Environmental Toxicology and Chemistry, Vol. 20, No. 9, pp. 1927–1935, 2001
© 2001 SETAC
Printed in the USA
0730-7268/01 $9.00 + .00

Environmental Toxicology

ACCUMULATION OF ^{137}CESIUM AND ^{90}STRONTIUM FROM ABIOTIC AND BIOTIC SOURCES IN RODENTS AT CHORNOBYL, UKRAINE

RONALD K. CHESSER,*†‡ BRENDA E. RODGERS,§ JEFFREY K. WICKLIFFE,§ SERGEI GASCHAK,‡
IGOR CHIZHEVSKY,‡ CARLETON J. PHILLIPS,§ and ROBERT J. BAKER‡§
†Department of Genetics, University of Georgia, Athens, Georgia 30602, USA
‡International Radioecology Laboratory, Slavutych, Ukraine
§Department of Biological Sciences, Texas Tech University, Lubbock, Texas 79409, USA

(*Received* 19 *July* 2000; *Accepted* 21 *December* 2000)

Abstract—Bank voles (*Clethrionomys glareolus*) and laboratory strains of house mice (*Mus musculus* BALB and C57BL) were relocated into enclosures in a highly contaminated area of the Red Forest near the Chornobyl (Ukraine) Reactor 4 to evaluate the uptake rates of ^{137}Cs and ^{90}Sr from abiotic sources. Mice were provided with uncontaminated food supplies, ensuring that uptake of radionuclides was through soil ingestion, inhalation, or water. Mice were sampled before introduction and were reanalyzed every 10 d for ^{137}Cs uptake. Levels of ^{90}Sr were assessed in subsamples from the native populations and in experimental animals at the termination of the study. Uptake rates in house mice were greater than those in voles for both ^{137}Cs and ^{90}Sr. Daily uptake rates in house mice were estimated at 2.72×10^{12} unstable atoms per gram (whole body) for ^{137}Cs and 4.04×10^{10} unstable atoms per gram for ^{90}Sr. Comparable rates in voles were 2.26×10^{11} unstable atoms per gram for ^{137}Cs and 1.94×10^{10} unstable atoms per gram for ^{90}Sr. By comparing values from voles in the enclosures to those from wild voles caught within 50 m of the enclosures, it was estimated that only 8.5% of ^{137}Cs was incorporated from abiotic sources, leaving 91.5% being incorporated by uptake from biotic materials. The fraction of ^{90}Sr uptake from abiotic sources was at least 66.7% (and was probably much higher). Accumulated whole-body doses during the enclosure periods were estimated as 174 mGy from intramuscular ^{137}Cs and 68 mGy by skeletal ^{90}Sr in house mice over 40 d and 98 mGy from ^{137}Cs and 19 mGy from ^{90}Sr in voles over 30 d. Thus, uptake of radionuclides from abiotic materials in the Red Forest at Chornobyl is an important source of internal contamination.

Keywords—Radiation Chornobyl Chernobyl Dose Rodents

INTRODUCTION

The explosion and subsequent fire at the Chornobyl nuclear facility on April 26 through May 6, 1986, resulted in the atmospheric dispersion of vast quantities of highly radioactive particulate matter [1]. Approximately three to six exobecquerels (EBq) were released into the environment [2,3] and distributed across eastern and northern Europe [4]. Complex particles associated with the nuclear fuel inventory of Reactor 4, however, were scattered into the habitats immediately surrounding the reactor facility [5]. In the regions of the Red Forest and Glyboke Lake (Ukraine), heavy depositions of radioactive ash on trees and water bodies resulted in the death of more than 400 ha of pine forest [6]. Presently, less than 2% of the radionuclide activity originally released by the Chornobyl accident remains. The remaining fraction is comprised of dangerous levels of ^{137}Cs and ^{90}Sr, which have considerable biological affinity and long half-lives (30.1 and 28.2 years, respectively).

Physical and chemical transformation of the radionuclide material has led to the incorporation of high concentrations of ^{137}Cs and ^{90}Sr into the biota within the exclusion zone, which is an area with a 30-km radius around the Chornobyl nuclear power plant [7]. Small mammals from the Red Forest, approximately 2 km west/southwest of Reactor 4, averaged more than 13,000 nuclear disintegrations per second (Becquerels [Bq]) of ^{137}Cs in each gram of muscle tissue. This activity results in an average dose rate of almost 10 mGy/d (1 mGy = 0.1 rad), or 10-fold the maximum guidelines established by the International Atomic Energy Agency (Vienna, Austria) [8]. Although sterility resulting from chronic doses of 1 mGy/d or more has been reported [8,9], no evidence has been found for reproductive inhibition in mammals from the Red Forest [7,10,11]. In fact, the density of mammals in the exclusion zone is comparable to that in areas with relatively low radioactive contamination only 30 km from the reactor [12]. In previous studies of native rodent species, the bank vole (*Clethrionomys glareolus*) consistently showed the highest specific activity of intramuscular ^{137}Cs, averaging almost 25 kBq/g, which yields a dose rate of approximately 18 mGy/d [7,13].

The primary route for incorporation of radionuclides into the mammals inhabiting the Chornobyl exclusion zone likely is biotic transfer. Radiocesium is a biological analogue of potassium, whereas radiostrontium is an analogue of calcium. Concentrations of radionuclides remain substantially higher in the soils and sediments (i.e., abiotic concentrations) than the concentrations that are transferred into the biotic community. Thus, ^{137}Cs and ^{90}Sr will be absorbed from the soil through plant root systems and incorporated into cells, and ingestion of plant material will convey radioactive material to herbivores. Typically, ^{137}Cs and ^{90}Sr in the biotic portion of a contaminated ecosystem will be less than 1% of that in the abiotic fraction [14–16]. Small mammals often construct underground burrows and nests and, therefore, surely ingest and inhale soil particles containing radionuclides. Clearly, uptake of free particulate matter by inhalation, drinking, and ingestion of soils may be an important avenue for loading mammalian tissues with radionuclides. The potential for ingestion of soil deposited

---

* To whom correspondence may be addressed (rchesser@ttu.edu).

on leaves by resuspension [17–19] was reported in the Chornobyl region [17], but the rate of transfer to herbivores was not investigated. Abiotic avenues of radionuclide uptake, therefore, may be significant contributors to dose rates of animals living in contaminated environments [20,21].

Because of the disparity in the concentrations of ^{137}Cs and ^{90}Sr in the biotic and abiotic components of the contaminated habitats, it is important to discern the degree to which each may contribute to the absorbed dose and the ultimate biological risk. This study was designed to estimate the uptake of ^{137}Cs and ^{90}Sr from biotic and abiotic materials in a highly contaminated area of the Red Forest. The separation of these variables was accomplished by comparing native animals living in this region to those transported from relatively uncontaminated areas and placed into enclosures built in the Red Forest. The latter were fed with uncontaminated food, thus maximizing their uptake through inhalation, drinking, and ingestion of abiotic radionuclide particles. The enclosure study was conducted using the bank vole (*C. glareolus*), which is native to the region, and laboratory strains of the house mouse (*Mus musculus*).

MATERIALS AND METHODS

Previous research has revealed that the most radioactive mammals (*C. glareolus*) are consistently captured from a specific region in the Red Forest (UTM: 36 295545 U5697040) [7], located approximately 2.0 km west/southwest of the Chornobyl nuclear power plant. We selected this site to place 11 enclosures to accommodate mice for relocation studies. The enclosures measured approximately 1 × 0.67 × 0.67 m and were covered with aluminum mesh fencing (opening size, ~1 cm) stretched over a wooden frame. Hinged tops were fastened with bolts and wing nuts, which allowed periodic access to the animals inside. Each enclosure was filled with approximately 30 cm of soil from the placement site. All 11 enclosures were placed within a 30- × 20-m area of the Red Forest. Although some vegetation was growing within the cages before introduction of the mice, constant disruptions to capture the animals eliminated most of the vegetation. Previous efforts to rear mice in the enclosures using natural vegetation were unsuccessful in all instances. No evidence was found that the vegetation in the cages was consumed.

