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Raccoon (*Procyon lotor*) Harvesting on and near the U.S. Department of Energy's Savannah River Site

Utility of Metapopulation Modeling for Prediction and Management of Hunter Risk

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Understanding the toxicodynamics of wildlife populations in contaminated ecosystems is one of the greatest challenges in ecotoxicology today. The goal is to manage these populations to minimize risk to ecosystem integrity as well as human health. Ecological risk assessments (ERAs) in the United States are designed to meet the regulatory mandates of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) and the Resource Conservation and Recovery Act. According to the U.S. Environmental Protection Agency, an ERA evaluates the potential adverse effects that human activities have on the flora and fauna that define an ecosystem (U.S. Environmental Protection Agency 1997). When conducted for a particular geographic location, the ERA process can be used to identify vulnerable and valued resources, prioritize data collection, and link human activities with their potential effects. Risk assessment results provide a common framework for comparing different management options, thus enabling decision makers and the public to make better informed decisions about the management of ecological resources. The ERA uses available toxicological and ecological information to estimate the occurrence of a specified undesired ecological event or end point. The types of end points targeted for investigation depend on the objectives and the constraints imposed upon the risk assessment process (Newman and Strojan 1998) based on all of the relevant stakeholders; therefore, multiple endpoints at different scales may be necessary but are not commonly used (Gaines et al. 2004). In this case, the stakeholders are the public who live near and hunt on and near the Department of Energy's (DOE) Savannah River Site (SRS; figure 4.1). To date, there is a dearth of knowledge concerning how environmental risk can be managed at the population level when using wildlife as endpoint (receptor) species.

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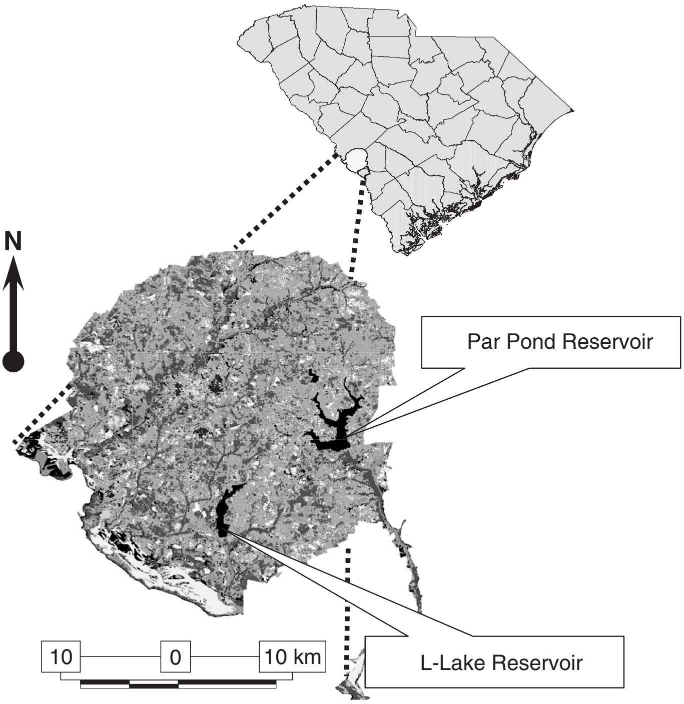

Figure 4.1. The Department of Energy's Savannah River site located on the west-central border of South Carolina along the Savannah River Swamp. This habitat map shows the distribution of the major drainage systems of the Savannah River site, including the former nuclear reactor cooling reservoirs, Par Pond and L-Lake (where raccoons are possible contaminant vectors), which are contaminated with ^{137}Cs.

