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Toward an Ecological Framework for Assessing Risk to Vertebrate Populations from Brine and Petroleum Spills at Exploration and Production Sites

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ABSTRACT: Brine and petroleum spills may affect terrestrial vertebrates through loss of reproductive habitat or reduced food availability rather than direct toxicity. A proposed ecological framework for evaluating impacts of these spills includes individual-based population models, a site conceptual trophic model, habitat suitability maps, and a stochastic brine spill generator. Simulation results for mammal populations in the Tallgrass Prairie Preserve petroleum exploration and production (E&P) site in Oklahoma are presented. The persistence of simulated American badger (*Taxidea taxus*) populations decreased with increasing brine spill area. The decline in persistence and average final population size was much steeper in highly fragmented landscapes. The simulated time to extinction for prairie vole (*Microtus ochrogaster*) populations showed a threshold at 30% habitat loss from spills; above this threshold the time to extinction decreased with increasing spill area. Vole density was sensitive to the interaction of predation and fragmentation, with fragmentation causing population extinction in the presence of predation, yet stabilizing the population in the absence of predation. We anticipate that our results will aid in future development of "exclusion criteria" for leaving unrestored habitat at E&P sites.

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Introduction

Petroleum exploration and production (E&P) sites are often located in rural areas with diverse populations of mammals and birds. Terrestrial vertebrates may be exposed to hydrocarbons from petroleum spills or salts from brine spills; however, the more important exposures may be indirect, i.e., the removal of habitat and forage vegetation. Therefore, researchers from Lawrence Livermore National Laboratory and Oak Ridge National Laboratory have investigated the role of disturbance patches on vertebrates at the Tallgrass Prairie Preserve (TPP) in Osage County, Oklahoma, an E&P site (reports at http://gis.llnl.gov/mei/). This research has two, long-term goals: (1) to develop an ecological framework for evaluating impacts of brine and/or oil spills at E&P sites, utilizing population models based on patchiness of landscapes and, in some cases, trophic transfer; and (2) to develop thresholds (if possible) based on size and distribution of spills that would result in *de minimis* impacts on wildlife populations. These "exclusion criteria" could be applied to exclude certain well or spill locations from formal ecological risk assessment. In addition, this ecological approach could be used to inform (1) restoration priorities and strategies for companies that may be undertaken prior to exiting a site, or (2) siting and construction of drilling and road locations and associated E&P infrastructure in newly accessed areas.

An ecological approach may be superior to a toxicological approach for assessing population viability at E&P sites. Few studies have measured direct toxicity to vertebrates at petroleum-impacted sites, and these have typically measured biomarkers within an individual, not abundance or reproduction within a population (Charlton et al. 2001, McBee and Wickham 1988; McMurry et al. 1999). Moreover, several factors mitigate against toxicological risk at E&P sites. Hydrocarbon and salt contaminants are not generally taken up by the components of the wildlife diet. Plant uptake of hydrocarbons is usually low (Chaîneau et al. 1997; Anghern et al. 1999). Moreover, where phytotoxicity is evident, as with brine scars (API 1997), plants are largely absent. Earthworms, a common component of the wildlife diet in many ecosystems, tend to avoid moderate levels of hydrocarbons in soil (Wong et al. 1999) and saline soils (Piearce 1982). Metal constituents of crude oil may not be present at toxic concentrations. Thus, toxicological risk to wildlife at E&P sites may often be negligible.

If forage is absent, habitat suitability is low. Even if invertebrate or vertebrate prey are present, vertebrate consumers often avoid disturbed areas. However, the empirical evidence concerning avoidance relates to infrastructure rather than spills and is equivocal. For example, caribou avoided human developments, including wells, roads and seismic lines, showing maximum avoidance distances of 1000 m from wells and 250 m from roads (Dyer et al. 2001). Elk avoided a recently installed oil well but continued to include it within their home range (van Dyke and Klein 1996). Anecdotal evidence suggests that badgers (*Taxidea taxus*) do not construct burrows on spill sites at the TPP. Moreover, the odor of hydrocarbons may deter some species; food avoidance has been observed for other contaminants, particularly pesticides (Pascual et al. 1999; Kononen et al. 1987). We found as many exceptions to the hypothesis of avoidance: (1) caribou did

not avoid oil field infrastructure (Cronin et al. 1998); (2) house mouse (*Mus musculus*) abundance was higher in petroleum-contaminated, disturbed areas than in uncontaminated, disturbed areas in former tallgrass prairie ecosystems (Lochmiller et al. 2000); (3) most black bears did not alter the size or location of home ranges at an E&P site in Alberta, Canada (Tietje and Ruff 1983); and (4) lesser prairie chickens use lek (male group display) sites on abandoned oil pads and soils denuded by herbicide treatments (Haukos and Smith 1999; NRCS and Wildlife Habitat Council 1999). Furthermore, several studies of small mammals suggest that habitat fragmentation can sometimes have beneficial or neutral effects on population densities (Dooley and Bowers 1998; Aars et al. 1999). These vertebrates may not avoid disturbed areas.

Habitat disturbance can have adverse effects on population abundance or reproduction for various reasons. Individuals unable to find territories may emigrate. Movement costs may increase for animals that avoid or do not settle in disturbed areas. Forage vegetation or prey may be less available. Remaining habitat may provide fewer refuges from predators. Population declines due to habitat loss may ultimately lead to local extinction at low population densities because of the inability to find mates or breeding territories ("Allee effect," Allee 1938).

Brine and hydrocarbon spills at E&P sites (as well as wellheads, roads, burned areas, grazed areas, mowed areas, etc.) can be considered islands of disturbance in a sea of good habitat (Fig. 1). As accidental brine spills occur during production, underlying soil becomes saline, and the exposed area of the landscape is usually denuded of vegetation. Denuded soil is exposed to erosion, causing an enlargement of the denuded area with time (API 1997). Restoration of brine scars is possible, but revegetation with nonnative, salt-tolerant species or unpalatable plants (Keiffer and Ungar 2002) does not necessarily restore suitable habitat for native species. In contrast, hydrocarbons degrade rapidly, and fertilization can lead to recovery of production of vegetation within a few years.

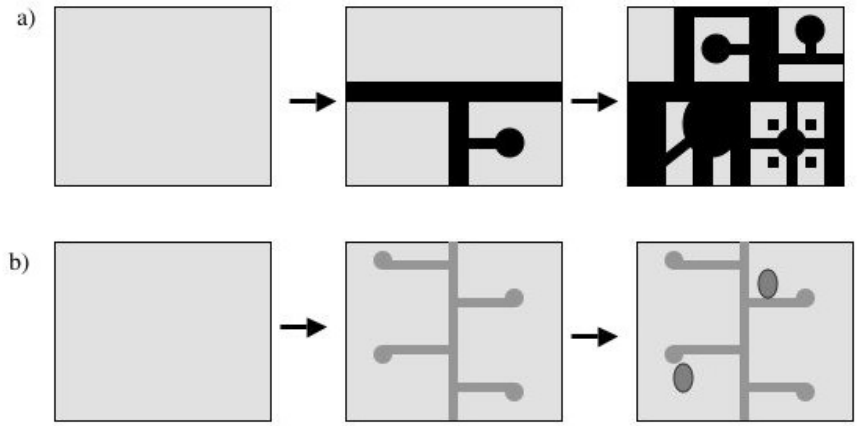


FIG. 1–*Two views of habitat fragmentation: a) "Traditional" habitat fragmentation resulting from industrial or residential development, black representing paved roads and buildings that create a hard barrier to migration and dispersal; and b) fragmentation from chemical releases to the environment, gray representing dirt or gravel roads and well pads that form a "porous" barrier (adapted from Carlsen et al. 2004)*

Data on the spatial and temporal frequencies of spills at E&P sites are not readily available. 567 brine spills were reported in Louisiana between 1990 and the first half of 1998 (Bass 1999), but others may have gone unnoticed. 900 brine spills per year were reported by the state of Oklahoma between 1993 and 2002 (Jager et al. 2004a).
Even though the evidence above suggests that habitat disturbance is more important at E&P sites than chemical toxicity, most spatially-explicit models that have been developed for ecological risk assessment emphasize foraging and chemical bioaccumulation through the food chain, rather than habitat preferences and species life histories (Freshman and Menzie 1996; Clifford et al. 1995; Baveco and de Roos 1996). Other models include habitat preferences but do not incorporate species life histories or address situations in which habitat is removed (Linkov et al. 2001; Henriques and Dixon 1996; Hope 2000). Population models that address habitat fragmentation simulate movement of animals between patches of suitable habitat, rather than population-level effects of unsuitable habitat (Gustafson and Gardner 1996). In contrast, individual-based models (IBMs) can simulate mechanistic linkages between the physical environment, as modified by human activities, and animal populations.