Clethrionomys glareolus ($n = 11$) were captured using Sherman aluminum live traps (H.B. Sherman, Tallahassee, FL, USA) in the forest south of the village of Nedanchichy, Ukraine (UTM: 36 336455 U5707606). This region received little fallout from the Chornobyl accident, and intramuscular ^{137}Cs levels in the mice were presumed to be low [22]. All mice were individually marked with a unique toe clip. Male–female pairs were placed into five separate cages, and a single pregnant female was placed into a sixth enclosure in the Red Forest. Additional *C. glareolus* ($n = 27$) were captured within 50 m of the enclosures. These animals were sacrificed to determine the intramuscular ^{137}Cs and skeletal ^{90}Sr activities. Laboratory strains of domestic house mice (*Mus musculus*: BALB, $n = 8$; C57BL, $n = 8$) were purchased from a research breeding colony in the city of Chornobyl. These mice were individually marked by a unique toe clip and placed into five separate enclosures in the Red Forest.

Uncontaminated vegetables, dried bread, and rat chow were liberally provided to each cage every 2 to 3 d. Uncontaminated food supplies ensured that uptake of ^{137}Cs and ^{90}Sr occurred via drinking, ingestion, or inhalation of particulate matter from the soil. All mice were subjected to a whole-body determination of ^{137}Cs levels before their release into the enclosures. Radiocesium readings were performed by counting each mouse for 3 to 10 min (depending on activity level) using a 7.6-cm NaI crystal contained in a lead housing (Canberra Industries, Meridian, CT, USA). Although longer reading times would have provided more accurate estimates of activity for the initial reading, efforts were made to reduce the trauma of confinement in a 4- × 6-cm cylindrical scintillation jar. The counting methods, calibration, and standardization were as described by Chesser et al. [7], except that whole-body counts rather than muscle counts were performed. Background assessments were made after every third whole-body count.

The mice were recaptured, usually by excavating their burrow systems and subterranean nests, every 10 d. Mice were then placed in aluminum live traps, supplied with food and nesting material, and transported to the laboratory, where whole-body counts and body-mass measurements were obtained and blood samples were drawn. Animals were returned to the enclosures on the same day as their capture. *Mus musculus* were resampled four times in addition to the initial period, whereas *C. glareolus* pairs were resampled for three time periods in addition to the initial screening. The pregnant female *C. glareolus* was only resampled once, at the end of the 30-d period. At the termination of the study, all mice were sacrificed, and bone and muscle samples were taken. The activity of ^{90}Sr was determined from bone and soil, but ^{137}Cs from muscle and soil was measured using methods described in previous publications [7,23]. All radiometric measures in this manuscript are reported for wet weights of muscle, bone, or whole body.

Physical decay and biological losses

Rates of loss of radionuclides are not known for *Clethrionomys* and *Mus* sp. However, the biological half-life of radionuclides has been summarized for a variety of species, with a consistent relationship being found between retention half-time (T) and body mass (w) [24,25]. For ^{137}Cs and ^{90}Sr, these half-times were $T = 3.5w^{0.24}$ and $T = 107w^{0.26}$, respectively. Therefore, the biological half-life of ^{137}Cs in rodents is approximately 7.2 d, whereas that for ^{90}Sr is approximately 233 d [24,25]. Using these values, the daily loss rates (λ_B) of ^{137}Cs and ^{90}Sr by way of biological processes was calculated using

$$q = q_0 e^{\lambda_B t} \tag{1}$$

where q is the concentration of a radionuclide remaining within the animal at some time t and q_0 is the initial concentration of the radionuclide. Assuming that no further uptake of the radionuclide occurs, Δq will be affected only by physical and biological losses. After one half-life, the value of q will equal $q_0/2$. Therefore, the rates of loss by biological processes are

$$\lambda_{B(Cs)} = \frac{\ln(2)}{7.2} = 0.09627\text{d}^{-1}$$

$$\lambda_{B(Sr)} = \frac{\ln(2)}{233} = 2.975 \times 10^{-3}\text{d}^{-1} \tag{2}$$

Similarly, the physical half-life of ^{137}Cs and ^{90}Sr are 10,982 and 10,585 d, respectively, yielding daily decay rate constants (λ_D) of

$$\lambda_{D(Cs)} = \frac{\ln(2)}{10981.88} = 6.31 \times 10^{-5}\text{d}^{-1}$$

$$\lambda_{D(Sr)} = \frac{\ln(2)}{10585} = 6.55 \times 10^{-5}\text{d}^{-1} \qquad (3)$$

Combining the biological and the decay constants, the daily losses of radionuclides from the body of a rodent are defined by the rates

$$\lambda_{Cs} = \lambda_{B(Cs)} + \lambda_{D(Cs)} = 0.09633\text{d}^{-1}$$

$$\lambda_{Sr} = \lambda_{B(Sr)} + \lambda_{D(Sr)} = 3.04 \times 10^{-3}\text{d}^{-1} \qquad (4)$$

For ^{137}Cs and ^{90}Sr within the body of a rodent, the majority of unstable atoms ingested will be lost through biological depuration before physical decay. Therefore, most of the potentially radioactive material will pass through the body without conveying a dose. For ^{137}Cs and ^{90}Sr, the percentages of ingested, unstable atoms lost before undergoing physical decay are 99.94 and 97.8%, respectively ($\% = \lambda_B/[\lambda_B + \lambda_D]$).

Radionuclide activity and concentration values

Concentration values for ^{137}Cs and ^{90}Sr were determined using the specific activities as measured in Becquerels per gram (Bq g^{-1}), or the number of nuclear disintegrations per second per gram. Traditionally, only specific activity is reported in radioecologic studies. However, concentration values provide greater comparability with other investigations in toxicology, and they also facilitate the determination of projected doses as well as uptake and depuration rates. Specific quantities derived were the number of unstable atoms per gram of tissue (A), the total number of unstable atoms per individual (B), the number of grams of a radionuclide per gram of sample (C), and the parts per million of the radionuclide in the sample (*ppm* = micrograms of radionuclide per gram of rodent). These quantities were calculated as

$$A = \frac{Bq\ \text{g}^{-1}\ 86{,}400_{\text{s/d}}}{\lambda_D}$$

$$\left\{\begin{array}{l}\text{86,400 seconds per day}\\ \lambda_D = \text{daily decay constant (see above)}\end{array}\right\}$$

$$B = Aw \qquad \{w = \text{body mass, in grams}\}$$

$$C = \frac{Aa}{6.022045 \times 10^{23}}$$

$$\left\{\begin{array}{l}a = \text{atomic mass; Cs} = 55,\ \text{Sr} = 38\\ 6.022045 \times 10^{23} = \text{Avagadro's no.}\end{array}\right\}$$

$$ppm = C \times 10^6$$

Accumulation of unstable atoms

For plants and animals living in radioactively contaminated environments, the rates of change in the number of unstable atoms (A) of ^{137}Cs and ^{90}Sr are controlled by the uptake and loss rates via decay and biological processes. The following expressions were derived, denoting u as the rate of uptake of unstable atoms per gram of body mass, because uptake rates likely are a function of the size of the animal. The rate of change in the number of unstable atoms in each gram of tissue can be presented as a differential equation (the subscript accompanying λ to designate the radionuclide type has been dropped and can apply to any radionuclide):

$$\frac{dA}{dt} = u - \lambda A \qquad \{\lambda = \lambda_D + \lambda_B\} \qquad (5)$$

which, after integration, becomes

$$Ae^{\lambda t} = \int ue^{\lambda t}\,dt = \frac{u}{\lambda}e^{\lambda t} + K \qquad \{K = \text{constant}\}$$

$$A = \frac{u}{\lambda} + Ke^{-\lambda t} \qquad \left\{\begin{array}{l}t = 0 \rightarrow A_0 = 0,\\ K = A_0 - \dfrac{u}{\lambda}\end{array}\right\}$$

$$A = \frac{u}{\lambda}(1 - e^{-\lambda t}) + A_0e^{-\lambda t} \qquad (6)$$

At the limit ($t \rightarrow \infty$), this equation converges on

$$\lim_{t\rightarrow\infty} A = \frac{u}{\lambda} \qquad (7)$$

showing that concentrations of unstable atoms within the rodents are expected to approach constant (i.e., asymptotic) values over time. The right side of Equation 6 is made of two parts. The first represents the proportion of the ultimate asymptotic concentration of unstable atoms that has been attained. The second is the rate of loss of the initial concentration of unstable atoms. By rearranging the result of Equation 6, the uptake rates and asymptotic concentrations for a radionuclide can be estimated as

$$u = \frac{A - A_0e^{-\lambda t}}{1 - e^{-\lambda t}}\lambda \qquad \hat{A} = \frac{u}{\lambda} = \frac{A - A_0e^{-\lambda t}}{1 - e^{-\lambda t}} \qquad (8)$$

Because the values of λ, A_0, and A are known or empirically estimated for both ^{137}Cs and ^{90}Sr in mice from the enclosures, the above quantities were readily computed.