Until recently, the landscape approach has been rarely used in ERAs when assessing wildlife receptor species, and especially on large federal facilities that would benefit from a landscape-level implementation. That is, contaminant exposure assessments have taken into account neither the spatial distribution of the pollutant nor the movements of groups of individuals over the landscape. Rather, fact gathering has remained biased toward lower levels of ecological organization, despite the acknowledged need for and relevance associated with information about effects at higher levels, for example, effects on higher trophic levels or populations (Taub 1989; Cairns 1996). Methods are rapidly changing due to the recognition that if a site is spatially heterogeneous with respect to either contamination or wildlife use, then models must be modified to include the dynamics imposed by those spatial constraints (Sample and Suter 1994). Although humans are often not considered a logical endpoint in an ERA, in many cases arguably they are the most appropriate. When considering the landscape structure of industrial sites such as the SRS (especially those that allow hunting) that are surrounded by rural areas, hunters are one of the main components influencing the population of many wildlife species and subsequently the structure of the ecosystem's food web. If hunters were not allowed to take game from these sites due to high consumption risks, it could have an impact on the population structure of the wildlife in those ecosystems and possibly contribute to new risks due to redistribution and movement of contaminants offsite. In this chapter, we describe how wildlife populations can be managed through harvest to minimize the bioavailability of toxicants in the environment for the SRS. We use raccoons as the focal species; however, many other ecologically important game species contribute to the environmental toxicodynamics of the SRS and similar landscape-level industrial sites that can also be managed to minimize both ecological and human risk. In the Southeast, raccoon hunting is extremely popular, and harvested individuals are used for both meat and fur (Gaines et al. 2000). In South Carolina, where the SRS is located, the raccoon-hunting season is usually from mid-September to mid-March, with no bag or possession limit. Thus, a diligent hunter who eats the meat could legally consume as much raccoon meat as desired.

Methods

Study species and area

The SRS is a 778-km^2 former DOE nuclear production and current research facility located in west-central South Carolina (33.1° N, 81.3° W; figure 4.1) that was closed to public access in 1952. On numerous occasions, both terrestrial and aquatic SRS ecosystems have been contaminated with radionuclides, metals, and organics, and areas have been affected by thermal effluents (White and Gaines 2001). In 1972, the entire SRS was designated as the nation's first National Environmental Research Park to provide tracts of land where the effects of human impacts upon the environment could be studied (Davis and Janecek 1997; White and Gaines 2001). Much of the suitable forested area of the SRS is managed primarily for commercial timber (pine) production by the U.S. Forest Service. More than 20% of the SRS is covered by wetlands, including bottomland hardwoods, cypress-tupelo swamp forests, creeks, streams, ponds, Carolina bays (which are natural elliptical depressions that vary in size and in the degree to which they retain water; Ross 1987), and two large former reactor cooling reservoirs (Par Pond, L-Lake) and associated floodplains and outflows that have been contaminated with the gamma-emitting radionuclide radiocesium (^{137}Cs). Hunting is allowed on the SRS proper. Both white-tailed deer (*Odocoileus virginianus*) and wild hogs (*Sus scrofa*) are hunted on site and are monitored for ^{137}Cs. Raccoons are hunted in the Crackerneck Wildlife Management Area located in the southwestern portion of the site, and individuals are not monitored for contaminant burden. Previous studies (authors' unpublished data) indicate that the raccoon population from Crackerneck Wildlife Management Area is not accumulating ^{137}Cs significantly above background levels. However, raccoons are hunted on the border of the SRS, and past studies (Gaines et al. 2000, Chow et al. 2005) have revealed elevated ^{137}Cs levels in individuals from populations that reside in contaminated areas on the SRS and along the SRS border. In this chapter, we focus on raccoon populations inhabiting two such border areas: the SRS border located west of the Par Pond reactor cooling reservoir, and the SRS border located south of the L-Lake reactor cooling reservoir along Steel Creek (figure 4.2). These two sites were chosen based on their juxtaposition relative to the SRS border, ^{137}Cs contamination, and the potential for harvest. Other border sites may also pose a risk of raccoons becoming contaminant vectors, but the toxicokinetics have not been well studied for those areas and therefore were not modeled.

