted and dashed lines reflect patterns of area sensitivity shown in (*A*).

to create an end product, typically a map that is used to guide decisions. The inability of some of these data to support inferences, however, may be masked by poor or limited understanding of the study region, ecological processes, data, analyses, and the final map products.

Above all, model developers should be familiar with the landscape, the species of interest, and pertinent issues and programs. Spatial analyses rely heavily on remotely sensed data, and in many cases, people conducting spatial analyses are remote from the study region and lack knowledge to catch problems with data. An example of this is a commonly used database that shows protected areas in the United States. The largest polygon identified as a protected area in North Dakota (Fig. 12.5) was classified as fee-title land held by the USFWS but is actually an administrative boundary for an area dominated by private land. This area is almost the size of Yellowstone National Park and should jump out at the most rudimentary level of data screening. Similarly, Native American tribal lands are depicted as encompassing

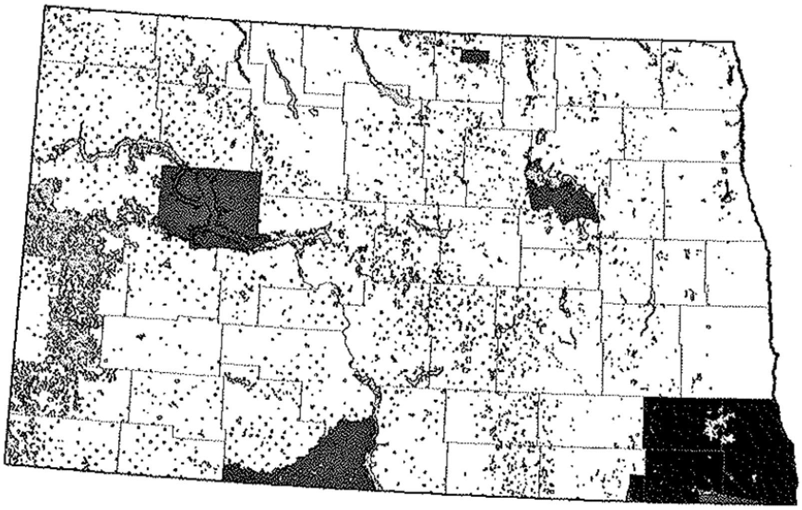

Figure 12.5. Protected lands (shaded areas) in North Dakota, USA, as identified by a commonly used database of protected areas in the United States. The black polygon in the southeastern portion of the state is listed as fee title land held by the US Fish and Wildlife Service (USFWS) but is actually an administrative boundary for a private lands easement acquisition program and represents no USFWS fee title lands. Medium-shaded polygons are identified as tribal lands but represent boundaries within which >50% of land is privately owned.

substantial portions of North Dakota (Fig. 12.5), but these polygons are again administrative boundaries, and >50% of land within these polygons is privately owned (Bureau of Indian Affairs 2019). Estimates of contributions of conservation efforts developed with these data will be inflated, and assessments of biodiversity of protected areas relative to unprotected areas could be biased. To the credit of the database compilers, erroneous ownership in the largest polygon (Fig. 12.5) has been corrected in recent iterations, but not before results of analyses using the erroneous data were published in at least one peer-reviewed journal (Wood et al. 2014).

Conservation practitioners should use data at resolutions and scales that are appropriate to the species, question, and treatment being considered (Wiens 1989, Boyce 2006, Mayor et al. 2009). For example, element occurrence analyses note records of a species' recorded occurrence (and, conversely, non-observation) in counties, watersheds, map quadrangles, major land resource areas, hexagons, or other coarse-grained units. The coarse resolution of many reporting units, which may be tens to thousands of times the size of intended treatments, precludes meaningful targeting of conservation actions (Fig. 12.6). In addition, element occurrence analyses do not identify or incorporate biological relationships that increase understanding and guide conservation treatments and therefore cannot incorporate appropriate scales for these treatments. Finally, counties, watersheds, major land resource areas, and many other units are variably sized, which contributes to biased estimates of species presence or richness owing to passive sampling (Connor and McCoy 1979), and data used in such analyses are generally opportunistic, often containing biases owing to species and observer location or interest (Anderson 2001, McKelvey et al. 2008, Niemuth et al. 2009).