Accumulated dose calculations

The accumulated dose is a function of the number of the unstable atoms per gram that undergo physical decay (λ_D) within the body of the mice. Restructuring the equations described by Chesser et al. [7] shows that the absorbed dose ($\Re$, in mGy) per gram of tissue over the total time period is determined by

$$\Re = (\text{disintegrations per gram})\bar{E}_{\text{MeV/Dis.}}(1.6 \times 10^{-6}_{\text{erg/MeV}})\frac{1_{\text{mGy}}}{10_{\text{erg/g}}}$$

$$= (\text{disintegrations per gram})\bar{E}(1.6 \times 10^{-7})_{\text{mGy}} \qquad (9)$$

where $\bar{E}$ is the average energy (in MeV) released by each nuclear disintegration that is absorbed into the tissues. The rate of dose accumulation can be represented by the differential equation

$$\frac{d\Re}{dt} = \lambda_D E(1.6 \times 10^{-7})A \qquad (10)$$

where A is the number of unstable atoms remaining per gram of tissue. However, the primary interest was to calculate the doses caused by the uptake of radionuclides for the period after the mice were introduced into the enclosures. This quantity, $A - A_0e^{-\lambda t}$, will exclude dose due to the initial concentration of unstable atoms. Using this modification to the value for A derived from Equation 6, the above equation can be expanded and resolved as

Table 1. Activities and concentration values for ^{137}Cs in the bank vole (*Clethrionomys glareolus*) and house mouse (*Mus musculus*) relocated from uncontaminated environments to enclosures in the Red Forest near the Chornobyl (Ukraine) reactor

Activities and concentrations[a]	Becquerels per gram (Bq g^{-1})	No. of unstable atoms per gram (A)	No. of unstable atoms per individual (B)	Grams of ^{137}Cs per gram of mouse (C)	Parts per million of ^{137}Cs
Period 0					
C. glareolus	43.64	5.98×10^{10}	1.24×10^{12}	5.46×10^{-12}	5.46×10^{-6}
$n = 11$	(29.4)	(4.01×10^{10})	(8.29×10^{11})	(3.68×10^{-12})	(3.68×10^{-6})
M. musculus	153.32	2.10×10^{11}	3.85×10^{12}	1.92×10^{-11}	1.92×10^{-5}
$n = 16$	(98.6)	(1.35×10^{11})	(1.95×10^{12})	(1.23×10^{-11})	(1.23×10^{-5})
Period 1					
C. glareolus	1,079.85	1.48×10^{12}	3.25×10^{13}	1.35×10^{-10}	1.35×10^{-4}
$n = 11$	(276.6)	(3.80×10^{11})	(7.30×10^{12})	(3.45×10^{-11})	(3.45×10^{-5})
M. musculus	1,863.61	2.55×10^{12}	5.92×10^{13}	2.33×10^{-10}	2.33×10^{-4}
$n = 16$	(278.3)	(3.80×10^{11})	(4.79×10^{12})	(3.50×10^{-11})	(3.50×10^{-5})
Period 2					
C. glareolus	1,594.89	2.18×10^{12}	4.94×10^{13}	1.99×10^{-10}	1.99×10^{-4}
$n = 10$	(430.7)	(5.91×10^{11})	(1.28×10^{13})	(5.38×10^{-11})	(5.38×10^{-5})
M. musculus	1,767.24	2.42×10^{12}	6.40×10^{13}	2.21×10^{-11}	2.21×10^{-4}
$n = 16$	(228.3)	(3.13×10^{11})	(1.04×10^{13})	(2.85×10^{-11})	(2.85×10^{-5})
Period 3					
C. glareolus	1,570.56	2.15×10^{12}	4.96×10^{13}	1.96×10^{-10}	1.96×10^{-4}
$n = 9$	(339.3)	(4.63×10^{11})	(1.22×10^{13})	(4.23×10^{-11})	(4.23×10^{-5})
M. musculus	2,327.90	3.18×10^{12}	8.20×10^{13}	2.91×10^{-10}	2.91×10^{-4}
$n = 16$	(259.5)	(3.55×10^{11})	(1.05×10^{13})	(3.25×10^{-11})	(3.25×10^{-5})
Period 4					
C. glareolus	—	—	—	—	—
M. musculus	1,810.45	2.47×10^{12}	5.68×10^{13}	2.26×10^{-10}	2.26×10^{-4}
$n = 7$	(239.0)	(3.27×10^{11})	(7.98×10^{12})	(2.99×10^{-11})	(2.99×10^{-5})

[a] From whole-body measurements on live mice. The sample periods were approximately 10 d apart. Standard errors of concentrations were estimated from the standard error of specific activity (Bq g^{-1}) and are in parentheses.

$$\frac{d\Re}{dt} = \frac{\bar{E}(1.6 \times 10^{-7})u\lambda_D}{\lambda}(1 - e^{-\lambda t})$$

$$\int \frac{1}{\bar{E}(1.6 \times 10^{-7})u\lambda_D}\, d\Re = \int (\lambda^{-1}(1 - e^{-\lambda t}))\, dt$$

$$\Re = \bar{E}(1.6 \times 10^{-7})u\lambda_D\left[\frac{e^{-\lambda t} + t\lambda}{\lambda^2} + K\right]$$

$$\{K = \text{constant}\} \qquad (11)$$

At $t = 0$, $\Re = 0$, enabling the constant K to be resolved as $K = -\lambda^{-2}$. Using this solution, the accumulated dose over t days is

$$\Re_{\text{mGy}} = u\bar{E}(1.6 \times 10^{-7})\lambda_D\left(\frac{e^{-\lambda t} + t\lambda - 1}{\lambda^2}\right) \qquad (12)$$

Using the average absorbed energy per disintegration of ^{137}Cs for a rodent (0.21 MeV) [7] and substituting the physical and biological decay coefficients for ^{137}Cs, the accumulated dose over t days becomes

$$\Re_{\text{Cs}(t)} = u[2.14 \times 10^{-11}t - (1.946 \times 10^{-10})(1 - e^{-0.0991t})] \qquad (13)$$

The absorbed energy from ^{90}Sr decay (and its decay product, ^{90}Y) was estimated by Chesser et al. [7] to be 0.23 MeV. Therefore, the accumulated dose from skeletal ^{90}Sr over t days is

$$\Re_{\text{Sr}(t)} = u(2.63 \times 10^{-7})(e^{-0.00303t} + 0.00303t - 1) \qquad (14)$$

Daily dose rate calculations

For mice relocated into the enclosures, the daily dose rate (mGy d^{-1}) from ^{137}Cs after the asymptotic value for A is attained can be estimated as [7]:

$$\hat{\Re}_{\text{Csd}^{-1}} = \hat{A}\lambda_D\bar{E}(1.6 \times 10^{-7}) = \hat{A}(2.12 \times 10^{-12}) \qquad (15)$$