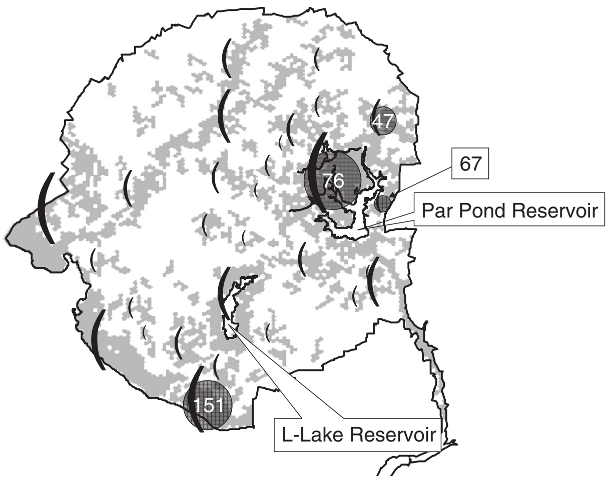

Figure 4.2. The population structure of Savannah River site raccoons as estimated by the RAMAS Metapop program (circles). A probability surface of raccoon distribution was used to determine suitable habitat for raccoon population establishment (light gray pixels). Only populations with $n > 5$ were used in the analysis. The populations that were harvested (population ID #s 47, 67, 76, and 151) in the simulations are labeled and shaded.

Previous investigations (Arbogast 1999; Boring 2001; Gaines et al. 2000, 2005) have shown elevated levels of ^{137}Cs in these raccoon populations and have documented that raccoons move freely on and off the SRS. Thus, the raccoon is a useful species for both human risk assessments and ERAs. Since this species is extremely mobile, the likelihood of an animal's presence in specific microhabitats and the time spent in those habitats must be estimated to calculate reasonable risk estimates. For this study, home range habitat utilization information coupled with three years of harvest data and ^{137}Cs monitoring on and near the SRS supplied the data needed to develop a spatially explicit model to investigate population-level toxicodynamics.

Wildlife species are used as endpoints in the risk assessment process by the DOE, and raccoons in particular are a focal species. Specifically, raccoons have been used as a receptor species in both human risk assessments and ERAs for the SRS and other DOE sites using current information regarding home range, contaminant uptake, and food habits for populations both on and off the SRS. Several life-history characteristics of raccoons make them potential agents of contaminant distribution, including (1) high population levels with an extended range throughout North America in a variety of habitats, (2) their ability and proclivity to travel extended distances (Glueck et al. 1988; Walker and Sunquist 1997; Gehrt and Fritzell 1998), (3) a propensity to utilize human-altered habitats in combination with an ability to move freely in and out of most toxic waste sites (Hoffmann and Gottschang 1977; Clark et al. 1989; Khan et al. 1995), and (4) a broadly omnivorous diet that includes components of both terrestrial and aquatic food chains (Lotze and Anderson 1979; Khan et al. 1995).

Experimental methods

Toxicokinetics

Radiocesium uptake models were constructed from information collected for male raccoons from three consecutive harvests in the L-Lake corridor located near the border of the SRS (figure 4.2) adjacent to a private hunting preserve. This population was used because individuals spent 100% of their time in contaminated areas (as determined from radiotelemetry). Mean ^{137}Cs levels declined significantly from the first trap effort to the third trap effort (year 1, 127 Bq/kg; year 2, 63 Bq/kg; year 3, 29 Bq/kg; all activities are reported for wet weight; for analytical counting methods, see Arbogast 1999; Gaines et al. 2000; Boring 2001). The first two trapping efforts (Arbogast 1999; Gaines et al. 2000) harvested individuals from the population each year (n = 10, spring 1997; n = 13, spring 1998). Areas were trapped until no more individuals were caught after an additional two-week period. Therefore, it is assumed that the sample size represents the population of male raccoons for the immediate area, thus providing a baseline for the determination of the amount of ^{137}Cs that new recruits will accumulate in one year (for a detailed discussion of the ^{137}Cs dynamics in this raccoon population, see Gaines et al. 2000, 2005). For the first trapping effort, muscle was removed from raccoons and analyzed for ^{137}Cs. For the second trap effort, both muscle and whole-body ^{137}Cs burdens were determined, and a simple linear regression was performed to determine their predictive relationship. For the third trap effort, whole-body ^{137}Cs burdens were determined for all captured raccoons (n = 14). The muscle concentration was estimated using the simple linear regression model developed from the second trap effort (wet-weight muscle concentration [Bq/kg] = 1.7041 × whole body [Bq/kg] + 3.1; $r^2 = 0.9617$; Arbogast 1999). The aforementioned declining rates from year to year were used to estimate the proper "average weight" parameter in the Stages submenu of the model procedure in the RAMAS Metapop program described below. The maximum observed burden of 1,000 Bq/kg was used to allow conservative risk calculations.