Characteristics of environmental predictor data are frequently overlooked in spatial analyses, but predictors such as landcover and climate data are themselves the product of models. A primary assumption of regression analyses is that predictors are measured without error, which will almost never

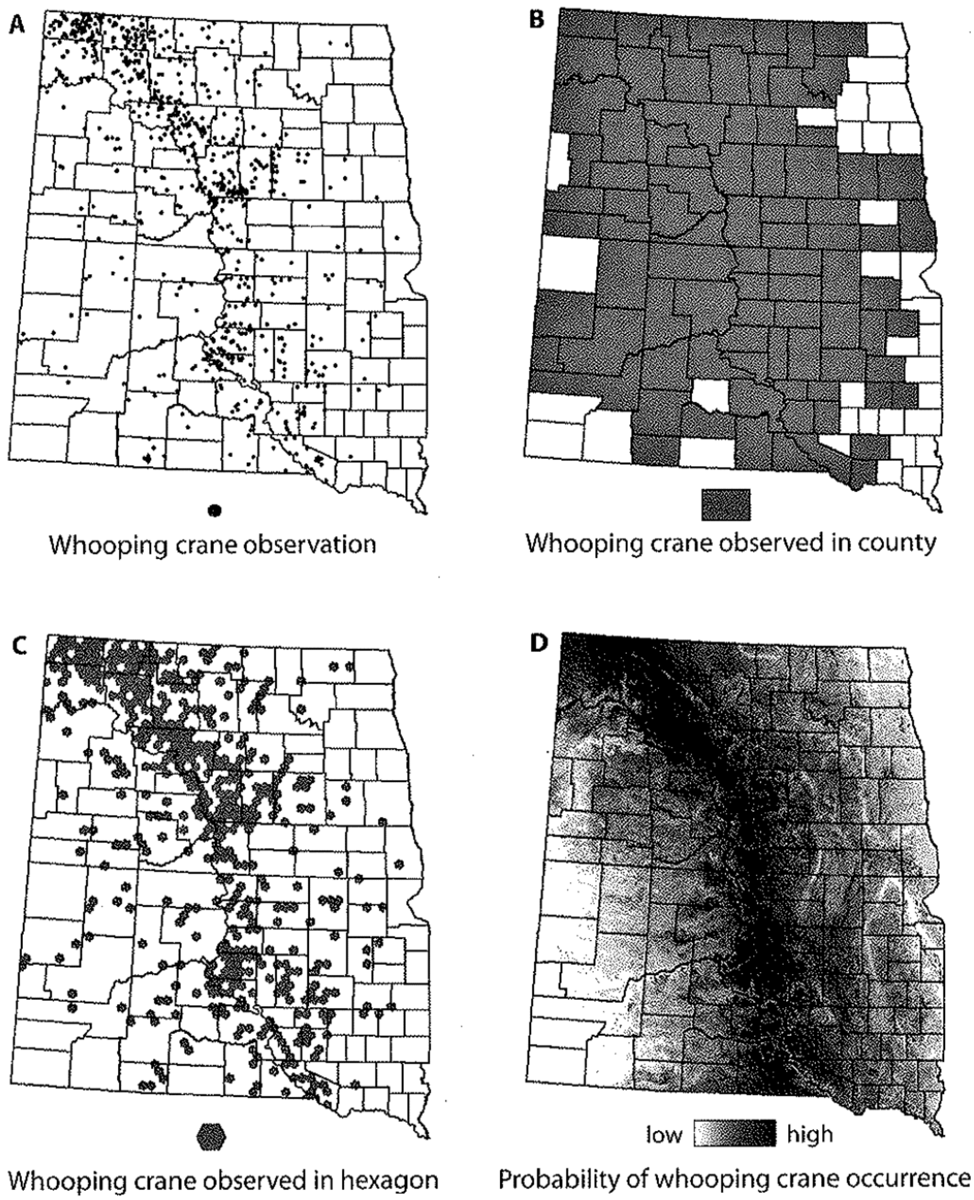

Figure 12.6. Conservation guidance maps produced from a given data set can vary greatly depending on how data are processed and analyzed. Black dots (*A*) indicate observations from the Cooperative Whooping Crane Tracking Project database (Tacha et al. 2010) of whooping cranes (*Grus americana*) in North Dakota and South Dakota, USA, from 1955 to 2014. County-level element occurrence analysis (*B*), following species profile (US Fish and Wildlife Service 2019*b*), includes much non-habitat, identifies no biological relationships, and provides little ability to target specific actions. Element occurrence analysis (*C*) using nested hexagon grid (Western Association of Fish and Wildlife Agencies 2019) greatly reduces area of polygons identified as occupied, but again identifies no biological relationships and excludes much area that is likely habitat. Probability surface (*D*) developed using environmental predictors and generalized linear mixed models (Niemuth et al. 2018) captures biological relationships and enables quantitative ranking and evaluation of potential conservation sites but shares some limitations associated with use of opportunistic data.