Because equilibrium is attained rapidly (~20 d) for ^{137}Cs in rodents, wild *C. glareolus* captured in the immediate vicinity of the cages in the Red Forest were assumed to have already attained equilibrium concentrations for ^{137}Cs. For those mice, the daily dose rates were calculated by modifying an equation described by Chesser et al. [7] that was derived for measurements in Bq g^{-1} to apply to wet weight measurements:

$$\Re_{\text{Cs(wild) d}^{-1}} = \text{Bq g}^{-1}(2.896 \times 10^{-3})_{\text{mGy d}^{-1}} \qquad (16)$$

Comparison of the daily dose rates for caged mice and wild-caught mice enabled estimation of the fractions contributed by particulate uptake or via ingestion of ^{137}Cs through the food chain. Daily dose rates for the observed concentrations of ^{90}Sr at the time of capture of wild *C. glareolus* could be estimated from empirical measurements of Bq g^{-1} [7]:

$$\Re_{\text{Sr/d}} = \text{Bq g}^{-1}(3.5 \times 10^{-3})_{\text{mGy d}^{-1}} \qquad (17)$$

RESULTS

Empirical values for intramuscular ^{137}Cs specific activity and concentrations for mice at the initiation of the enclosure study and at the subsequent resampling periods are presented in Table 1. The accumulation of ^{137}Cs was consistently, but not significantly, higher in *Mus* than in *Clethrionomys* sp. At the end of the 30-d period, *Clethrionomys* averaged 1,570 Bq g^{-1}, whereas *Mus* measured 1,810 Bq g^{-1} after a 40-d period. Initial values were significantly lower than those in subsequent time periods for both species ($p < 0.05$). The distributions of ^{137}Cs specific activity were not log-normally distributed, unlike data collected on native rodents [7]. Distributions did not deviate from normality in sampling periods 0, 1, and 2 in *Cleth-*

Table 2. Activities, concentrations, and daily whole-body dose rates for ^{90}Sr in *Clethrionomys glareolus* and *Mus musculus* sampled from the enclosures and from wild *C. glareolus* captured in the vicinity of the enclosures in the Red Forest near the Chornobyl reactor (Ukraine)[a]

Activities and concentrations	Becquerels per gram bone	No. of unstable atoms per gram	Grams ^{90}Sr per gram bone	Parts per million (µg/g)	Daily dose rate (mGy d^{-1})
Period 0					
C. glareolus	1,448.0	1.91×10^{12}	1.21×10^{-10}	1.21×10^{-4}	5.07
$n = 8$	(1,022.0)	(1.4×10^{12})	(8.51×10^{-11})	(1.40×10^{-3})	(3.58)
M. musculus	112.0	1.48×10^{11}	9.32×10^{-12}	9.32×10^{-6}	0.392
$n = 3$	(59.1)	(7.8×10^{10})	(4.92×10^{-12})	(9.20×10^{-5})	(0.21)
Enclosures					
C. glareolus	421.4	5.56×10^{11}	3.51×10^{-11}	3.51×10^{-5}	1.48
$n = 9$	(263.4)	(3.48×10^{11})	(2.19×10^{-11})	(3.80×10^{-4})	(0.92)
Period 3	[1,743.2	$[2.30 \times 10^{12}$	$[1.45 \times 10^{-10}$	$[1.45 \times 10^{-4}$	[6.10
	(457.4)]	$(6.03 \times 10^{11})]$	$(3.81 \times 10^{-11})]$	$(1.05 \times 10^{-3})]$	(1.60)]
M. musculus	1,148.6	1.52×10^{12}	9.56×10^{-11}	9.56×10^{-5}	4.02
$n = 11$	(291.6)	(3.84×10^{11})	(2.43×10^{-11})	(6.82×10^{-4})	(1.02)
Period 4	[1,247.80	$[1.65 \times 10^{12}$	$[1.04 \times 10^{-10}$	$[1.04 \times 10^{-4}$	[4.37
	(343.9)]	$(4.54 \times 10^{11})]$	$(2.86 \times 10^{-11})]$	$(7.63 \times 10^{-4})]$	(1.20)]
Wild-caught					
C. glareolus	7,249.0	9.56×10^{12}	6.03×10^{-10}	6.03×10^{-4}	25.37
$n = 17$	(860.9)	(1.14×10^{12})	(7.16×10^{-11})	(3.49×10^{-3})	(3.01)

[a] For wet weight of bone. The *C. glareolus* and *M. musculus* were sampled after 30 and 40 d in the enclosures, respectively. Standard errors (in parentheses) were estimated from the standard error for specific activity (Bq g^{-1}). Values in brackets include initial concentrations of ^{90}Sr in mice before introduction into the enclosures.

rionomys or in periods 0 through 4 in *Mus*. The specific activities were due to the physical decay of a fraction of an average of unstable atoms of ^{137}Cs per gram of tissue, averaging 2.15×10^{12} unstable atoms in *Clethrionomys* and 2.47×10^{12} unstable atoms in *Mus*. Because animals were expected to attain 90% of their asymptotic value for ^{137}Cs within 20 d and no significant differences were found in specific activity for sample periods 1 through 4, empirical asymptotic values for A were estimated by averaging the number of unstable atoms per gram on the values attained in all time periods subsequent to the first. The asymptotic estimates of $\hat{A}$ were 2.17×10^{12} for *Clethrionomys* and 2.74×10^{12} for *Mus*. These values were used to calculate the uptake rates, u, of unstable ^{137}Cs atoms per gram per day as 2.09×10^{11} for *Clethrionomys* and 2.64×10^{11} for *Mus*.

Concentrations, specific activities, and daily dose rate estimates for ^{90}Sr in *Clethrionomys* and *Mus* at the initiation (sample period 0) and termination (sample period 3 or 4) of the enclosure study as well as for wild-caught *Clethrionomys* are reported in Table 2. As for ^{137}Cs, the ^{90}Sr distributions of Bq g^{-1} did not deviate from normal. Specific activities for the unexposed animals were significantly lower than those measured in the enclosure mice at the termination of the study ($p < 0.05$). When the measures were adjusted for the initial concentrations, the uptake rate for *Mus* was, again, considerably higher than that for *Clethrionomys*. Although *Clethrionomys* had higher ultimate concentrations than *Mus*, this discrepancy resulted because *C. glareolus* from Nedanchichy had surprisingly high levels of ^{90}Sr, and because approximately 91% of those original concentrations remained after the 30-d enclosure study. Estimated uptake rates in the enclosures were 1.94×10^{10} ^{90}Sr atoms per gram for *Clethrionomys* and 4.04×10^{10} atoms per gram for *Mus*. Thus, the ultimate asymptotic values were estimated as 6.38×10^{12} and 1.33×10^{13} ^{90}Sr atoms per gram in *Clethrionomys* and *Mus*, respectively. Wild-caught *Clethrionomys* contained significantly ($p < 0.05$) greater ^{90}Sr activity than animals in the enclosures, with almost 7,250 Bq g^{-1} of bone. Daily dose rates due to skeletal ^{90}Sr in wild-caught mice from the vicinity of the enclosures averaged more than 25 mGy d^{-1}, whereas *Clethrionomys* received approximately 1.5 mGy d^{-1} and *Mus* approximately 4 mGy d^{-1} from internally deposited ^{90}Sr at the termination of the enclosure study.