Population Model

Spatial structure A spatially explicit model of raccoon distribution for the SRS was developed using data from a raccoon radiotelemetry study and visualized with GIS. An inductive approach was employed to develop three submodels using the ecological requirements of raccoons studied in the following habitats: (1) man-made reservoirs, (2) bottomland hardwood/riverine systems, and (3) isolated wetland systems. Probabilistic resource selection functions were derived from logistic regression using habitat compositional data and landscape metrics (for further details, see Gaines et al. 2005). The final distribution model provides a spatially explicit probability (likelihood of being in an area) surface. This surface was used as the base map in the Spatial Data program. Specifically, the habitat suitability function used a probability threshold of 0.85 and a neighborhood distance of one 100 m × 100 m cell to derive the habitat suitability map used to determine the metapopulation spatial structure. Initial abundances for each population were calculated from the population density of raccoons based on the trapping efforts during and two years prior to the radiotelemetry study. Specifically, it was estimated that a maximum of eight individuals would occupy a 250-hectare (average home range) area (for further details, see Boring 2001). Therefore, the initial raccoon abundance was calculated based on the following equation:

$$\text{Initial abundance} = (\text{noc} \times 0.072)\ \text{ahs}, \tag{4.1}$$

where noc is the number of cells in the patch, ahs is the average habitat suitability, and 0.072 is the number of raccoons per cell. Populations with initial abundances of fewer

Table 4.1. Population parameters used for simulations of raccoon (*Procyon lotor*) populations residing in the Department of Energy's Savannah River site.

Metric	Juvenile	Yearlings	2–3 years	4+ years
Fecundity[a]	0	1.4	1.55	1.0
Survival	0.8	0.8	0.8	0.64
Relative dispersal	1.0	1.0	0	0
^{137}Cs muscle concentration (Bq/kg)[b]	1.0	1.0	1,000	1,000

[a] Based on Zeveloff (2002).
[b] Using the average weight parameter in the Stages menu of the Metapop program.

than five individuals were not considered to be viable populations and therefore were removed from further analysis.

Population parameters Since natural resources required by raccoons are abundant in the Southeast, and both male and female raccoons have overlapping home ranges on the SRS, an exponential growth to carrying capacity (ceiling) density dependence was used. Moreover, the sex structure of the population was assumed to be mixed since there is little differentiation between how males and females use SRS habitats (although females may disperse less). As such, to maximize harvest rates, the populations were modeled at carrying capacity (K) with a 10% standard deviation, thus producing a conservative estimate for risk assessment purposes. For the purposes of harvest management, only a two-stage model is needed, juveniles and adults, since only adults are harvested. However, a five-stage model (juveniles, yearlings, 2-year-olds, 3-year-olds, and 4+ years) was needed to properly estimate contaminant burden and relative dispersal. Specifically, the average weight parameter in the Stages submenu of the model procedure in the RAMAS Metapop program (Akçakaya 2005) was used as a proxy for the concentration of ^{137}Cs in raccoon muscle tissue. For the purposes of this simulation, it was assumed that juveniles and yearlings have little to no contaminant body burden. Although past studies have shown that juveniles may inherit a burden from their mother (von Zallinger and Tempel 1998) and that yearlings will attain approximately one-third of their maximal burden (Boring 2000), to ensure that new recruits have background burden (the Metapop program will not differentiate weights on a per population basis) based on the toxicokinetic model described above, both juveniles and yearlings were modeled with a muscle concentration of 1 Bq/kg. To apply a conservative estimate for risk assessment purposes, the absolute maximum burden observed in the wild (1,000 Bq/kg) plus 10% was simulated for the 2- to 4-year stages. This absorbed the potential error of setting all juveniles and yearlings to have negligible burden. The dispersal-distance function used was based on movement data collected during the radiotelemetry study. The model assumed that only juveniles and yearlings disperse, and that the dispersal rate is distance dependent, with a negative exponential slope distance factor of 2 and a maximum dispersal distance of 25 km (the average diameter of the SRS). A 20% coefficient of variation was used for simulating environmental stochasticity in dispersal; due to the small sample size used to determine some population parameters, a 40% coefficient of variation was applied for sampling error of N under harvest management. It was also assumed that fecundity and survival were correlated, but carrying capacity was independent for the correlation structure of the stochastically varying parameters within each population. Table 4.1 shows additional population parameters used to construct model matrices. Fecundity was calculated by dividing the estimated litter size by 2 multiplied by the survival rate (0.8), assuming postbreeding census.