be true when using many types of predictor data. Adding to the complexity of using these data is that classification accuracy varies throughout the data set because of differences in timing, availability of cloud-free images, and many other factors (Thogmartin et al. 2004, Gallant 2009). Accuracy of many landcover data sets can be improved through combining of similar classes where appropriate or the addition of ancillary data from other sources. For example, in the case of analyses where perennial cover is important, researchers might incorporate spatial data from the US Department of Agriculture National Agricultural Statistics Service that identify alfalfa fields (Boryan et al. 2011). Classification accuracy of landcover data should always be reported along with other pertinent quirks or characteristics that might affect interpretation and application of the models (Thogmartin et al. 2004, Gallant 2009). Characteristics of other forms of predictor data also should be assessed prior to analysis, and any limitations of the data to answer the question at hand should be acknowledged, paying particular attention to matters of accuracy and scale.

No data are perfect, whether they are used as predictors or response variables. Every analysis should begin with an assessment of the data and the need to account for biases, and data should be appropriate to the question. Data collection and analysis might differ even for the same species if different portions of its life cycle are being addressed. For example, monitoring and conservation of migrant or wintering waterfowl and waterbirds on large impoundments with moist-soil management along the Mississippi River, USA, will be different than monitoring and conservation of breeding waterfowl in small, dispersed prairie wetlands, and programs addressing the first question should not be forced into situations where they are not appropriate.

When acquiring or assessing data, whether it be for response or predictor variables, analysts should ask what questions could responsibly be addressed with the data rather than forcing inappropriate uses. Predictor and response data are readily available for landscape analyses, but many of them suffer from low quality caused by their opportunistic collection, sampling bias, classification error, or limited relevance to the landscape analysis under consideration (McKelvey et al. 2008, Lozier et al. 2009, Niemuth et al. 2009). Use of poor data is sometimes justified by saying the data are the best available information, but researchers should assess data quality prior to analysis, and not use data that cannot support inferences regarding the question of interest. In many cases, it might be preferable to develop a conceptual model that can capture important biological relationships (Clevenger et al. 2002, Johnson et al. 2010), or it might be necessary to collect appropriate data to address a specific question.

If data are collected when developing a spatial model, a sampling frame should be implemented that considers pertinent factors relative to the conservation need. Because landscape analyses cover broad spatial extents and a large range of landcover and environmental variation, it is imperative to consider the question being addressed prior to data collection to ensure that sample size is sufficiently large to cover the range of variation, support consideration of appropriate candidate predictor variables, and reduce extrapolation when models are applied to create maps. Again using the example of area-sensitive species, data should be collected along a continuum of patch size (or amount of habitat in the surrounding landscape) so that the analysis can identify relationships and inflection points. Too often, biologists only want to sample where they know a species occurs, which greatly reduces power to make inferences about habitat selection. Data collected as part of a properly designed study will simplify analysis and provide stronger inferences than data collected with little thought or in a manner that is convenient (Krebs 1989, Anderson 2001).

Analytical Considerations

After a treatment-specific purpose has been identified and quality of data assessed, an appropriate analytic technique must be chosen to identify and quantify the biological relationships that are the core of a useful model. A discussion of analytical techniques is beyond the scope of this chapter, but numerous resources provide in-depth guidance for analysis of spatial data (Shenk and Franklin 2001, Guisan and Thuiller 2005, Elith et al. 2006, Millspaugh and Thompson 2009, Zuur et al. 2009). In fact, the problem for analysis is generally not lack of an available method but the selection of an appropriate one (Jones-Farrand et al. 2011). Johnson (2001:113) wrote, "A model has value if it provides better insight, predictions, or control than would be available without the model." This philosophy leaves a lot of flexibility for model types and development, but Johnson noted that a model's value is determined in light of its intended application, once again reinforcing the idea that addressing explicitly identified conservation needs and treatments is key to success. Choice of analytical technique can substantially affect the final form and value of a spatial model (Wilson et al. 2005, Thompson and Millspaugh 2009), and we present observations on factors that can, in our experience, affect the usefulness of a model.

