

¹³⁷Cs ELIMINATION BY CHRONICALLY-CONTAMINATED LARGEMOUTH BASS (*MICROPTERUS SALMOIDES*)

Eric L. Peters*[†] and Michael C. Newman*[‡]

Abstract—The temperature-dependent ¹³⁷Cs biological half-times (T_b) of lifetime-exposed largemouth bass (*Micropterus salmoides*) from a nuclear cooling reservoir at the U.S. Department of Energy's Savannah River Site were calculated from whole-body measurements of live fish and compared with literature records for acutely and chronically-contaminated fish. The T_b 's of the bass averaged 322 d (95% CI: 311–333 d), 225 d (95% CI: 220 to 230 d), and 140 d (95% CI: 137 to 143 d) at 15, 20, and 26 °C, respectively. These mean T_b 's were 1.7 to 2.5 times longer than would be expected for acutely contaminated fish, and 1.2 to 1.8 times longer than those predicted for fish at steady-state with their environment according to recent models. This slower elimination did not appear to result from slower elimination from skeletal muscle compared with other soft tissues, in that the muscle to whole-body ¹³⁷Cs concentration ratios after the elimination period were similar to those of freshly-caught bass. Our results suggested that elimination rates estimated from the terminal elimination components of acutely-dosed fish may not reflect the elimination rates of fish exposed to contaminants throughout their lifetime, even when care is taken to allow sufficient time for absorption of the dose. *Health Phys.* 76(3):260–268; 1999

Key words: ¹³⁷Cs; biokinetics; dose, absorbed; environmental impact

INTRODUCTION

AS A result of nuclear accidents and weapons detonations, significant levels of ¹³⁷Cs have been released throughout the world. ¹³⁷Cs is a high-yield product of nuclear fission, with a long physical half-time (30.2 y) and high bioavailability (Whicker and Schultz 1982) and is, in fact, the most important long-lived contaminant released during the Chernobyl accident (Anspaugh et al. 1988; Medvedev 1990; Kryshev 1992). Because ¹³⁷Cs is readily accumulated and concentrated in skeletal muscle,

consumption of contaminated fish and other wildlife can be an important pathway for human exposure (Forseth et al. 1991; Whicker et al. 1993). Due to the large number of wetland areas subjected to Chernobyl fallout, there has been renewed interest in the fate of radionuclide contaminants in aquatic ecosystems, and new studies of the importance of fish as indicators of the presence of radionuclides and/or their potential contribution to human dose commitments are emerging (e.g., Korhonen 1990; Forseth et al. 1991; Koulikov and Ryabov 1992).

Some recent studies of aquatic vertebrates (e.g., Evans 1988; Peters and Brisbin 1996) have combined physiological and ecological models to predict ¹³⁷Cs kinetics. However, most radiocesium kinetics studies of fish continue to emphasize direct measurements of biological half-time (T_b) in the laboratory, generally after acute dosage. However, both field and laboratory studies have revealed a wide range of T_b values for ¹³⁷Cs in freshwater fish (Table 1). Within individual species, ¹³⁷Cs T_b 's are affected by factors that influence metabolic rate, increasing by a factor (Q_{10}) of 2 to 3 for each 10 °C increase in temperature, and to a lesser degree in proportion to body mass (Kevern 1966; Gallegos and Whicker 1971; Fagerström 1977; Mailhot et al. 1989; Ugedal et al. 1992). As is also true of metabolic rate, other factors such as the age of the fish and the conditions under which it is acclimated may also influence ¹³⁷Cs elimination rates. Unfortunately, most prior ¹³⁷Cs elimination studies have not provided information on these conditions and, more critically, have also failed to report the experimental temperatures, the sizes of the fish, or both. These deficiencies were noted in a recent effort (Rowan and Rasmussen 1995) to produce a general model to predict ¹³⁷Cs T_b 's of fish as a function of temperature and body mass using data from published elimination studies. Because acutely-dosed fish often exhibit artificially high initial losses of administered radionuclides (e.g., through rapid loss of unabsorbed oral doses via egestion or leakage from injection sites), care was taken by the authors to eliminate studies (or portions of studies) that included such "short components" of elimination. Despite this, the authors noted that the elimination rates of chronically-contaminated fish were about 40% longer than expected for acutely-exposed fish. They hypothesized that certain tissues (especially skeletal muscle) accumulate and eliminate cesium more slowly than others, and developed a common model

* Savannah River Ecology Laboratory, P.O. Drawer E, Aiken, SC 29802; [†] Present address: Department of Biological Sciences, Chicago State University, 9501 S. King Drive, Chicago, IL 60628-1598; [‡] Present address: Department of Environmental Sciences, The College of William and Mary, Virginia Institute of Marine Science School of Marine Science, Gloucester Point, VA 23062-1346.