This chapter describes individual-based modeling methodologies and results for two species (American badger, prairie vole) at the TPP, using models emphasizing different aspects of vertebrate ecology (e.g., habitat suitability, predator-prey relationships). We describe a spill generator program that can create permanent or temporary brine or hydrocarbon spills of varying size and number. An ecological framework for evaluating vertebrate population impacts at E&P sites is described which incorporates population models. In the future, results of population models are expected to inform recommendations for no-effect criteria that would exclude E&P sites with particular spill densities or patterns from rigorous ecological risk assessment requirements.

Developing an Ecological Framework: The TPP Case Study

Tallgrass Prairie Preserve

Our case study site, the TPP in northeastern Oklahoma (Fig. 2), consists of 15,200 hectares of prairie grassland owned by the Nature Conservancy (ONHI 1993; Hamilton 1996). Additional terrestrial habitats found at the TPP include upland deciduous forest, deciduous riparian forest, grassy riparian habitat, disturbed areas, and rocky outcrops (ONHI 1993; Payne and Caire 1999). About seven percent of oil and gas well locations in the conterminous U.S are in tallgrass prairie and 32 percent of wells are found in prairie ecosystems, generally (Fig. 3). Thus, results from the TPP would be expected to be somewhat representative of those that might occur at a large fraction of E&P sites.

The TPP supports a wide variety of plant and animal species (many of which are prairie-dependent) and represents one of the last substantial remnants of the tallgrass prairie ecosystem, which historically covered 5.7 million ha of the United States and Canada (Madson 1990). Bison were reintroduced to the TPP in 1993. Since that time, the Nature Conservancy has used fire (median burn rate of 6700 ha/yr) and bison grazing as management practices for prairie restoration, reenacting the natural disturbances that historically functioned to maintain the ecosystem (Hamilton 1996). The TPP is an E&P site with more than 600 historic and 120 active oil and gas wells. The site contains five

large, historic brine scars; several recent (within past two years) spills of brine, oil, or both; and older spill sites (8 to 15 years). The total brine spill area is approximately 17 ha (about 0.1% of the total preserve area), with a median spill size of 0.02 ha and a maximum spill of 4.9 ha. Additional wells are located outside of the TPP in Osage County. Approximate total areas of roads, well disturbance, vegetation classes, pastures, bison paddocks, and streams are available from the authors.

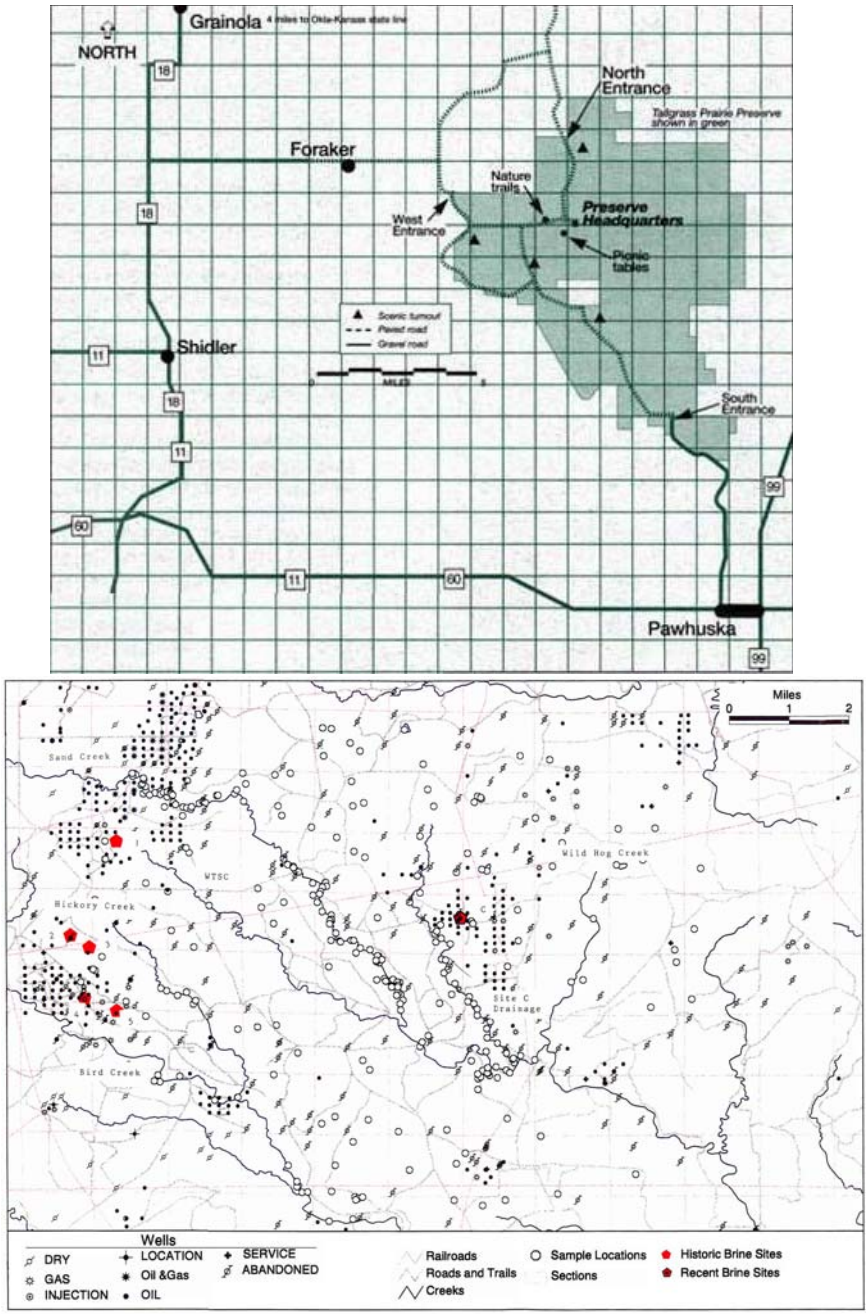


FIG. 2–*Location of the Tallgrass Prairie Preserve (used by permission fromBob Hamilton of the Nature Conservancy) and exploration and production activities (courtesy of Kerry Sublette, University of Tulsa).*