Models can be classified in many different ways, but for the purposes of this chapter we place models into three general categories: conceptual or expert opinion models, statistical or empirical models, and black boxes. Conceptual or expert opinion models can provide useful guidance based on biological relationships that are identified and related to spatial data (Clevenger et al. 2002, Sanderson et al. 2002, Johnson et al. 2010). These models have the advantage of making use of experience and knowledge of species experts, and results can be almost identical to those of statistical models (Niemuth et al. 2005). Involving experts and other stakeholders in model development can be politically powerful, and conceptual models can provide useful conservation guidance and a better understanding of factors that should be considered if data are collected to develop a statistical model. However, experts on species biology may not understand or agree upon spatial issues and interrelation of data, scale, and conservation treatments, which can complicate development of useful models. Even in the absence of perfect agreement, we believe that conceptual models are superior to the common alternative of drawing focal area boundaries based on the experience or favorite areas of participants. Both approaches use similar methods, but conceptual models force the identification of relationships and characteristics that are important to the question or species of interest (Starfield 1997). These relationships can then be applied to spatial data to identify multiple potential conservation treatments across the entire landscape, whereas drawing focal areas only requires identification of geographic areas, conveys no understanding of mechanisms, and applies only to areas familiar to the experts.

Statistical or empirical approaches have the advantage that relationships are inferred from data, thereby avoiding biases associated with expert opinion. As such, these models can provide insights that might be new or contradict conventional wisdom. Because of their reliance on data, statistical models are generally considered more rigorous than conceptual models and have the additional advantage that estimates of uncertainty are provided for parameter estimates. Statistical models often require substantial amounts of data, which can be expensive and time consuming to collect. Considerable training and technical expertise may be required for proper development and application of some statistical models, particularly with complex data sets often associated with spatial analyses. Consequently, statistics can override ecological relationships (Austin 2002), which can affect interpretation and application of models with respect to conservation treatments. For these reasons, it is imperative that biologists and field staff be involved with the model development process and that statisticians work with an appropriate set of a priori models (Burnham and Anderson 2002, Millspaugh et al. 2009, Noon et al. 2009).

Our distinction between statistical models and black boxes relates more to the process than the mechanisms of the analysis; in fact, black boxes may well be based on sound statistical procedures, but the inputs, assumptions, and process are often unclear, even though they might greatly affect model output (Merow et al. 2013). Black box approaches may include occurrence or richness data sets that obviate the need for analysis (US Fish and Wildlife Service 2019*a*, Western Association of Fish and Wildlife Agencies 2019) and software and websites that offer simple means of developing spatial models (NatureServe 2019), sometimes at the expense of process transparency. In many of these cases the process, including scale and analytical framework, may be determined by the software rather than need, purpose, or biology, with little or no link to explicit conservation treatments. In other cases, users may be able to subjectively weight different components of the analysis or rate data quality, with no understanding of how those values are used or affect the outcome, all of which reduce transparency and interpretability. We do not argue that analyses conducted within black boxes are wrong but suggest that users or administrators might be swayed by ease of use rather than go through the effort required to obtain answers pertinent to the question or treatment of interest. Most important, black box approaches can take the thought out of what should be a thought-intensive process (Starfield 1997).

No matter what type of analytical technique is used, the limitations of process, data, and models should be explicitly acknowledged, as should the importance of people in the field who use the models. Data and models are not reality, and spatial models provide only a view of the world at a fairly coarse scale. Wildlife require habitat at fine scales as well as coarse, and field staff are essential to ensure that all elements of habitat requirements are met and provide feedback about models based on their knowledge of treatments and local conditions.