For reprints contact: Reprints Dept., Savannah River Ecology Laboratory, P.O. Drawer E, Aiken, SC 29802.

(Manuscript received 7 July 1997; revised manuscript received 7 July 1998, accepted 23 October 1998)

0017-9078/99/0

Copyright © 1999 Health Physics Society

Table 1. Whole-body elimination half-times (T_b) of ¹³⁷Cs by freshwater fish.

Species	Wet mass (g)	Temperature (°C)	T_b (d)	Reference	
<i>Cyprinus carpio</i>	137 ^a	12.5/20	174/98	Kevern (1966)	
<i>Leuciscus rutilus</i> <i>Perca fluviatilis</i> <i>Salmo iridaeus</i>	NA	10–20	55–100 175–200 25–80	} Häsinen et al. (1967)	
<i>Onchorhynchus mykiss</i>	30–200	5–18.3	34–92		Gallegos and Whicker (1971)
<i>Lepomis macrochirus</i> <i>Esox lucius</i>	0.5–120 100–1,500	15.8 8–10	86–187 402–599		Kolehmainen (1972) Carlsson (1978)
<i>Salmo trutta</i> <i>Salvalinus alpinus</i>	250 ^b (normalized)	≈12.5 ^b	124 ^b 150 ^b	} Forseth et al. (1991)	
<i>Hypophthalmichthys molitrix</i>	500–7,000	NA	44–231		Koulikov and Ryabov (1992)
<i>Salmo trutta</i> <i>Ictalurus punctatus</i>	12–456 122–188	4.4–15.6 20/27.5	104–564 154/84.6	Ugedal et al. (1992) Peters et al. ^c	

^a Mean estimated indirectly from other parts of the study (Rowen and Rasmussen 1995).

^b Estimated by Rowan and Rasmussen (1995) from Forseth et al. (1992).

^c E. L. Peters, I. R. Schultz, and M. C. Newman (unpublished data).

containing a "correction factor" to allow prediction of the half-times of chronically contaminated fish. Although this refined model was successful in improving predictions of half-life for the two studies of chronically-contaminated fish included in their model (i.e., Kolehmainen 1972; Baptist and Price 1962), the authors suggested that further research was necessary, both to document the general prevalence of longer T_b 's in fish at equilibrium with the environment, and to examine possible internal redistribution kinetics of radiocesium that might be responsible for this effect.

Accurate predictions of radiocesium elimination are important in gauging potential health effects of environmental releases, as well as for estimating food consumption by free-ranging fish (e.g. Kevern 1966; Kolehmainen 1972; Forseth et al. 1992). However, the above-mentioned difficulties in obtaining such data from animals administered radiocesium in the laboratory suggest the need for more careful examination of fish from already contaminated habitats. Accordingly, the whole-body T_b 's of ¹³⁷Cs of lifetime-exposed individuals of a common game fish (largemouth bass, *Micropterus salmoides*) were measured to determine whether they conformed to other studies of acutely and chronically-contaminated species. Also, because none of the prior studies of ¹³⁷Cs elimination by chronically-contaminated fish was conducted at more than a single temperature, the measurements were conducted over much of the range of water temperatures to which the fish were likely to be exposed to determine whether any observed deviations from expected T_b 's were temperature-dependent. Finally, to determine whether muscle elimination kinetics of bass differed from that of the animal as a whole, the muscle: whole body ratios of ¹³⁷Cs of the bass after elimination of a substantial fraction of their whole-body burdens were compared with those of freshly-caught individuals.

MATERIAL AND METHODS

One population of *M. salmoides* inhabits Pond B, an abandoned nuclear reactor cooling reservoir located on the U.S. Department of Energy's Savannah River Site (SRS) near Aiken, South Carolina. Pond B was constructed in 1960, and received thermal effluents from one of the SRS weapons production reactors (R reactor) from September 1961 to June 1964. In 1964, R reactor was shut down, and the fuel assemblies were moved to a water-filled storage basin located near the discharge canal for the hot water effluents (R Canal). A faulty experimental fuel element contaminated the water in the storage basin, and this contaminated water, containing approximately 5.7 TBq of ¹³⁷Cs (Ashley and Zeigler 1980), was released into the R Canal. Although some of this contamination was diverted elsewhere, a substantial fraction was subsequently washed into Pond B.