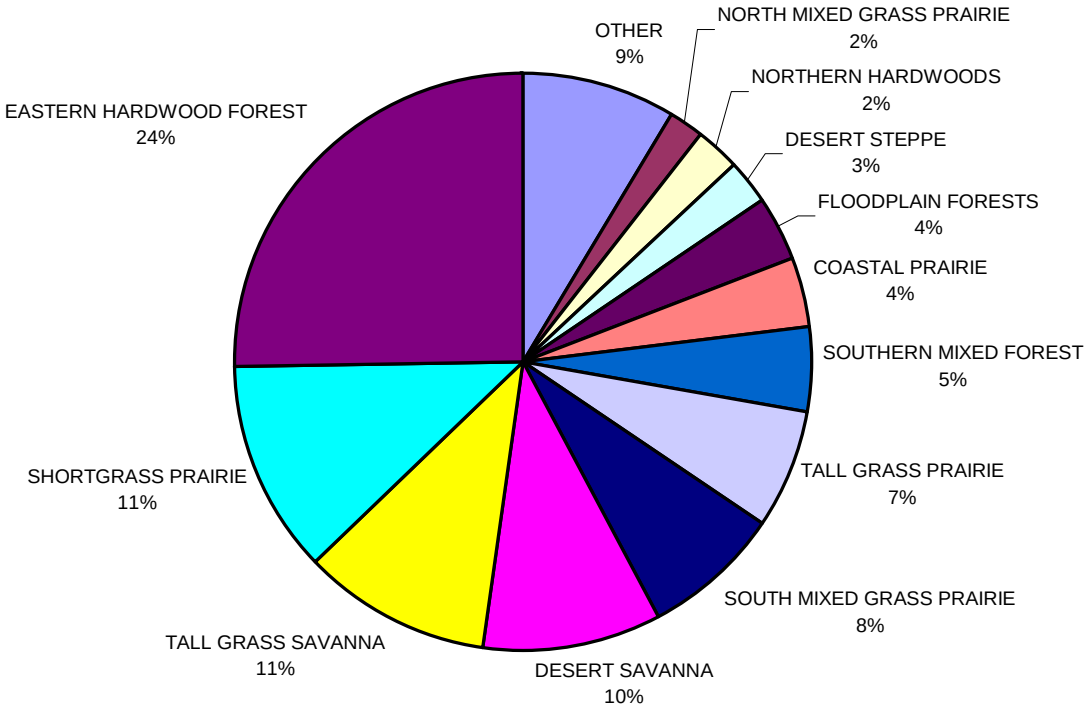


FIG. 3–*Predominant Kuchler vegetation forms in ¼ mile by ¼ mile cells in which productive and unproductive oil and gas wells in the U.S. are located. Data on well locations from the 1995 National Assessment of Oil and Gas Resources were obtained from David Ferderer at USGS and are now available at http://energy.cr.usgs.gov/oilgas/noga/.*

Geographic Information Systems (GIS)

Both the development of this ecological framework and the use of spatially-explicit IBMs require spatial data. We developed a GIS data collection protocol to investigate and manage ecological impacts at E&P sites (Hall et al. 2001). The protocol describes recommended data for assessing ecological impacts and their sources, as well as procedures for 1) assessing the quality, accuracy, precision and applicability of the data; 2) establishing a common projection system; 3) associating tabular data with spatial locations where useful; and 4) preprocessing or correcting the data when necessary. The GIS for the TPP was developed using Arc IMS, Arc INFO and associated modules as the primary GIS analytical engine. Table 1 shows the uses of these spatial data for modeling spills and vertebrate populations at E&P sites. National, state and site-specific sources of these data are identified in Hall et al. (2001).

TABLE 1—*GIS layers for use in modeling vertebrate populations*.

Data layer	Use
Digital elevation model	Predicting probability of pipeline rupture, flow of brine and oil, soil erosion potential, slopes unsuitable for animal movement
Raster coverage of vegetation categories	Depicting forage, predator refuges; contributing to habitat suitability designations
Digital Orthophoto Quarter-Quadrangles, Landsat Images	Depicting changes in spill boundaries and habitat suitability over time, with ground-truthing
Vector coverages of roads, fence-lines	Depicting potential barriers to movement or contributing to habitat suitability designations
Vector coverage of site boundary	Depicting boundary of local population of concern
Raster coverages of well locations, tank farms and other structures	Depicting potential barriers to movement or contributing to low habitat suitability designations
Raster coverages of vegetation disturb-ances (e.g, prescribed burns, grazing)	Contributing to habitat suitability designations
Raster coverage of soil taxonomy	Providing soil texture information relevant to burrowing mammals

Conceptual Trophic Model

A conceptual trophic model describes interactions among ecological receptors at a site and identifies the important populations that may be the focus of ecological evaluations
and management and significant trophic relationships that may be included in an individual-based, predator-prey model. Three steps are involved in the construction of a conceptual terrestrial trophic model (Stevenson et al. 2001): (1) creating a list of species expected to occur at the site, (2) assigning the species to guilds, and (3) constructing the food web through an analysis of the relationships between guilds. The conceptual model focuses on guilds with high societal interest or that are representative of many species.

Species data can be gathered from a variety of sources. For the TPP, these included a report of the Oklahoma Biological Survey (ONHI 1993), species lists obtained from the Nature Conservancy (1996) and Oklahoma State University (Palmer, unpublished data), and open literature references. Species data from similar, well-studied ecosystems (e.g., National Science Foundation Long Term Ecological Research sites) are also useful.

The procedure for assigning species to alpha and beta guilds is described in Stevenson et al. (2001), based on the guild definitions of Wilson (1999). Alpha guild members use a class of resources in a similar way. Beta guild members share similar space along environmental gradients (i.e., occupy the same or a similar niche). Alpha guilds are the most important for creating a conceptual food web, but beta guilds provide information about species interactions such as competition.

At the TPP, we determined relationships between the guilds based on dietary information collected in the creation of the alpha guilds. Sixteen alpha animal guilds and six beta guilds are found at the TPP (Stevenson et al. 2001). Table 2 shows the community food web at the TPP, describing the binary feeding relationships between the alpha guilds in the community. Figures in Stevenson et al. (2001) show the source webs (Pimm et al. 1991) of the guilds of vertebrates of the TPP.

TABLE 2—*Food relationships between the various alpha guilds at the Tallgrass Prairie Preserve, OK. An "x" denotes that a given prey species along the y-axis may be consumed by the corresponding predator on the x-axis.*

		Predators															
		AI	BI	HM1a	HM1b	HM2	OM1	OM2	OB	OH	CH1	CH2	CB1	CB2	CM1a	CM1b	CM2
Prey	Plants	X	X	X	X	X	X	X	X	X							
	Detritus	X	X				X	X	X	X				X	X		
	AI	X					X	X	X	X	X	X	X	X		X	
	BI		X						X	X	X		X		X		
	HM1a						X										X
	HM1b						X										X
	HM2						X		X			X		X			X
	OM1						X							X			X
	OM2						X		X			X		X			X
	OB						X		X	X		X		X			X
	OH						X										X
	CH1						X	X	X			X		X			X
	CH2						X					X		X			X
	CB1						X		X	X		X		X			X
	CB2																
	CM1a						X					X		X			X
	CM1b						X							X			
	CM2																

Key to Alpha Guilds

Plants	Plants and Fungi	OB	Omnivorous Birds
Detritus	Detritus and Carrion	OH	Omnivorous Herptiles
AI	Aboveground Invertebrates	CH1	Invertivorous Herptiles
BI	Belowground Invertebrates	CH2	Other Carnivorous Herptiles
HM1a	Herbivorous Mammals - grazers	CB1	Invertivorous Birds
HM1b	Herbivorous Mammals - browsers	CB2	Raptors
HM2	Small Herbivorous Mammals	CM1a	Fossorial Invertivorous Mammals
HB	Herbivorous Birds	CM1b	Other Invertivorous Mammals
OM1	Large Omnivorous Mammals	CM2	Large Carnivorous Mammals
OM2	Small Omnivorous Mammals		