The harvest management scheme simulated yearly hunting of focal populations near the two border sites (figure 4.2). The proportion of individuals harvested was between 0.1 and 0.9 at 0.1 increments for all populations at both locations to determine if the ^{137}Cs burden would decrease at the population level over time as the harvest rate increased. Only yearlings through 4+ years were harvested in the simulation. The harvest started in year 4 to allow cohorts to equilibrate. Since highly suitable raccoon habitat extends off the SRS along the Steel Creek corridor, the estimated population size for that area most likely was underestimated using equation 4.1; therefore, the initial population size was doubled based on the area of the Steel Creek corridor that extends off the SRS. This was not done for the simulation around Par Pond since the habitats abruptly stop at the SRS border and change to habitats much less suitable for raccoon populations. However, similar forested and wetland habitats do extend offsite, which could affect dispersal (see discussion of isotropic scenarios below). Simulations were conducted using 1,000 replications over 50 years. The final-stage abundances and harvest summary were exported from the Metapop program for populations from each area (L-Lake, Par Pond). Two major scenarios were performed. The first assumed that recruitment rates of border populations would be less based on an anisotropic (asymmetric) migration trajectory and therefore used the initial population structure calculated from the Metapop program. To calculate the body burdens, the harvest totals (e.g., total ^{137}Cs Bq/kg) were divided by the total number of individuals in the harvested cohorts from the previous year (t–1) multiplied by the harvest rate. However, since border populations could have isotropic recruitment of noncontaminated individuals from offsite, the second scenario adjusted the numbers of individuals in the focal populations to assume 100% recruitment of each stage after an initial harvest. This was achieved by dividing the body burdens determined in scenario 1 by the proportion of individuals that were not recruited for each cohort. We also assumed that the harvested populations would receive new individuals from every age class via dispersal. This phenomenon was seen on the SRS during studies performed by Gaines et al. (2000).

Results

For the first set of simulations (scenario 1), populations were not adjusted to maximize new recruits. That is, the total number of individuals within each harvested cohort was summed, multiplied by the harvest rate, and used as the denominator to divide the total ^{137}Cs muscle concentration for determination of the toxicant concentration on a per individual basis. This revealed that the projected ^{137}Cs muscle concentration (range: Par Pond, 521–805 Bq/kg; L-Lake, 669–758 Bq/kg) will increase, with harvest rate peaking at 60%, and then decline as harvest rates increase (table 4.2). Nevertheless, the mean body burden diminishes only slightly below the European Economic Community (1986) standard of 600 Bq/kg. However, the second set of simulations (scenario 2) reveals that if the numbers of new recruits were maximized and assumed to come from neighboring populations that had no burden (isotropic

Table 4.2. Mean ^{137}Cs muscle concentration (Bq/kg) and associated lower and upper standard deviation (LSD, USD) estimates over 50 years for raccoon populations associated with the contaminated reservoir areas on the border of the Savannah River Site.