Comparison of the empirical and predicted (Eqn. 6) accumulation of ^{137}Cs for enclosure mice showed an excellent fit (Fig. 1) for both species. Therefore, accumulated dose estimates for mice over the enclosure period (Fig. 2) were likely

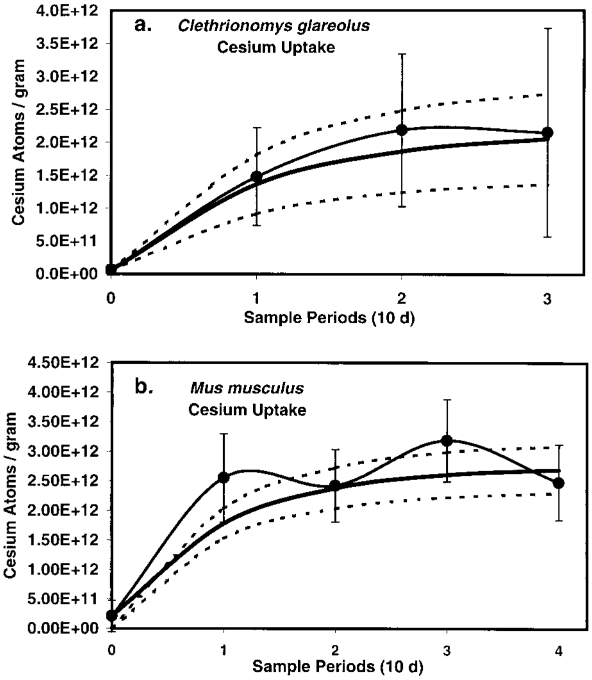

Fig. 1. Empirical and predicted uptake curves for whole-body ^{137}Cs in *Clethrionomys glareolus* (**a**) and *Mus musculus* (**b**) maintained in enclosures in the Red Forest near Chornobyl, Ukraine. Mean values are shown by the dots connected with a solid line; the upper and lower 95% confidence intervals are shown by the dashed lines and vertical bars. Sample periods: 0 = initial, 1 = 10 d, 2 = 20 d, and so on.

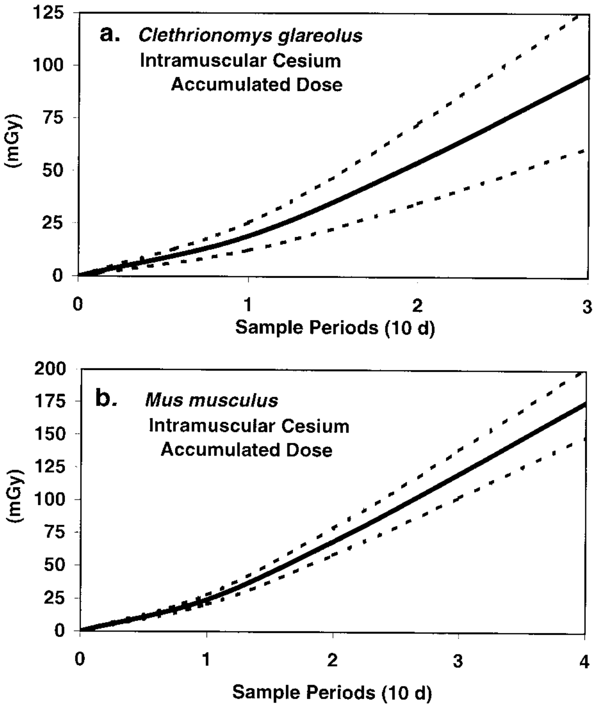

Fig. 2. Dose (in mGy) accumulation curves for intramuscular ^{137}Cs in *Clethrionomys glareolus* (**a**) and *Mus musculus* (**b**) that were maintained in enclosures in the Red Forest near the Chornobyl reactor. The *C. glareolus* were maintained in the enclosures for only 30 d, whereas *M. musculus* were removed after 40 d. ^{137}Cesium was measured from whole-body counts. Dashed lines denote the estimated upper and lower 95% confidence intervals.

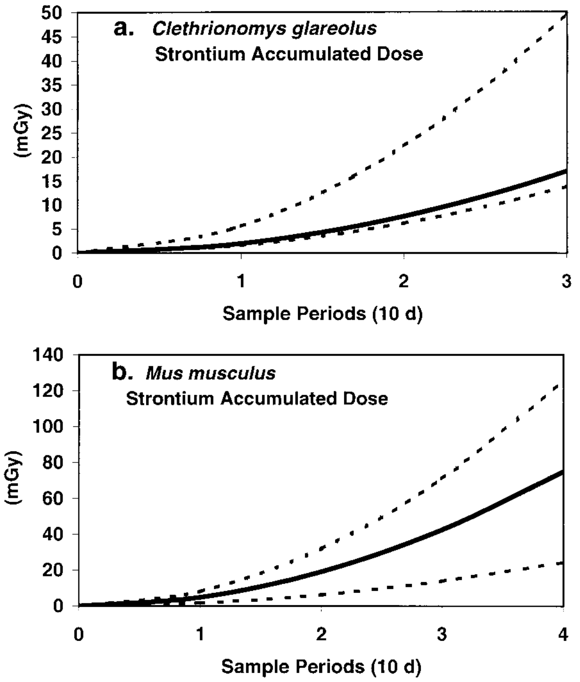

Fig. 3. Dose (in mGy) accumulation curves for skeletal ^{90}Sr in *Clethrionomys glareolus* (**a**) and *Mus musculus* (**b**) maintained in enclosures in the Red Forest near Chornobyl, Ukraine. The *C. glareolus* were maintained in the enclosures for only 30 d, whereas *M. musculus* were removed after 40 d. Because ^{90}Sr was measured from bone, repeated sampling during the course of the experiment was not possible. Thus, only initial (period 0, from a subsample) and terminal (period 4 or 5) measurements were made. Dashed lines denote the upper and lower 95% confidence intervals.

to be robust. *Clethrionomys* in the enclosures received approximately 98 mGy from intramuscular ^{137}Cs during their 30-d exposure. *Mus* received approximately 174 mGy as a result of intramuscular deposition of ^{137}Cs during their 40-d exposure. The uptake rates for ^{90}Sr in the enclosures resulted in accumulated doses of 18.64 mGy for *Clethrionomys* over 30 d and 68.4 mGy for *Mus* over 40 d (Fig. 3). These doses were due only to the fraction of ^{90}Sr incorporated after placement in the enclosures, and they do not include the doses that resulted from the ^{90}Sr that the animals contained before their introduction. Therefore, the total dose caused by internally deposited ^{137}Cs and ^{90}Sr during the course of the enclosure study was estimated as 116.7 mGy for *Clethrionomys* and 242.4 mGy for *Mus*. If these accumulations are extrapolated over the period of one year, then *Clethrionomys* would receive 1,590 mGy per year from ^{137}Cs and 1,856 mGy per year from ^{90}Sr. Similarly, *Mus* would receive annual doses of 2,011 mGy and 4,633 mGy from ^{137}Cs and ^{90}Sr, respectively. These results demonstrate that internal doses conveyed by uptake through abiotic avenues can be substantial.

For relocated adults, the accumulation rate for ^{90}Sr would predict that the ultimate asymptotic concentration is achieved after approximately 758 d. The very long accumulation time for ^{90}Sr is due to the low rate of loss through biological depuration. Clearly, enclosure mice were far from approaching asymptotic values after only 30 or 40 d. In fact, the 758 d required to reach 90% of asymptote for ^{90}Sr approaches the probable maximum life span for *Clethrionomys* in this region [12]. Accumulation rates for mice born within the contaminated region, however, likely are very different from those of relocated adults. Fetal and neonatal uptake of ^{90}Sr likely is rapid, due to downloading of ^{90}Sr from the mother's bone and milk. Higher uptake rates would dictate that newly weaned animals have higher ^{90}Sr concentrations than the equations presented here would predict. A litter of four *Clethrionomys* was born to the pregnant female housed separately in an enclosure and undisturbed during the study, and the young were sampled at the end of the weaning time. Average ^{90}Sr concentrations in these four were higher than those in all other enclosure animals, approximately 2.5-fold higher than concentrations in the mother (young, 8,731 Bq g^{-1}; mother, 3,600 Bq g^{-1}), and also higher than the average for wild-caught mice in the area (7,249 Bq g^{-1}). It is not unreasonable, therefore, to expect that native animals in the Red Forest are approaching asymptotic concentrations of ^{90}Sr very rapidly. Hence, concentrations of ^{90}Sr in wild-caught mice were assumed to be at their equilibrium value (loss rate = uptake rate = 2.91×10^{10} unstable atoms per gram, $\hat{A} = 9.56 \times 10^{12}$). The *Clethrionomys* in the enclosures would be expected to acquire an ultimate ^{90}Sr concentration of 6.38×10^{12} unstable atoms per gram. The ratio of expected asymptotic values for unstable ^{90}Sr atoms in enclosure *Clethrionomys* and wild-caught mice was 66.7%. In this instance, therefore, approximately two-thirds of ^{90}Sr is transported via abiotic means, leaving approximately one-third to be transported via biotic pathways.