Artificial Landscapes

A map of the brine spills at the TPP and a discussion of structures (e.g., pipelines, well heads, tank batteries) that affect spill probabilities is presented in Jager et al. (2004a). However, the TPP case study reported here relied on artificial maps of disturbance features. Artificial spills are necessary to identify potential spill area or fragmentation thresholds that result in Allee effects. Maps with different spill patterns also aid in understanding causes of declines. Three methods were used to develop

artificial landscapes. For the trophic (vole) model described below, we used a heuristic method where the spill area, size, number and placement were based on knowledge of existing or potentially possible conditions present at the TPP. For this method we conducted simulations that distributed various numbers of 0.09-ha spills (30 m by 30 m cells) randomly across the TPP and simulations that distributed impenetrable structures (such as roads and fences) across the landscape, resulting in fragmentation. For the badger model two spill generators were developed, one theoretical, and one more realistic and dependent on pipeline distribution (Jager et al. 2004a). The theoretical model distributes spill centers randomly in two-dimensional space. A Dirichlet distribution is used to allocate the total spill area across spill centers, and this area is distributed using a random walk algorithm to simulate diffusion into neighboring cells. The well-complex model simulates spills along gathering lines that connect each well in a rectangular grid (based on many of the well arrangements at the TPP) with a tank battery located at one corner. The user specifies the number and dimensions of well complexes. The model assumes that the likelihood of encountering a spill along any segment of pipe of a specified length is constant, so that the likelihood of a spill within a cell increases with the length of pipe located within its boundaries. The distance along the pipe to the next brine spill is a gamma variate, and area of each brine spill is simulated as a Dirichlet variate, which ensures that the specified total area of spill is exactly met. Badger simulations presented in this manuscript were performed using landscapes created with the theoretical spill generator.

Spatially Explicit Individual-based Models

Two spatially-explicit IBMs were developed for terrestrial vertebrates. The structure of the template for the two models is described in Fig. 4. Each spatial cell, as well as its immediate surroundings, is conceived as the source of food resources and shelter for individual animals. The models can simulate population changes over time in response to disturbances by fire, petroleum spills, and brine spills, though only static brine scars and other static habitat disturbances are presented here. Modeled events include local biological processes that influence individual animals (e.g., mortality, reproduction, aging, mating choice) and external or landscape-wide events (e.g., disturbances, redistribution of organisms).

Habitat Model

Habitat IBMs are well suited for studying the differential susceptibility of species with different life histories and habitat requirements to habitat loss from brine spills. We implemented a habitat-based model for the American badger (*Taxidea taxus*), a voracious, solitary predator with low tolerance for other individuals. Here, we provide a brief overview of the model, which is described more completely by Jager et al. (2004b). We assign habitat suitability indices to various vegetation categories based on known compatibilities with the presence of small, fossorial mammals or burrowing requirements. Brine spills, structures, and streams are designated as unsuitable habitat. Habitat quality of cells influences reproduction simulations through acquisition of territory used for

breeding and survival via movement costs and habitat-related mortality. This model does not explicitly represent foraging or predation.

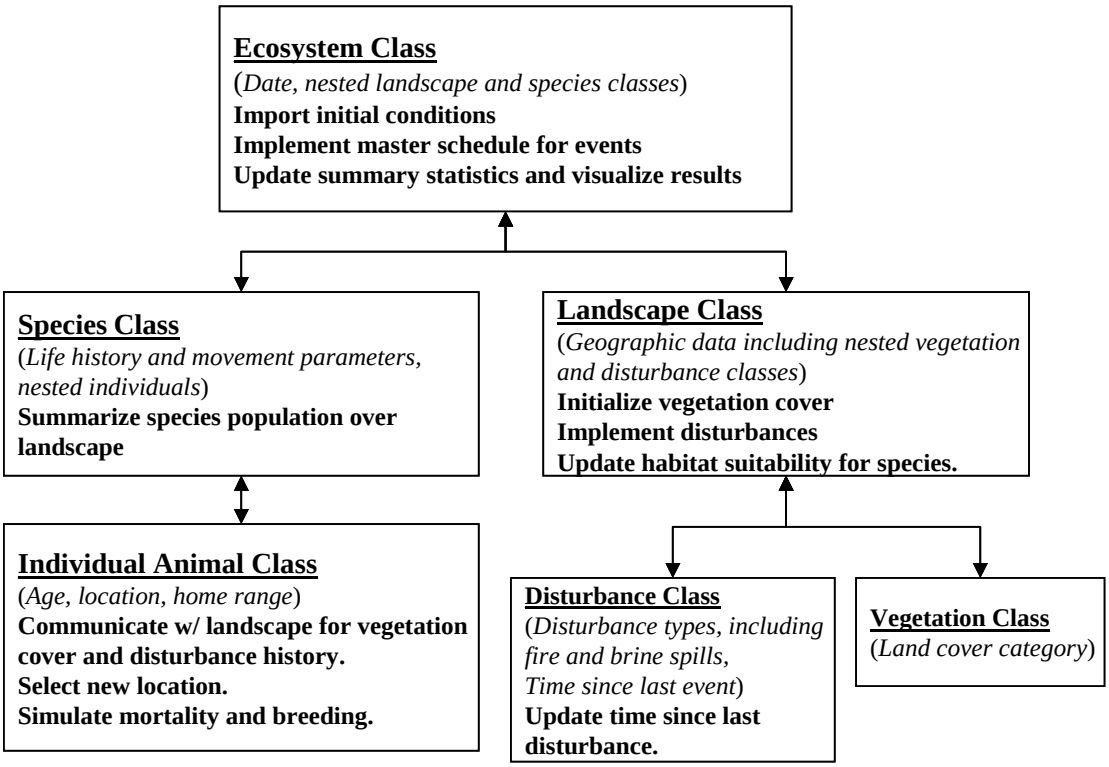


FIG. 4–*Diagram of the general model template. Objects in the model are defined by classes that include data members (italics) and member functions (bold). Each class is represented by a box in the diagram.*

Individuals pass through five periods of pre-breeding, mating, post-mating, birthing, and rearing of young. Juveniles seek to establish a permanent home range, equivalent to the breeding territory, once they leave the care of their mother. Once this range is established, a badger's movements are restricted to cells within its range. Movement depends on season and gender. Mating is assumed to occur for any mature female with a home range overlapping the home range of at least one mature male. Reproduction timing and survival of young are also described in Jager et al. (2004b). Sources of mortality include: baseline, age-related mortality; habitat-related mortality; mortality due to intraspecific aggression; mortality based on movement; and emigration from the study area. Sensitivity analyses are in progress.

We conducted a simulation experiment to investigate the effects of loss of habitat area and fragmentation (represented here by increasing numbers of spills). We used the statistical model described above and in Jager et al. (2004a) to generate spill landscapes with a specified target percentage of area covered by spills (0%, 1%, 10%, 20%, 30%, 40% and 50% and a specified number of spills (100 or 1000). Effects of spills on final average population sizes and the proportion of potential breeding females that successfully mated were compared. In addition, we quantified Allee effects.

Results showed a decrease in the average size of badger populations at the end of a 100-yr simulation with increasing area of habitat loss (Fig. 5a). This decrease was accompanied by a similar decline in the proportion of replicate populations that persisted (Fig. 5b). Results of the Jager et al. (2004b) study suggest that thresholds, defined as steeper declines in persistence with increasing habitat loss, occur when the habitat is highly fragmented by disturbances. Parameter explorations show that steeper, threshold-like declines occur when the mortality risk in poor habitat is high and when poor habitat is not excluded from the animal's territory. The decline in persistence associated with habitat loss was caused by a combination of elevated habitat-related mortality and increased difficulty in finding mates (Allee effects) (Fig. 5c). Fragmentation increased the difficulty in carving out high quality territories and increased mortality during the dispersal phase. The good news is that the likelihood of persistence is high for landscapes with fragmentation characteristics similar to those found at the TPP, that is 0.1% of the area covered with brine scars (and less than 1% of the area directly disturbed by wells, roads or spills). However, empirical verification of these modeling results is necessary before any conclusions can be drawn.