	L-Lake			Par Pond		
Harvest (%)	LSD	Mean	USD	LSD	Mean	USD
10%	258	717	1176	239	539	929
20%	307	698	1089	292	613	935
30%	351	751	1151	349	688	1027
40%	275	695	1114	231	521	812
50%	239	689	1140	245	623	1003
60%	325	758	1191	409	805	1201
70%	279	710	1142	329	697	1065
80%	287	702	1117	337	693	1050
90%	283	669	1054	331	664	998

The simulation assumes anisotropic recruitment of uncontaminated individuals after harvest due to border effects (e.g., no offsite recruitment).

recruitment), then the mean ^{137}Cs muscle concentration declines as the proportion of harvest increases (range over 50-year simulation: L-Lake, 164–813 Bq/kg; Par Pond, 156–654 Bq/kg; figure 4.3). Using the European Economic Community standard as a precautionary limit, a yearly harvest rate of at least 30% and 20% for the L-Lake and Par Pond populations, respectively, would need to be implemented to minimize risk to humans who may consume these animals.

To explore sensitivities in the model, a range of values for the movement and density dependence were used that would most likely affect the number of new recruits into harvested populations. Changing these parameters only altered the final number of raccoons in those populations for both simulations. However, the proportions of individuals within populations remained constant for the entire metapopulation, which is the most influential parameter regarding toxicant body burden.

Discussion

Most risk assessments that focus on the effects of toxicants to wildlife at the population level are concerned with impacts to population size due to such factors as reduced fecundity, acute toxification, or shifts in species composition. The population risk assessment presented here focuses on how managed populations through harvest may reduce contaminant mobility. Further, most ERAs do not use humans as an end point, assuming that they are independent from the trophic system of the focal environment. However, if hunting is not allowed due to risk concerns, wildlife may contain higher burdens than if harvests were implemented. The ^{137}Cs dynamics of the SRS is a typical