Programmatic and Staffing Considerations

Finally, even the most clearly articulated and demonstrated conservation needs will go unmet if a project to develop spatial tools lacks programmatic support and qualified staff. Conservation practitioners may have to adopt mindsets and approaches differing from their previous training and experience, and educate administrators as to why new approaches or data are necessary. Researchers should consider management issues and constraints, which may entail citing or publishing in the management literature or planning documents such as joint venture implementation plans (Fields 2017, Connelly and Conway, Chap. 10, this volume). Researchers generally prefer to publish in high-impact journals that are often theoretical or conceptual in nature (Aarssen et al. 2008); however, gray literature may contain specific information that is pertinent to conservation actions that is not available in the mainstream scientific literature (Corlett 2011). Decision makers should rely on science, decision support tools, and field staff to help make decisions and focus on the end goal of conservation rather than program development. The most effective way to foster development and implementation of useful spatial tools will be to identify a need, then develop useful tools that meet the need. People will seek out and use those tools that aid them in making decisions, just as consumers in a free market economy purchase products that meet their needs, whereas products in a command or centrally planned economy often languish because they do not meet a need.

The purpose of a spatial decision-support tool is to increase the efficiency of conservation by providing insights and helping guide decisions over spatial extents that exceed the knowledge of local experts (Guisan and Thuiller 2005, Franklin 2013). Models can increase the cost-effectiveness of conservation (Haight and Gobster 2009), but if there is no conservation action taking place or if the amount of action is minimal, the time and expense associated with developing a spatial tool may exceed the benefits received from the tool, potentially decreasing the amount of conservation taking place. Too often, people want to do monitoring or research, both of which are expensive in terms of time and money, without considering the return on investment that the final product should provide or ensuring that the data will be useful (Legg and Nagy 2006). Instituting a new program will do little good unless the information gathered informs future decisions.

Creating rigorous, useful spatial models and decision-support tools is much more complicated than simply having someone with GIS skills stack multiple spatial data layers or intersect species observation data with county polygons. The process requires capable people with a diversity of analytical skills, biological background, awareness of conservation issues and treatments, and significant experience with data processing and development of spatial tools. The people and process involved with developing a spatial model are critical to its success. Noss (2003) identified eight standards for conservation planning, each with specific sub-criteria: staff qualifications, choice of conservation target, methodological comprehensiveness and rigor, replicability, analytic rigor, peer review, overall quality of scholarship, and iterative improvement. These are simple concepts, but they are too often cast aside for the expedience of using simple methodology, existing staff, poor data, or the desire for a simple, universal spatial tool. Rather than working toward a single spatial tool that meets all needs, conservation practitioners would be better off developing a conservation toolbox, where accurate data and multiple rigorous spatial models serve as components from which decision-support tools can be developed that are appropriate to the needs and treatments under consideration. Such a toolbox and approach will facilitate the development of "small, simple models that focus relentlessly on the problem to be solved" (Starfield 1997:261).

Administrators must recognize that development of scientifically sound spatial tools over broad spatial and temporal scales requires a significant investment of time and money, which reinforces the necessity of clearly identified needs and treatments. Gains in efficiency and effectiveness that come with spatial decision-support tools must be weighed against the cost required to develop those tools. If little money is being spent on the ground, efficiency gained from development of a spatial model may be insufficient to offset the cost of data collection and/or model development. The time and financial investments necessary to develop spatial tools, along with an absence of in-house expertise, may necessitate collaboration with multiple agencies, organizations, and researchers. In these cases, all partners must be cognizant and supportive of the appropriate spatial and temporal scales and characteristics of the conservation program necessary to succeed.

Good science associated with solid data and rigorous spatial models and decision-support tools can provide transparency and accountability as well as increased efficiency for conservation delivery. But the definition of good science often varies among people depending on their personal values or their organization's position on a topic. Politics, money, and logistics are valid reasons for making decisions but should never be disguised as science. Decision makers also must learn to critically evaluate scientific products, as many spatial products are heavily promoted but lack the explicit purpose, appropriate scale, biological linkages, or data quality necessary to be useful for conservation delivery. Similarly, decision makers must realize that scenarios and hypothetical landscapes used in many publications might have heuristic value, but hypothetical landscapes are of less value to people who must make decisions based on the reality of risk of habitat conversion, land prices, and variability in resources.

Summary

Properly developed and implemented, spatial models and decision-support tools can substantially increase efficiency and cost-effectiveness of on-the-ground conservation delivery, but this does not mean that the output of spatial models should be followed blindly. For all the advantages that a model-based approach provides, a model is simply a useful abstraction that cannot characterize all the dynamics of a system (Fig. 12.7). When local knowledge is available that reliably surpasses the general relationships described in a landscape model, the local knowledge should be used. Model developers should work with biologists and field staff who deliver conservation to develop a strategy of continuous feedback that helps refine the model and improve delivery (US Fish and Wildlife Service 2008). This process might provide one of the greatest benefits of a model-based approach to conservation delivery, as it helps people understand the strengths, needs, and limitations of their conservation efforts.