Comparing minimum habitat requirements for a social and asocial prairie species, Wolff (2001) identified behavioral attributes that influence species response to habitat loss for mammals, including (1) habitat specificity, (2) social structure, (3) dispersal ecology, and (4) trophic level. In our framework, we view these attributes as part of the spatial life history of a species. Future research with the habitat IBM will focus on how differences in social structure influence species responses to fragmentation and habitat loss due to brine spills. The badger represents one extreme: an asocial animal that is solitary and highly intolerant of same-sex conspecifics. We hope to contrast our results for the badger with a social breeder, such as a prairie chicken. During the breeding season, male prairie chickens aggregate into lekking displays on bare, elevated areas surrounded by grasslands. Large breeding aggregations benefit from group defenses against predation, as well as access to mates. Simulations may show that social breeders are more susceptible to habitat loss than asocial species because of strong Allee effects. This result would be consistent with field observations that suggest a threshold lek size. Alternatively, simulations may show that social species are better at packing into small habitat areas, and benefit from a brine spill because it creates lekking sites.

Trophic Model

Trophic IBMs focus on interactions that may cause indirect, vertebrate population-level effects associated with habitat loss (e.g., vegetation growth and reduction due to grazing, herbivory, and bioenergetics). The trophic approach captures the interdependence between population density and environmental characteristics such as vegetation density, unsuitable vegetation, and climatic dependence. Trophic concepts were the leading principles of a model that was implemented for the prairie vole (*Microtus ochrogaster*) (Kostova and Carlsen 2003, Kostova et al. 2004), a monogamous herbivore that feeds on grassland vegetation and is preyed upon by predators such as owls, badgers, and snakes. A large number of well-established experimental values are available for parameterization of the prairie vole model. Depending on the availability of

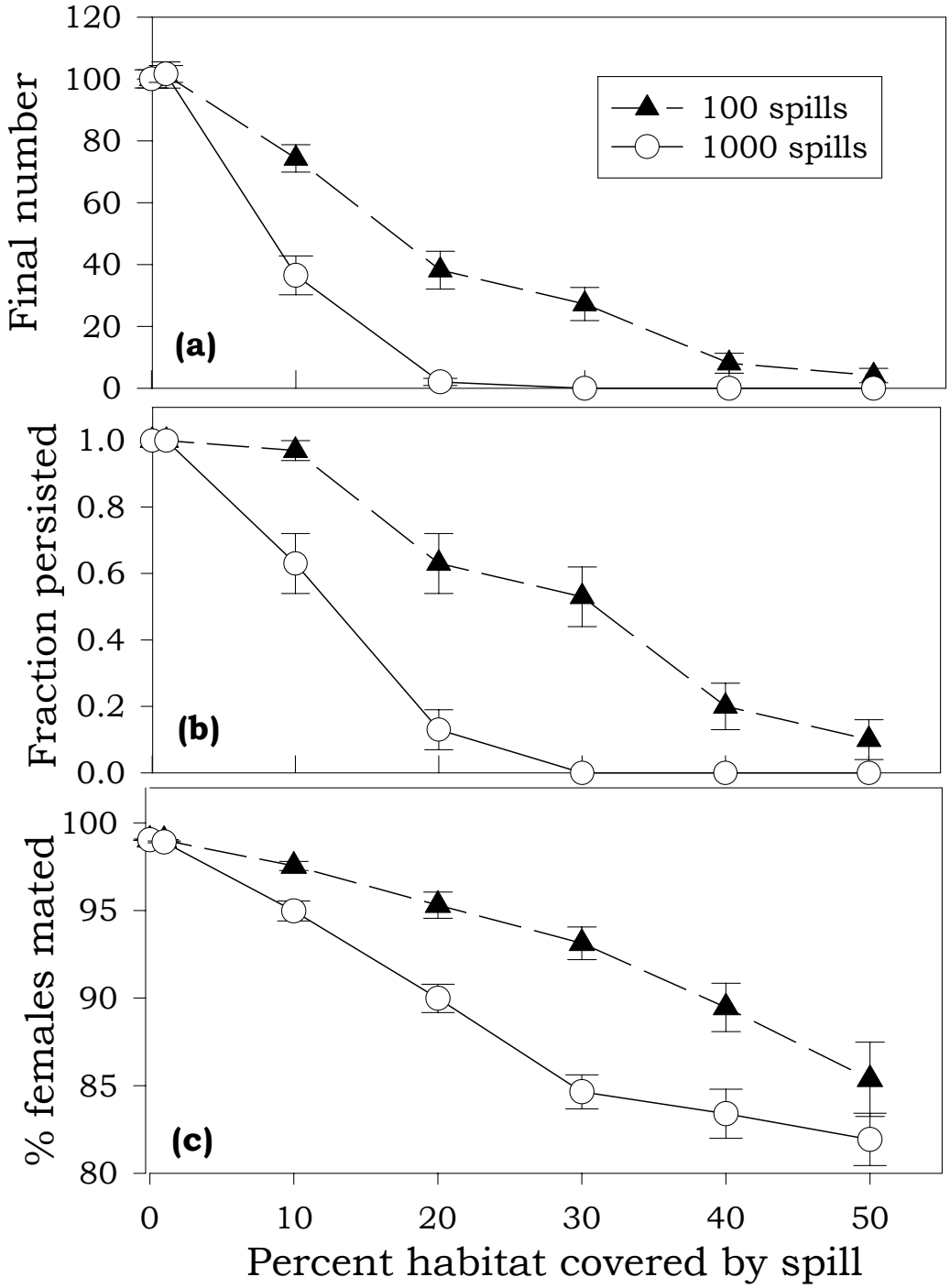


FIG. 5–*Response of simulated (a) final population size and (b) fraction of replicate populations that persist, and (c) fraction of females eligible to breed that find mates to habitat loss. Error bars show 1 standard error surrounding the mean of 30 replicate simulations for landscapes with 100 and 1,000 spills (Jager et al. 2004b)*

sufficient species data, the model is extendable to other monogamous rodent and can be adapted for polygamous species as well.

The model uses growth rates of grasses obtained from a 30-year simulation of CENTURY v. 4.0 (Parton et al. 1993) with historical temperature and precipitation data. The vegetation growth rates are combined with the grazing rate of the cumulative vole population in each cell. Body mass and metabolic status of each individual determine behavioral characteristics, such as territorial competition, mating success, and dispersal. Metabolic status is adjusted for the pregnancy or weaning status of females. Voles do not produce offspring unless a pair is formed from two dispersing animals (floaters) that occupy the same cell.

The survival of an individual vole depends on the availability of vegetation and the individual's physiological status. Starvation, age and predation are factors that contribute to the death rate. Predation is incorporated into the model by removing a density-dependent fraction of the voles.

The spatial structure of the model is based on the notion of home range. A simulated landscape is represented as a collection of cells whose size is equal to the home range of the vole. Voles are residents of a cell or floaters. The status changes over time depending on vegetation availability, age, body size, presence of a potential mate, etc. Floaters choose new cells based on vegetation suitability and quantity as well as on opportunities for mating. In cases when the current cell is on the border of the region and the floater cannot find an unpopulated cell into which to move, it is forced to leave the modeled region.

A series of runs with random initial animal distributions and spill locations were performed in order to establish the dependence of population density and average time-to-extinction (ATE) on factors such as predation level, available habitat size, fragmentation caused by barriers dividing the landscape into connected patches and fragmentation caused by spills. Simulations were performed on artificial, square habitats with uniformly growing tallgrass vegetation as well as on a landscape representing the TPP, using geospatial and vegetation data.