Because both wild-caught and captive *Clethrionomys* likely had approached asymptotic values for concentrations of ^{137}Cs, direct comparisons were made to estimate the fraction of uptake, ultimate concentrations, and dose rates caused by the biotic and abiotic pathways. Despite the rather substantial incorporation rate of ^{137}Cs in enclosure animals, it was estimated that only 8.5% of the concentration and dose rate in *Cleth-*

Table 3. Comparison of ^{137}Cs activity, concentration values, and dose rate estimates for wild-caught and captive *Clethrionomys glareolus* in the Red Forest near the Chornobyl (Ukraine) reactor and the fraction of these measures contributed from abiotic (soil, air and water) and biotic (plant) sources[a]

Clethrionomys glareolus	Wild-caught animals (n = 27) (Total = Abiotic + Biotic)	Enclosure animals (n = 11) (Abiotic)	Estimated biotic fraction (Total − Abiotic)
Becquerels per gram (wet weight)	22,200	1,580	20,620
	(1,902)	(339.3)	(2,241)
^{137}Cs atoms per gram	3.04×10^{13}	2.17×10^{12}	2.83×10^{13}
	(2.26×10^{12})	(3.70×10^{11})	(2.59×10^{12})
^{137}Cs uptake rate	2.93×10^{12}	2.26×10^{11}	2.72×10^{11}
(atoms/g/d)	(2.17×10^{11})	(3.57×10^{10})	(2.53×10^{11})
Dose rate	64.53	5.48	59.04
(mGy/d)	(5.52)	(1.3)	(6.83)
% Dose rate (mGy/d)	100	8.5	91.5

[a] Values for enclosure animals are estimates of the asymptotes attained during the time of confinement (30 d), whereas those for the wild-caught animals are means (standard errors in parentheses) derived from free-ranging animals captured in the vicinity of the enclosures.

rionomys from the Red Forest was due to uptake by abiotic means (Table 3). Therefore, the vast majority (91.5%) of the intramuscular ^{137}Cs load is promoted by ingestion of vegetative material that is contaminated with radioactive material. Wild-caught *Clethrionomys* in the vicinity of the enclosures were experiencing internal dose rates from ^{137}Cs of more than 64 mGy d^{-1}, and it is estimated that 59 mGy d^{-1} was due to transfer from the biotic community and only 5.5 mGy d^{-1} from uptake of free particles.

Specific activities and concentrations in soil samples from the enclosures were 1,296.06 Bq g^{-1} (1.71×10^{12} unstable atoms/g) for ^{90}Sr and 3,317 Bq g^{-1} (4.54×10^{12} unstable atoms/g) for ^{137}Cs, yielding a ^{90}Sr:^{137}Cs ratio of 0.38. The estimated ^{90}Sr:^{137}Cs ratio at asymptote in mice was 0.38, which was calculated using the skeleton as comprising 0.12% of the body mass and the muscle and soft tissues as comprising 0.88% [7]. Although this ratio is a close match to the soil concentration ratio, we know of no functional reason to expect that they would coincide. Comparisons of the uptake rates for mice in the enclosures to the concentrations in the soil indicate that incorporation of ^{137}Cs from soil into muscle and tissue mass would require ingestion of approximately 0.88 g of soil per day for *Clethrionomys* and 1.3 g of soil per day for *Mus*. The required volumes of soil ingested per day to satisfy the uptake rates for ^{90}Sr were much less, with 0.027 g of soil per day for *Clethrionomys* and 0.52 g of soil per day for *Mus*.

It was estimated that the average adult mouse was receiving an effective dose from 62.8 g of soil while in their nest cavity (assumed spherical dimensions, 4π[5 cm]), and that approximately 3 and 5% of the incidental energy from nuclear disintegrations were absorbed beyond surface areas (e.g., skin, hair, eyes) for ^{137}Cs and ^{90}Sr (and ^{90}Y), respectively. These estimates yielded daily external dose rates of 18.1 mGy d^{-1} from ^{137}Cs and 14.24 mGy d^{-1} from ^{90}Sr in the soil. Therefore, the estimated accumulated external doses (Table 4) received during the 30-d period for *Clethrionomys* in the enclosures were 543.3 mGy from ^{137}Cs and 427.2 mGy from ^{90}Sr. Expected accumulated external doses for *Mus* during the 40 d of captivity in the enclosures were 724 mGy for ^{137}Cs and 570 mGy for ^{90}Sr. The total dose resulting from internal and external sources of radiation in *Clethrionomys* was 641.3 mGy from ^{137}Cs and 446 mGy from ^{90}Sr, yielding the grand sum of 1,087 mGy over 30 d of exposure. Comparable total doses for *Mus* during the 40 d in the enclosures were 898 mGy from ^{137}Cs and 638 from ^{90}Sr, yielding a total accumulated dose of 1,536 mGy.

DISCUSSION

The results reported here for the animals housed in the enclosures demonstrate that substantial uptake, internal concentrations, and resultant absorbed doses are possible due to the incorporation of ^{137}Cs and ^{90}Sr from abiotic material. In regions like the Red Forest, the risk of internal contamination via particulates present in the soil, air, and water is considerable. The accumulated doses from the uptake of ^{137}Cs and ^{90}Sr from abiotic sources were only a small fraction of those accrued by wild mice living in areas adjacent to the enclosures. Nevertheless, internal doses of 117 mGy in *Clethrionomys* and 242 mGy in *Mus* warrant attention in regards to the speed and magnitude of their accumulation from the uptake of abiotic material.

The uptake dynamics for ^{137}Cs and ^{90}Sr were quite different because of the properties for biological decay rates. Although the concentration of unstable atoms for ^{137}Cs was more than 2.5-fold greater than that of ^{90}Sr in the soils, almost 20-fold more soil would need to be ingested to satisfy the daily incorporation of ^{137}Cs atoms into biological tissues. The rapid turnover of ^{137}Cs in muscle and soft tissues quickly eliminated the initial concentrations and allowed asymptotic levels to be attained within three weeks. High electrolytic demand for potassium and, hence, its analogue (^{137}Cs) was evidenced by the low percentage transferred by abiotic (8.5%) relative to biotic (91.5%) means. Although each gram of soil averaged more than 4.5×10^{12} unstable ^{137}Cs atoms, ingestion of abiotic material was insufficient to attain the very high concentrations found in adjacent, wild-caught mice that were ingesting ra-

Table 4. Estimates of the contributions to the total accumulated whole-body dose (in mGy) for *Clethrionomys glareolus* and *Mus musculus* during the enclosure period[a]

	Internal (mGy)		External (mGy)		Total (mGy)	
Dose	^{137}Cs	^{90}Sr	^{137}Cs	^{90}Sr	^{137}Cs	^{90}Sr
C. glareolus						
(30-d exposure)	98	19	543	427	641	446
	Σ = 117		Σ = 970		Σ = 1,087	
M. musculus						
(40-d exposure)	174	68	724	570	898	638
	Σ = 242		Σ = 1,294		Σ = 1,536	

[a] All values were estimated from models derived in the text. Assignment of standard errors and performance of statistical tests for these values were not practical.

dionuclides from both biotic and abiotic sources. Several scenarios can explain the large difference between ^{137}Cs uptake from biotic and abiotic sources. Because the Red Forest is very near Reactor 4, where the accident occurred, many of the radionuclides are presently either associated with fuel-rod particles or are bound in the soil matrix [14]. Therefore, the ^{137}Cs from soil that passes through the gastrointestinal tract may be largely unavailable for biological uptake [14] or excreted shortly after uptake. Vast quantities of soil ingestion may be required to meet the biological demand. The ^{137}Cs from vegetation, however, is largely unbound and, probably, is transferred readily. Even if concentrations of ^{137}Cs are lower in vegetation than in the soils, the biological availability may enhance the uptake rate. Also, the quantity of vegetation ingested by native rodents is expected to far exceed the amount of soil ingested. Reported soil ingestion by voles comprised only 2.4% of their total diet [26].