The value of spatial tools goes beyond delivery. Spatial models can be used to demonstrate the biological benefits of conservation programs, provide spatially explicit population estimates, and evaluate potential stressors, to name a few uses (Reynolds et al. 2006, Niemuth et al. 2018). A rigorous program that uses spatial models to target conservation delivery and demonstrate conservation benefits is also highly attractive to administrators and potential funders; for example, a variant of the waterfowl thunderstorm map (Reynolds et al. 2006) was used to support the acquisition and guide the expenditure of $590,000 (USA) for ruddy duck (*Oxyura jamaicensis*) conservation in North Dakota and South Dakota following the April 2000 Chalk Point oil spill off the coast of Maryland, USA. Similarly, spatial models developed for waterbirds (Niemuth et al. 2009) were used to justify the award of $6,000,000 (USA) for black tern (*Chlidonias niger*) conservation in North Dakota and South Dakota following the April 2010 Deepwater Horizon oil spill in the Gulf of Mexico.

Benefits such as these, however, are unlikely to be realized if spatial models and decision-support tools suffer from lack of purpose, poor resolution, inaccurate data, inappropriate analysis, or are not closely linked to implementation. A model that purports to identify the best places for conservation but does not specify what those places are best for will have little real-world value. Virtually every publication describing a species distribution model contains an obligatory sentence or paragraph indicating that the model can be used to guide conservation, but the reality is that, in our experience, the value of many published models for conservation delivery is limited, as they simply do not have an explicit link to delivery. Development of useful spatial tools requires a substantial investment of time, money, and, most importantly, thought about how the tools will be developed, who the users will be, and how the output will be used to deliver conservation. Many of the problems we have described can easily be avoided if conservation practitioners work in the context of a conservation issue, do their homework, maintain a strong biological foundation, and remember that the end goal of

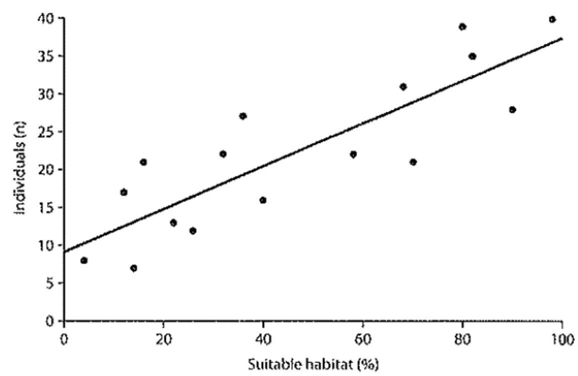

Figure 12.7. Spatial models and decision-support tools will include error, whether from sampling error, unexplained variation in data, or misclassification of landcover data, but the general patterns they portray will be useful for guiding decisions. In this hypothetical example predicting number of individuals as a function of percentage of the landscape in a suitable habitat class, the model (solid line) does not exactly fit any of the observations (black dots) and is substantially wrong for some, but it is useful for characterizing the overall relationship between landscape composition and number of individuals. Similarly, local knowledge that reliably surpasses a statistical model can and should be used when appropriate; if possible, such knowledge should then be incorporated into future analyses to improve the value of the model.

applied landscape ecological research is not program development or publication but conservation.