Effect of Area and Predation—Patch size and predation were found to have a combined effect on population density. The reduction of habitat area led to higher vole population density in the absence of predation and dispersion, which destabilized the vole population and decreased the ATE (Kostova et al. 2004). However, the reduction of habitat area had little effect on the maximum population densities in simulations if changes in predation and dispersion were taken into consideration (Fig. 6). Increasing the predation level had the clear effect of decreasing population density. However, the shapes of the density curves, i.e. the locations of the minima and maxima and periodicity, were not sensitive to either area or predation level change (Kostova and Carlsen 2003). Both area and predation level had a significant effect on the time to extinction. Figure 7 represents the ATE at three predation levels.

A direct relationship was observed between the habitat area and the ATE; the larger the patch, the higher the ATE. On the largest patches for the low and intermediate predation levels, almost all simulations produced populations that persisted for the whole 30-year period.

Low predation levels led to dramatically decreased persistence, which can be explained by high vole densities leading to overgrazing in the months of low vegetation.

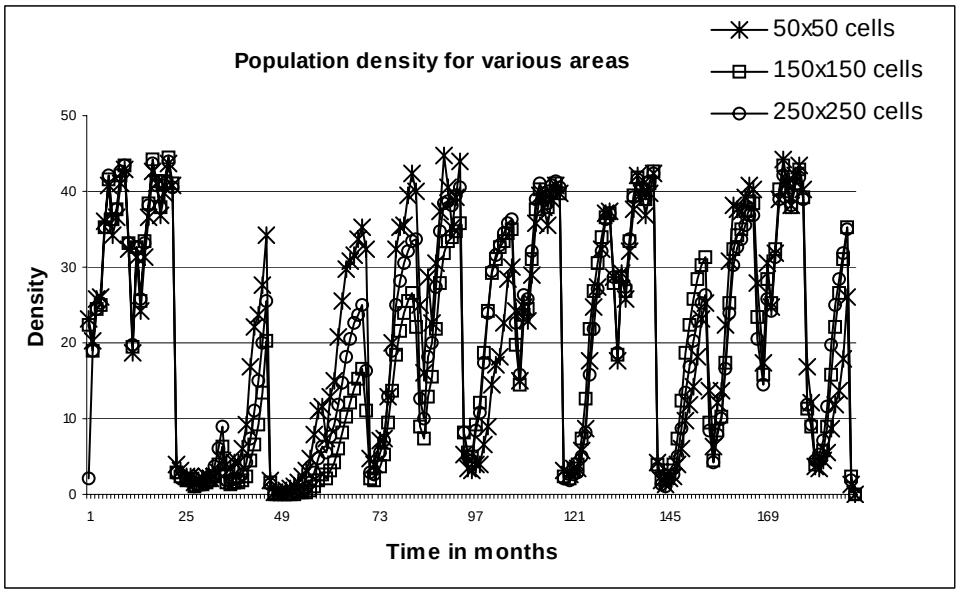


FIG. 6–*Predicted population density for three different areas of artificial square habitat; a_A=0.02, a_J=0.04, where a_A and a_J are fraction of adults and juveniles removed, respectively.*

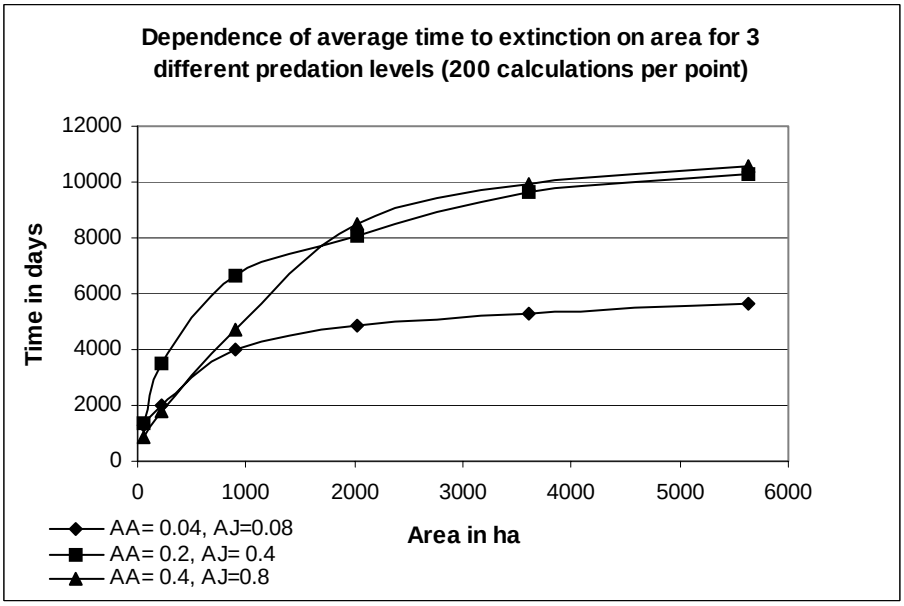


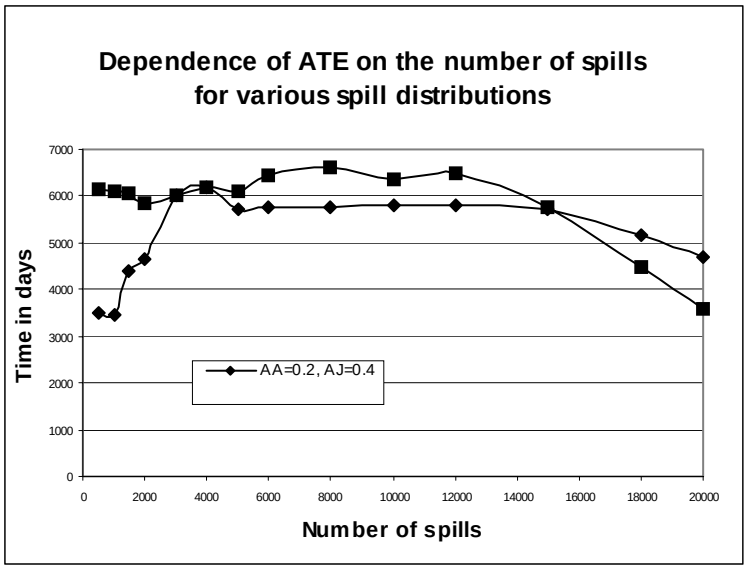
FIG. 7–*Average time to extinction (ATE) as a function of habitat area and predation, averaged over 200 simulations. If the population persisted for the whole simulation period, ATE was taken to be 30 years (10800 days). AA and AJ are fraction of adults and juveniles removed, respectively.*

However, patches of small area displayed low ATE at all predation levels. Because the maximum densities were similar for various areas, the high extinction risk on the smallest simulated patches is not connected with density effects. One possible explanation is that small patches provide lower numbers of surviving individuals to restore the population after a period of insufficient vegetation and no births (e.g., in winter months). The simulations reveal that for patches of practically all sizes, there is an "optimal" predation level for which the ATE is highest (Kostova and Carlsen 2003). Using the "optimal" predation coefficients also resulted in vole population densities characteristic for tallgrass prairie, i.e., 5-7/ha (Getz et al. 2001).

Effect of Non-spill Fragmentation—Habitat fragmentation contributes to the reduction of available habitat and would be expected to lead to a lower ATE. However, the effect of fragmentation on population persistence appears to depend on the population density of voles. We modeled enclosures (no dispersal across boundaries) with no predation (Kostova et al. 2004). The simulations yielded high population densities with mortality mainly due to winter starvation. Fragmentation had a positive effect on population persistence, as it reduced population density and stabilized the populations.