The solubility of ^{90}Sr is typically much greater than that of ^{137}Cs, making ^{90}Sr more biologically available after years of weathering and erosion [14]. High biological availability, together with very slow biological elimination, of ^{90}Sr means that relatively low quantities of ingestion (either biotic or abiotic) are required to meet biological demands. These properties may contribute to a high percentage (66.7%) of ^{90}Sr uptake from abiotic sources relative to the uptake in wild mice. On the contrary, the high solubility of ^{90}Sr could hasten its loss from biological systems by downward movement into the soil profile. However, incorporation of only approximately 0.03 g of soil per day was necessary to supply sufficient unstable atoms to approach asymptotic values for ^{90}Sr in enclosure mice. Data from ^{137}Cs indicate that at least 0.88 g of soil need to be processed to meet the ^{137}Cs levels observed. This volume of soil should provide an excess of ^{90}Sr. Even a low efficiency of acquiring chemically bound ^{90}Sr should be sufficient to saturate tissues. Therefore, that abiotic intake did not account for a higher percentage of ^{90}Sr intake than that estimated here is surprising.

Although the abiotic uptake of ^{90}Sr is very high compared to that of ^{137}Cs, the percentage may be higher than that predicted above. The long biological half-life for ^{90}Sr made it difficult to estimate the asymptotic values for either wild or captive animals. Data for the young born in the enclosures showed that ^{90}Sr concentrations were very high in postweaning offspring, leading to the conclusion that wild animals were at asymptotic values. This conclusion likely is incorrect, however, because the asymptotic values may, in fact, be lower than the concentrations seen in the wild animals. The uptake rates of ^{90}Sr during suckling probably are much higher than those after weaning, when animals shift to adult diets (e.g., seeds, plants, insects, lichens, fungi). Concentrations of ^{90}Sr in young animals were found to contain higher ^{90}Sr concentrations than those in adults [4], and newly weaned voles born in the enclosure had ^{90}Sr concentrations that were more than 2.5-fold greater than that of their mother as well as greater than those of wild-caught adult and subadult/juvenile mice. Furthermore, the process of bone growth is fundamentally different from that of bone maintenance. In bone growth during suckling, the ^{90}Sr from lactation would be directly transferred to new bone. Thus, uptake rates (u) of suckling young would be much higher than those of adults. This would lead to the asymptotic value of ^{90}Sr concentration in the bone of sucklings being equal to the concentration in milk. Postweaning young would experience a lower uptake rate than they did before weaning, leading to a gradual decay, rather than an increase, in the concentration of skeletal ^{90}Sr over time. The slow rate of biological decay would dictate that most rodents would never attain this lower asymptotic value during their lifetimes. Thus, the concentrations of ^{90}Sr in wild rodents likely are in excess of their ultimate asymptotic value, albeit that asymptote may never be attained. Taking this into account, the percentage of uptake of ^{90}Sr from abiotic contributions would be greater than the estimated two-thirds calculated above. Unfortunately, we do not have sufficient information to test these scenarios, and future studies will be aimed at obtaining the parameters necessary to refine our models. Hence, the calculated 66.7% uptake rate of ^{90}Sr from abiotic sources is a conservative estimate of the true value.

The separation of abiotic and biotic components of radionuclide uptake by comparison of wild animals to those in enclosures involved several assumptions. The conditions of the enclosures, although intended to simulate the natural habitat, may have presented the animals with unique situations that either stimulated or inhibited uptake through abiotic means. Disturbance of the soil profiles during construction of the enclosures, recapture of mice, and extensive burrowing by the rodents eliminated the vertical distribution of radionuclides. Also, the soil particles in the enclosures were not as tightly bound by roots, due to the very scant vegetation, as would be found in natural conditions, which possibly enhanced abiotic ingestion and inhalation. Furthermore, the calcium and potassium availability in natural vegetation relative to that in the clean diet supplied to the enclosures is unknown. The competition of stable elements with molecular analogues usually decreases the efficiency of radionuclide uptake (calcium and potassium are preferentially incorporated over ^{90}Sr and ^{137}Cs [14]). Therefore, as in any experimental treatment, the estimates derived here are potentially biased by other factors not controlled for in the design. Despite these limitations, the data are nonetheless irrefutable in regards to the potential for significant abiotic contributions to radionuclide load.

Availability and uptake of ^{137}Cs and ^{90}Sr to the biota have been increasing since the time of the Chornobyl accident [27–29]. Dissociation of radionuclides from fuel particles as well as physiochemical and biochemical transformations of the radionuclides in regions near the reactor have recently led to greater internal loads than were evidenced during the first several years after the accident [28,29]. Populations more distant from the reactor, in fact, had greater ^{137}Cs concentrations because of an inverse relationship between particle size and distance from the reactor [28,29]. Clearly, the data from the Red Forest now indicate that these radionuclides are sufficiently mobile in the environment to pose biological risks through uptake from abiotic and biotic sources.

Chronic radiation doses in excess of 1 mGy d^{-1} reduce fertility in rodents [9]. The International Atomic Energy Agency [8] has established 1 mGy d^{-1} as the upper limit of the dose rate acceptable for terrestrial vertebrates. The animals in the enclosures were experiencing dose rates far in excess of this statute from internally deposited radionuclides alone. External dose rate estimates were 18.1 mGy d^{-1} from ^{137}Cs and 14.24 mGy d^{-1} from ^{90}Sr. Despite these dose rates, a pregnant female *Clethrionomys* raised four offspring to weaning, and several female *Clethrionomys* and *Mus* conceived and produced living progeny in the course of the enclosure study. Although further research is needed regarding the efficacy of some important biological endpoints, such as reproduction [9] and genetic damage [13,30–40], the contributions of various segments of the radioactive environment to these endpoints must be clearly defined. This study demonstrated that considerable biological risk in the Red Forest at Chornobyl remains, even when the uptake from biotic sources is eliminated. It is anticipated that future dissociation of radionuclides from fuel and soil particles will enhance the potential for uptake from abiotic components beyond that reported here.

Acknowledgement—We thank I. Shokelevitch, M. Bondarkov, and B. Oskolkov. Special gratitude is extended to V. Kholosha for granting access and transport within the Chornobyl exclusion zone. This work was supported by contract DE-FC09-96SR18546 between the University of Georgia and the U.S. Department of Energy, by Texas Tech University, and by funds from an anonymous donor. We thank the Ministry of Ukraine for Environmental Protection and Nuclear Safety, the Ministry of Emergency Situations and Protection of the Population from the Consequences of the Chornobyl Nuclear Accident, the Cabinet of Ministers, Chornobyl Center for Nuclear Safety, Radioactive Waste, and Radioecology, and Industrial Association Chornobyl NPP. This work was conducted as a program in the International Radioecology Laboratory in Slavutych, Ukraine.