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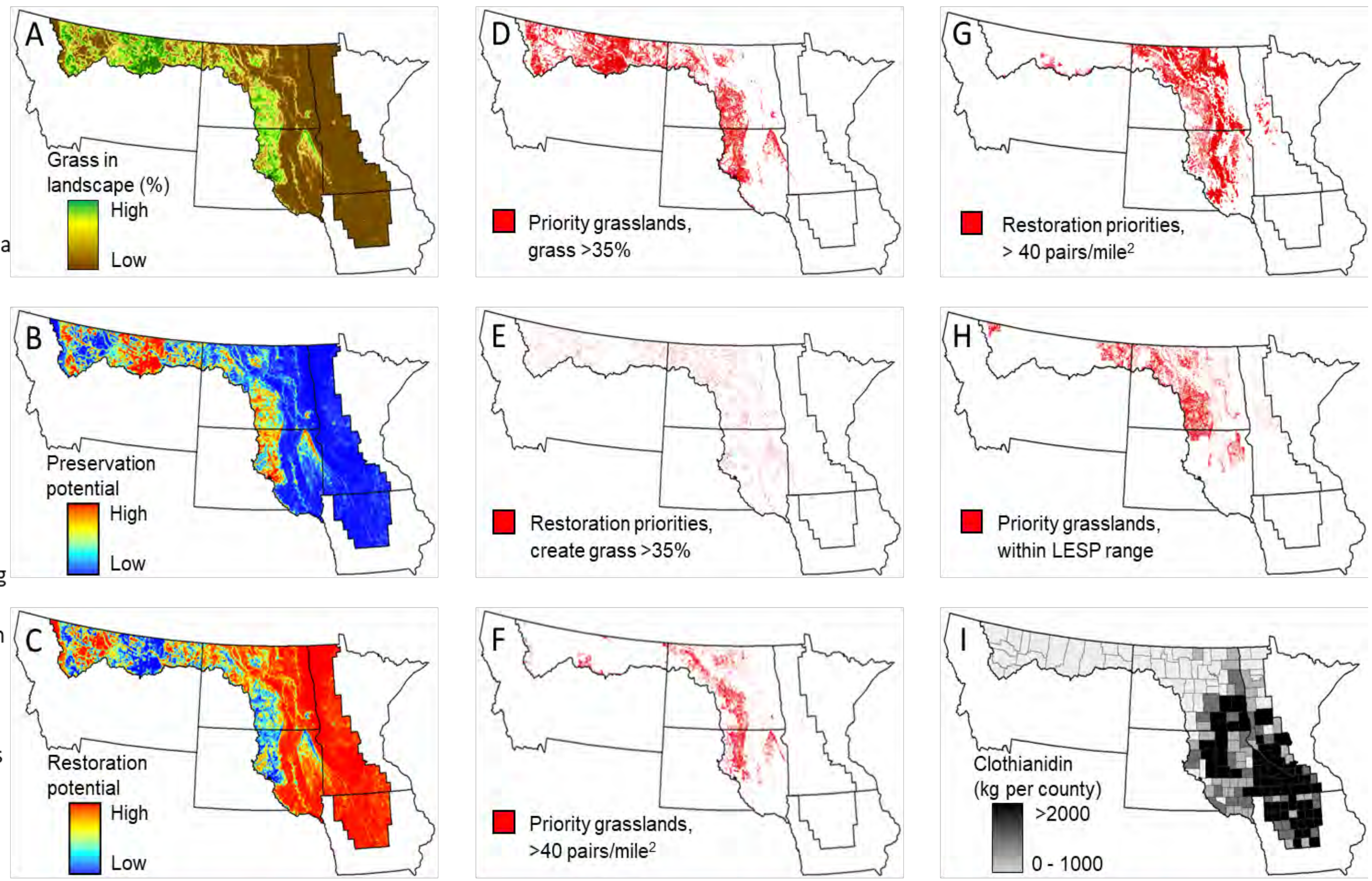
Priority areas can vary greatly depending on the question being addressed. For example, the amount of grassland in the Prairie Pothole Region (PPR) of Iowa, Minnesota, Montana, North Dakota, and South Dakota, Minnesota, and Iowa, USA, is generally lowest in the east and higher in the west (A), but priority areas for conservation differ greatly depending on intended treatments. Potential for grassland protection is highest where grass remains (B); potential for grassland restoration is highest where grass has been converted (C); areas with >35% grassland are located primarily in the western PPR (D); opportunities to restore a quarter-section of grass to create areas with >35% grassland are more broadly distributed in the region (E); areas of existing grass with access to >16 pairs of upland-nesting ducks per km^2 are located primarily in the western portion of the PPR in North Dakota and South Dakota (F); areas that are not grass but have access to >16 pairs of upland-nesting ducks per km^2 are located primarily in the central and eastern portion of the PPR in North Dakota and South Dakota (G); areas of existing grass within the range of LeConte's sparrow (*Ammospiza leconteii*) are located primarily in the north-central PPR (H); and county-level application rates of clothianidin in 2012 are highest in the southeastern portion of the region, which will influence treatment options related to grasslands and insects (I). Landcover data from the National Land Cover Database (Homer et al. 2015); clothianidin data from Baker and Stone (2015).

Color version of Figure 12.2



None of these examples consider cost or risk, which further complicate decisions.