Effect of Spills—The introduction of predation and dispersal as well as more accurate trophic calculations produced lower population densities (Kostova and Carlsen 2003). The effect of introducing randomly distributed "spills" consisting of separate polluted cells in artificial vegetation grids was investigated by performing simulations with an increasing percentage of spills on the patch. The area of the grids was increased so that the inhabitable (unpolluted) area was kept constant at 10,000 cells. Figure 8 presents the results of simulations with two predation levels. Each point represents the result of 200 simulations performed by either fixing an initial animal distribution and varying the number of spills or fixing the spill distribution and varying the initial animal distributions. The fragmentation actually had a beneficial effect, increasing the ATE, in the case with "low" predation if as much as 50% of the area was covered by random spills. Fragmentation had a similar effect as predation in decreasing population densities, and this explains the observed phenomenon. In the case of the "optimal" predation level, fragmentation did not have any effect on the ATE if up to 30% of the area was covered with spills. For spill areas above these levels, the ATE decreased with increasing percentage of spill area.

Effect of Spills and Other Sources of Fragmentation at the TPP—We investigated the combined role of spills and other sources of fragmentation on the persistence of vole populations at the spatial scale of the TPP. Runs with different initial vole distributions inevitably resulted in extinction of the vole population in the southern and western portion of the TPP, which are very fragmented by roads (not considered as barriers in the experiments) and in rivers and patches of non-grass vegetation (considered as barriers). Voles persisted in the northeastern part of the preserve, which is not as fragmented (Fig. 9a). In other experiments, hypothetical random distributions of 1000 spills (or development sites) of the size of one cell (0.09ha) were placed in the northeastern portion of the preserve, and simulations were carried out at the "optimal" predation level. (Fig. 9b). However, due to the large scale of the simulation on the TPP landscape, the number of simulation repetitions was insufficient for a valid prediction. The relatively small number of simulations resulted, in some of the cases, in the extinction of the population



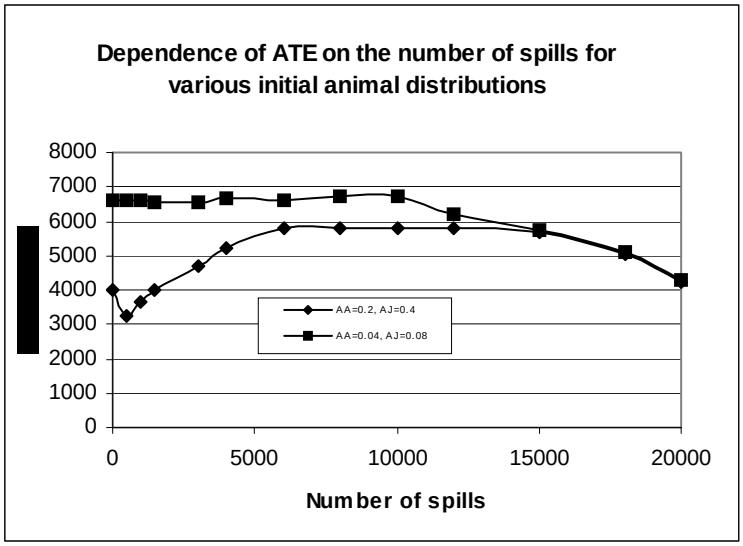


FIG. 8–*Dependence of ATE on the number of spills in an area with 10000 inhabitable cells for various spill and animal distributions. AA and AJ are fraction of adults and juveniles removed, respectively*

in the areas with spill fragmentation even before this happened in the naturally fragmented areas, while the same populations persisted for the whole 30-year period in the absence of spills.

Model Validation

Rigorous model validation has not yet been performed. However, the results of simulations with the trophic model for voles were compared with time series density data from the literature (Krebs et al. 1969; Getz et al. 2001), and the model was adjusted accordingly. The model predicted correctly the average vole density, multiyear

a) b)

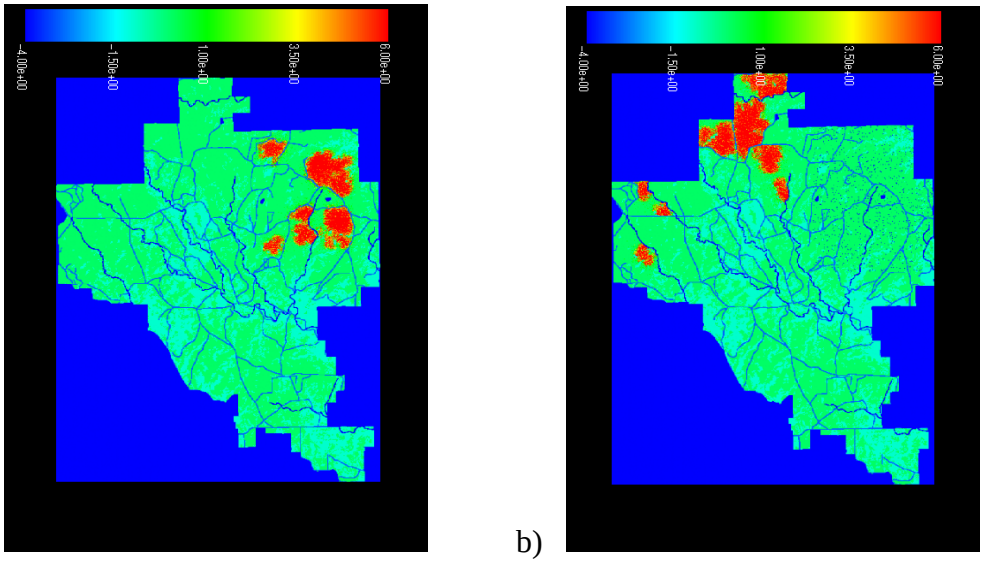


FIG. 9–*Two instantaneous plots of simulations of vole density dynamics at the scale of the TPP, using a) a realistic representation of roads (blue), rivers (dark blue) and patches of non-grass vegetation (light blue), and b) a distribution of 1000 artificial spills in the Northeast. Red represents areas of high vole density that change in location and size during the year.*

fluctuations at higher densities, annual fluctuations at lower densities, and annual dynamics with peaks in October to December and minima in February to March. Studies of impacts of brine and E&P sites that would be useful for validation are rare, and even studies of habitat loss are rare for some vertebrate species. Moreover, multiple stressors that are present at E&P sites can confound field results. For example, Cronin et al. (1998) cite several investigators who recognize the difficulty of distinguishing human impacts from environmental stochasticity affecting caribou herds. Field verification of model results is planned.

Toward an Ecological Framework for E&P Sites

Framework Components

A preliminary ecological framework for evaluating terrestrial vertebrate populations at E&P sites is presented in Fig. 10. Assessment endpoint populations are chosen using a site conceptual trophic model and other management criteria. The framework includes two parallel paths for determining risk from toxicity or habitat loss. The potential for exposure to contaminants is determined by contaminant bioavailability and animal behavior. The threshold for conducting a toxicological risk assessment may be lower for threatened and endangered populations than for other populations. The spatial exclusion criteria (contaminated area thresholds) that determine whether an exposure to habitat disturbance may be significant and may require a spatial ecological assessment as described below. Species life history information, trophic relationships, and habitat