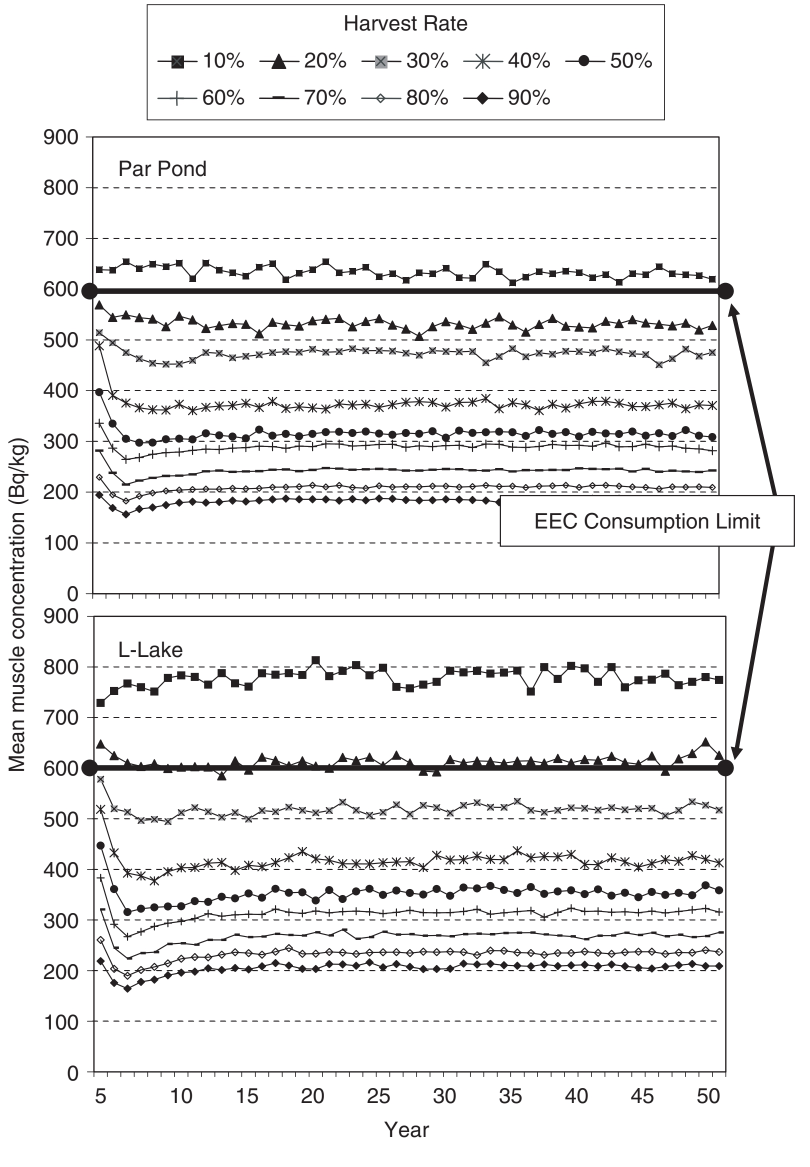

Figure 4.3. Mean ^{137}Cs muscle concentration (Bq/kg) estimates over 50 years for raccoon populations associated with the contaminated reservoir areas on the border of the Savannah River site (Par Pond, L-Lake). The simulation assumes maximum (isotropic) recruitment of uncontaminated individuals after harvest. The European Economic Community (EEC) standard of 600 Bq/kg is referenced as a precautionary limit, showing that a yearly harvest rate of at least 20–30% would need to be implemented to minimize risk to humans who may consume these animals.

example of how a coupled human–natural system drives ecological risk. Ecosystem dynamics control the ecological half-life of ^{137}Cs, while hunting in and around the SRS influences receptor species population dynamics and thus the bioavailability of ^{137}Cs to humans and other consumers, as well as contaminant transport within the ecosystem.

The premise of this model is that the most important components of estimating the environmental toxicokinetics of contaminants in wildlife populations are residence time of different age cohorts of organisms and the residence time of the contaminant in the environment. We used values from the literature best suited for the U.S. Southeast to estimate many of the population parameters. Cohort survival and dispersal are regionally distinct (Zeveloff 2002) and will affect toxicant exposure to populations through their residence time in the contaminated environment. The physical half-life of ^{137}Cs is approximately 30 years. The biological turnover rates within a given organism are generally much shorter and are influenced by metabolism. Therefore, biological half-life should change based on biotic and abiotic parameters such as age, overall health, seasonality, and food availability and also depends on the sources and bioavailability of the contaminants within the animal's home range. However, bioavailability in these systems has additional complexities. When radioactive isotopes are released into ecosystems such as those associated with L-Lake or Par Pond on the SRS, the isotopes will theoretically also have an ecological half-life. This is the amount of time required for the level of an isotope (in this case, ^{137}Cs), once established and at equilibrium within a given ecosystem compartment, to decrease by 50%. This is a result of the isotope either becoming ecologically unavailable or being physically removed from a system (Brisbin 1991). The concept of ecological half-life is further constrained by the fact that most ecosystem compartments are extremely dynamic and rarely come to equilibrium. As the time required to achieve effective equilibrium increases, it becomes less likely that these conditions will remain constant (Peters and Brisbin 1996). Therefore, having a relative estimate of the ecological half-life of the toxicant as it relates to the species uptake and depuration rate is essential in modeling contaminant mobility. That is, once these parameters are known, the spatiotemporal patterns can be better predicted.

Specifically, in this model, the residence times of age cohorts within contaminated areas were a direct function of both home range and dispersal dynamics. The home range dynamics of raccoons are known from the SRS (Boring 2001), and raccoons have proved to be useful sentinel species in these and other contaminated ecosystems (Bigler et al. 1975; Smith et al. 2003). It is how raccoons disperse from contaminated populations that drives the risk process. The effect of dispersal on the demography of local populations is dependent on whether the populations are connected in a metapopulation spatial arrangement (Johnson et al. 2005) and whether the relationship between any two populations is in the context of source or sink (Pulliam 1988). Although dispersal contributes to spatiotemporal variation in population size (Nichols et al. 2000), its relative importance, compared to local recruitment, in any subpopulation is unclear due to a lack of empirical data (Bennetts et al. 2001; MacDonald and Johnson 2001). Until recently (Cam et al. 2004), much of metapopulation theory has been based upon simulations and has lacked rigorous empirical testing (Hanski 2001; MacDonald and Johnson 2001). Specifically, the anisotropy of dispersal caused by edge effects when modeling has not been addressed adequately. Since a potential study site boundary may or may not correlate to habitat boundaries, the effect on dispersal rates into peripheral populations can be greatly in error. That is especially why these effects were addressed in this study since peripheral subpopulations are important in terms of management considerations.