REFERENCES

1. Kownacka L, Jaworowski Z. 1994. Nuclear weapon and Chernobyl debris in the troposphere and lower stratosphere. *Sci Total Environ* 144:201–215.
2. Sich AR. 1994. Chernobyl accident management actions: Implications for source term estimates. *Nucl Saf* 35:1–24.
3. Borovoi AA, Sich AR. 1995. The Chernobyl accident revisited, part II: The state of the nuclear fuel located within the Chernobyl sarcophagus. *Nucl Saf* 36:1–32.
4. Shevchenko VA. 1994. *The Ecology of the Chernobyl Disaster.* Man & Biosphere Series. Parthenon Publishing, Paris, France.
5. Izrael YUA, et al. 1987. Radioactive pollution of the natural environment in the zone of the accident of the Chernobyl Atomic Power Plant. *Meterologiya Hydrologiya* 2:5–18. (In Russian).
6. Medvedev ZA. 1994. Chernobyl: Eight years after. *Tree* 9:369–371.
7. Chesser RK, et al. 2000. Concentrations and dose rate estimates of ^{134,137}Cesium and ^{90}Strontium in small mammals at Chornobyl, Ukraine. *Environ Toxicol Chem* 19:305–312.
8. International Atomic Energy Agency. 1992. Effects of ionizing radiation on aquatic organisms and ecosystems. Series 332. Technical Report. Vienna, Austria.
9. Turner FB. 1975. Effects of continuous irradiation on animal populations. *Adv Radiat Biol* 5:83–144.
10. Chesser RK, Baker RJ. 1996. Life continues at Chernobyl. *La Recherche* 286:30–31. (In French).
11. Baker RJ, Chesser RK. 2000. The Chornobyl nuclear disaster and subsequent creation of a wildlife preserve. *Environ Toxicol Chem* 19:1231–1232.
12. Baker RJ, et al. 1996. Small mammals from the most radioactive sites near the Chornobyl nuclear power plant. *J Mammal* 77:155–170.
13. Rodgers BE, Baker RJ. 2000. Frequencies of micronuclei in bank voles from zones of high radiation at Chornobyl, Ukraine. *Environ Toxicol Chem* 19:1644–1648.
14. Whicker FW, Schultz V. 1982. *Radioecology: Nuclear Energy and the Environment,* Vol II. CRC, Boca Raton, FL, USA.
15. Whicker FW, Pinder JE III, Bowling JW, Alberts JJ, Brisbin IL Jr. 1990. Distribution of long-lived radionuclides in an abandoned reactor cooling reservoir. *Ecol Monogr* 60:471–496.
16. Whicker FW. 1997. Impact on plant and animal populations. In *Health Impacts of Large Releases of Radionuclides.* Ciba Foundation Symposium 203. Ciba Foundation, London, UK, pp 74–93.
17. Hinton TG, Kopp P, Ibrahim S, Bubryak I, Syomov A, Tobler L. 1993. Contaminated soil on Chernobyl vegetation. In Kathren RL, Denham DH, Salmon K, Richland WA, eds, *Proceedings,* 26th Topical Meeting Health Physics Society, Coeur D'Alene, ID, USA, pp 407–421.
18. Mazurak AP, Mosher PN. 1968. Detachment of soil aggregates by simulated rainfall. *Soil Sci Soc Am Proc* 34:798–800.
19. Milham RC, Schubert JF, Watts JR, Boni AL, Corey JC. 1976. Measured plutonium resuspension and resulting dose from agricultural operation on an old field at the Savannah River Plant in the southeastern United States of America. In *Transuranium Nuclides in the Environment.* STI/PUB/410. International Atomic Energy Agency, Vienna, Austria, pp 409–421.
20. Romney EM, Wallace A. 1977. Plutonium contamination of vegetation in dusty field environments. In White MG, Dunaway PB, eds, *Transuranics in Natural Environments.* Gatlinburg, TN, USA, pp 287–302.
21. Kitchings JT, Dunaway PB, Story JD. 1969. Uptake and excretion of ^{134}Cs from fallout simulant and vegetation by cotton rats. *Health Phys* 17:265–277.
22. National Academy of Sciences of Ukraine. 1996. *Atlas of Chernobyl Exclusion Zone.* Kiev, Ukraine.
23. Jagoe CH, Dallas CE, Chesser RK, Smith MH, Lingenfelser SK, Lingenfelser JT, Holloman K, Lomakin MD. 1998. Contamination near Chernobyl: Radiocesium, lead and mercury in fish and sediment Radiocesium from waters within the 10 km zone. *Ecotoxicology* 7:1–9.
24. Kitchings T, DiGregorio D, Van Voris P. 1976. A review of the ecological parameters of radionuclide turnover in vertebrate food chains. In Cushing CE, et al., eds, *Radioecology and Energy Resources.* Dowden Hutchinson and Ross, Stroudsburg, PA, USA, pp 304–313.
25. Reichle DE, Dunaway PB, Nelson DJ. 1970. Turnover and concentration of radionuclides in food chains. *Nucl Saf* 11:43–54.
26. Beyer WN. 1994. Estimates of soil ingestion by wildlife. *J Wildl Manage* 58:375–382.
27. Belli M, et al. 1996. Dynamics of radionuclides in forest environments. In Karaoglou A, Desmet G, Kelly GN, Menzel HG, eds, *The Radiological Consequences of the Chernobyl Accident.* European Commission, London, UK, pp 69–80.
28. Eriksson O, et al. 1996. Evolution of the contamination rate in game. In Karaoglou A, Desmet G, Kelly GN, Menzel HG, eds, *The Radiological Consequences of the Chernobyl Accident.* European Commission, London, UK, pp 147–154.
29. Jacob P, et al. 1996. Exposures from external radiation and from inhalation of resuspended material. In Karaoglou A, Desmet G, Kelly GN, Menzel HG, eds, *The Radiological Consequences of the Chernobyl Accident.* European Commission, London, UK, pp 251–269.
30. Lingenfelser SF, Lingenfelser JT, Holloman K, Jagoe CH, Kind JA, Chesser RK, Smith MH, Dallas CE. 1998. Flow cytometric analysis of erythrocyte and leukocyte DNA in fish from Chernobyl-contaminated ponds in the Ukraine. *Ecotoxicology* 7:211–219.
31. Matson CW, Rodgers BE, Chesser RK, Baker RJ. 2000. Genetic diversity of *Clethrionomys glareolus* populations from highly contaminated sites in the Chornobyl region. *Environ Toxicol Chem* 19:2130–2139.
32. Goncharova RI, Ryabokon NI. 1995. Dynamics of cytogenetic injuries in natural populations of bank vole in the republic of Belarus. *Radiation Protection and Dosimetry* 62:37–40.
33. Shevchenko VA, Pomerantseva MD, Ramaiya LK, Chekhovich AV, Testov BV. 1992. Genetic disorders in mice exposed to radiation in the vicinity of the Chernobyl nuclear power station. *Sci Total Environ* 112:45–56.
34. Cristaldi M, D'Arcangelo ED, Ieradi LA, Mascanzoni D, Mattei T, Van Axel Castelli I. 1990. ^{137}Cs determination and mutagenicity tests in wild *Mus musculus domesticus* before and after the Chernobyl accident. *Environ Pollut* 64:1–9.
35. Cristaldi M, Ieradi LA, Mascanzoni D, Mattei T. 1991. Environmental impact of the Chernobyl accident: Mutagenesis in bank voles from Sweden. *Int J Radiat Biol* 59:31–40.
36. Sugg DW, et al. 1996. DNA damage and radiocesium in channel catfish from Chernobyl. *Environ Toxicol Chem* 15:1057–1063.
37. Ellegren H, Lindgren G, Primmer CR, Moller AP. 1997. Fitness loss and germline mutations in barn swallows breeding in Chernobyl. *Nature* 389:593–596.
38. Stephan G, Oestreicher U. 1989. An increased frequency of structural chromosome aberrations in persons present in the vicinity of Chernobyl during and after the reactor accident: Is this effect caused by radiation exposure? *Mutat Res* 223:7–12.
39. Scheid W, Weber J, Petrenko S, Traut H. 1992. Chromosome aberrations in human lymphocytes apparently induced by Chernobyl fallout. *Health Phys* 64:531–534.
40. Dubrova YE, Nesterov VN, Krouchinsky NG, Ostapenko VA, Neumann R, Neil DL, Jefferys AJ. 1996. Human minisatellite mutation rate after the Chernobyl accident. *Nature* 380:683–686.