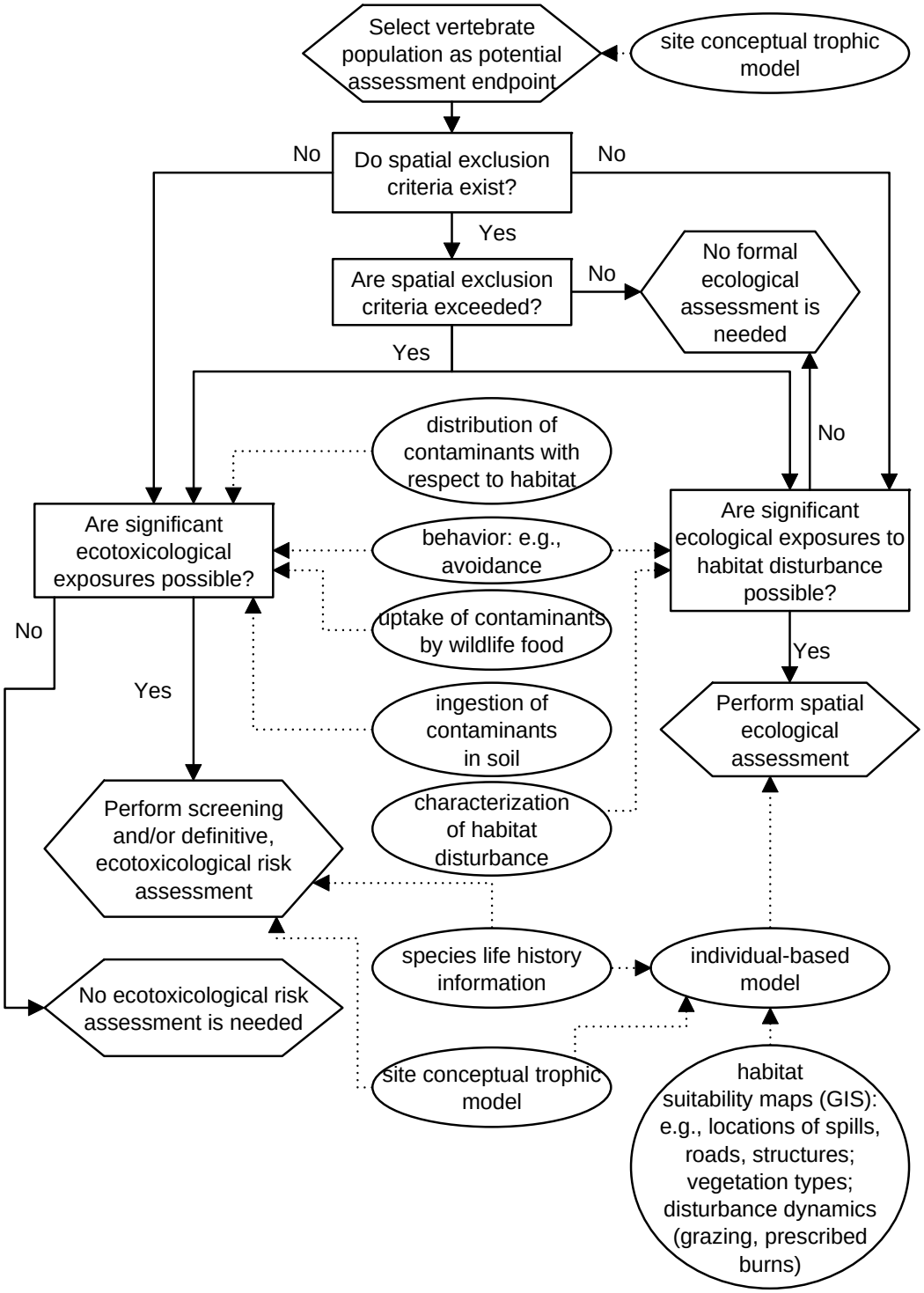


FIG. 10–*A preliminary ecological framework for evaluating terrestrial vertebrate populations at E&P sites.*

suitability may be explicitly or implicitly modeled with an IBM. If an ecological risk assessment is performed, the level of effort should be proportional to the magnitude of the risk management decision.

Spatial Exclusion Criteria

The large number of small brine and oil spills on E&P sites of high habitat value prompts the question of whether simple field criteria (e.g., threshold total area or particular distributions of spills associated with *de minimis* population-level effects) may be used to exclude the spills from formal ecological risk assessment. In the past, this question has been treated as a cost-effectiveness issue, with small spills simply being excavated, restored, or allowed to recover. Sorensen and Margolin (2002) review spatial scale ecological screening criteria for contaminated sites in various states. For example, the Pennsylvania Department of Environmental Protection assumes that two acres of surface soil contamination does not pose risk to vertebrate populations (PADEP 1998). One American Petroleum Institute paper suggests that a petroleum release to surface soil is not of environmental concern if it is farther than 500 ft from the nearest receptor or habitat (Claff 1999). However, these values are not based on a landscape approach to ecological assessment that specifically considers vertebrate populations.
Some guidance regarding habitat loss from spills might be distilled from existing ecological literature. Carlsen et al. (2004) review minimum patch size requirements (e.g., areas below which species are never found or which are associated with unsustainable populations) of several species and taxonomic groups for potential use in screening-level ecological risk assessments at E&P sites. Similar information may be available on the number of territories required to support sustainable populations. A caribou avoidance distance from wells is derived in Dyer et al. (2001). Massey (2001) notes that the Bureau of Land Management has the regulatory authority to move drill pads 200 m away from known lesser prairie chicken lek sites. A rule of thumb for carnivore density states that 10,000 kg of prey supports about 90 kg of carnivore, and this relationship "provides a basis for identifying species that require conservation measures" (Carbone and Gittleman 2002).

In this study, insufficient species, ecosystems, and model structures have been tested to recommend general criteria for excluding E&P sites from formal ecological assessment. However, relevant results are available for American badger and prairie vole. Based on our limited modeling of the American badger in grasslands, this species shows a decline in final population size with increasing habitat loss. If the modeling results were confirmed with field studies, a risk manager could set spatial exclusion criteria in the following manner. If a risk manager wanted an 80% likelihood of population persistence and 1000 spills were anticipated, then a spatial exclusion criterion of greater than 1% and less than 10% spill area could be chosen. One would choose a similar spatial exclusion criterion if a population of at least 50 badgers were desired at the TPP.

Our limited modeling of the prairie vole suggested a threshold at 30% habitat loss due to spills. Below this threshold, the average time to extinction was not affected. Above this threshold, the average time to extinction decreased with increasing spill area. Vole density was sensitive to the interaction of predation and fragmentation, with

fragmentation causing population extinction in the presence of predation and stabilizing the population in the absence of predation. Where threshold spill areas for population-level effects are observed, acceptable levels of effects are relatively easy for risk managers to specify.

Conclusions and Caveats

Modeling results from the TPP indicate that vertebrate populations may decline as the area of brine spills at E&P sites increases. However, the spill area associated with detrimental effects is probably much larger than the actual fractional landscape area directly disturbed by spills at the TPP (0.1%). The impacts of increased habitat fragmentation caused by spills, structures, and/or roads can range from beneficial (simulation of vole populations with no predation) to adverse (simulation of vole populations with predation and simulation of badger populations). Simulations of simplified ecosystems with only one explicitly modeled species at a time, on a relatively homogenous grassland landscape, yield complex results. Until sensitivity analyses are performed, the relative importance of life history parameters, habitat suitability designations, bioenergetics, territory acquisition algorithms, impenetrable barriers, predation, edge behavior algorithms, and other factors will be unknown. The dynamic nature of brine and petroleum spills, including chemical degradation, active restoration, or natural recovery time frames was not considered in these IBMs. Moreover, the modeled results have not been verified in field studies. Studies such as ours may help focus scientific and regulatory attention on potential ecological impacts and potentially away from potential toxicological impacts. Conceptual trophic models can be useful in focusing an assessment on appropriate species. IBMs may incorporate many realistic variables, and sensitivity analyses may identify those that are most important. The habitat model applied to the American badger identified situations leading to the existence of steep threshold responses to increasing disturbance areas. Results from both models can be used as qualitative guidance for land managers and regulatory agencies, although field experiments should be designed to check model predictions for quantitative accuracy.

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