Two major simulations were performed due to the above considerations: One scenario assumed that there was no recruitment from offsite populations after harvest, and the other scenario assumed that "clean" raccoons from offsite would maintain these border populations at levels close to carrying capacity. High-quality raccoon habitat on the eastern border of the SRS near Par Pond does not continue offsite, which would make the former scenario more likely due to population isolation. However, suitable habitat does exist nearby, so both management scenarios should be considered. Conversely, since the southern border of the SRS where the L-Lake population resides has contiguous habitat that persists offsite as well as noncontiguous patches much like the SRS interior, the latter scenario is more biologically plausible. Additionally, previous investigations (Boring 2001; Gaines et al. 2005) have demonstrated that new recruits do move into the L-Lake border population after harvest and that the ecological half-life is at least twice the biological half-life. Specifically, based on this estimate, new recruits should become equilibrated within at least two years, which is why juveniles and yearlings were assumed to have no contaminant burden, and raccoons two or more years of age were assumed to have ^{137}Cs muscle concentration of 1,000 Bq/kg (the maximum observed in SRS raccoons). Federal facilities such as the DOE's SRS have sufficiently characterized the abiotic and biotic systems within their boundaries; therefore, in this case, understanding how ecosystem components offsite influence wildlife movement was more critical. To maintain isotropy for simulations assuming maximum recruitment, we used the population structure and trajectory of populations located on the interior of the SRS. Other population parameters used for this model are congruent with other studies investigating raccoon movement (Broadfoot et al. 2001).

Metapopulation dynamics can be extremely important in long-term management scenarios for species of human concern. Source–sink dynamics may influence the management of harvested (McCoy et al. 2005) or endangered (Kauffman et al. 2004) species or may be created by the harvest or management itself (Novaro et al. 2005). While studies are beginning to link metapopulation dynamics with physical resource transport between subpopulations as reciprocal food web subsidies (Nakano and Murakami 2001), the use of such models for contaminant transport studies within and between ecosystems (including human ecosystems) is a novel aspect of this study.

The simulations showed that recruitment type (local vs. dispersal) affects a population's estimated body burden. If clean individuals are recruited and harvested before they reach equilibrium with contaminants in the environment, then contaminant mobility into humans and other predators is truly minimized. Based on the harvest simulations, at least a 20% and 30% yearly harvest rate for Par Pond and L-Lake, respectively, would be required before the muscle concentrations would fall below the European Economic Community limit of 600 Bq/kg (figure 4.3). Managed hunting near L-Lake and along the adjacent SRS border is plausible since a private hunting preserve is already established outside but adjacent to the SRS. Since L-Lake most likely has new recruits coming from "clean" populations offsite, this would be an optimum management action to minimize the risk of contaminant movement into the human food chain.

Conversely, if edge populations have limited recruitment due to anisotropic migration, then the survivors in each cohort would more likely be original residents of the contaminated areas and thus have a higher burden. This is most likely the case for the Par Pond population. If hunting would continue in that area, then harvest success would be reduced due to lower population size; however, those individuals harvested would contain the highest burdens. Since the Par Pond SRS border is commercially owned, offsite hunting is unlikely to be encouraged as a management option. Given that the SRS is concerned with raccoons potentially moving contaminants offsite, the best risk management option would be to implement an onsite hunt to lower the population even though this would not reduce toxicant burden. However, caution should always be used in this type of scenario since reducing population size as a long-term management goal may have consequences on ecosystem integrity and function.

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