The physical, chemical, and biological environment of Pond B is well described and is discussed elsewhere (Alberts et al. 1987; Whicker et al. 1990; Pinder et al. 1992; Kelly and Pinder 1995). Briefly, Pond B is a large (87 ha) impoundment, with a mean depth of 4.3 m and a maximum depth of 12.5 m. Pond B is monomictic, stratifying from April through October, with a thermocline at 6–7 m. Since the input of cooling water was discontinued, the lake basin has remained filled through precipitation and subsurface drainage alone, and the concentrations of dissolved solids are low, as is characteristic of other surface waters in the region: 1.6 (Na), 0.5 (K), 1.3 (Ca), 0.5 (Mg), and 6.6 (bicarbonate), all in mg L⁻¹. Largemouth bass from this lake have high ¹³⁷Cs levels (approximately 3 kBq kg⁻¹ wet mass, or three orders of magnitude greater than local background levels), making them exceptionally suitable for long-term measurements of ¹³⁷Cs elimination.

Experimental animals

Twenty-eight largemouth bass (130–423 g) were collected from Pond B by angling and transported to a holding facility at the University of Georgia's Savannah River Ecology Laboratory. Upon arrival, the fish were placed in individual 18-L white polyethylene buckets with snap-on lids. The buckets had numerous 1-cm holes drilled through the bottoms and lids to permit water exchange, and were floated in one of three 1,090-L polyethylene tanks.⁸ Each tank contained 10–11 fish, with less than 3.5 g fish L⁻¹ water. Water temperature was maintained using immersion heaters and refrigeration units.¹¹ Each tank was provided with flow-through (2.3 L min⁻¹) well water (median pH = 7.1, range 7.0–7.1, $n = 15$), which had a total alkalinity and hardness of 4.7 and 7.1 mg L⁻¹ as CaCO₃, respectively. For the first week after capture, the fish were maintained at the ambient temperature of the well water (20 ± 0.5 °C). The temperatures of two of the tanks were then increased or decreased by 1.0 °C per day until they reached temperatures of 26 or 10 (± 0.5) °C. The bass were maintained at these three treatment temperatures for at least 4 wk prior to beginning the elimination measurements. Throughout the acclimation and experimental periods, the fish were hand-fed live minnows (*Notropis* sp.) weekly (approximately 4, 6, and 8% of their wet mass at 15, 20, and 26 °C, respectively). These minnows contained global fallout concentrations of ¹³⁷Cs not exceeding 1 Bq kg⁻¹ wet mass (more than three orders of magnitude lower concentration than the Pond B bass).

Whole-body measurements

Whole-body measurements of the live bass were made on a 15.2-cm-wide × 10.2-cm-high NaI(Tl) solid scintillation detector/photomultiplier and a Canberra multichannel analyzer.⁹ Prior to each measurement, each fish and its cage was removed from the tank and immersed in an aerated 19-L bucket containing approximately 15 L of 200 mg L⁻¹ MS-222 for 2–3 min, until the fish's behavior indicated that stage II anesthesia had been attained (Schultz et al. 1996). The fish was then removed from its cage, weighed, and transferred to a polyethylene freezer container containing 3 L of a 50% dilution of the anesthetic solution. To ensure a uniform counting geometry, the fish was held in position inside the container with sponges and counted with its right side lateral to the top of the detector (Fig. 1). Each fish was counted for 300 s and then returned to the tank. The total time spent out of the tank was approximately 15 min for each measurement. To ensure comparable accuracy in measuring elimination at the different temperatures, the interval between elimination measurements was increased with decreasing temperature, i.e., approximately every 9, 14, and 21 d for fish at 26, 20, and 15 °C, respectively.

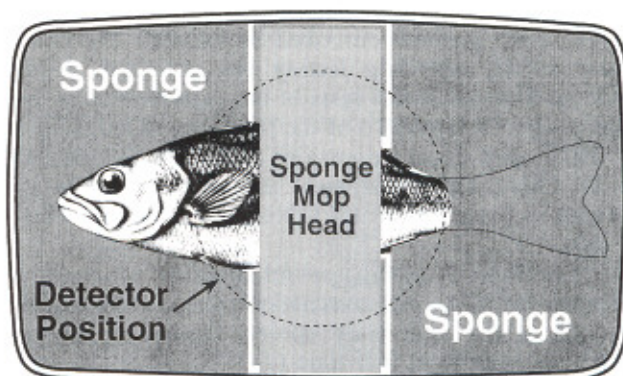


Fig. 1. Diagram of the counting geometry used to estimate ¹³⁷Cs activity of largemouth bass. The fish were held in 18-L cages floating in three 1,090-L aquaria maintained at 15, 20, or 26 °C. At intervals, the cages containing the fish were removed from the aquaria and immersed in an anesthetic solution. The lightly anesthetized fish were placed in a polyethylene freezer container containing 3 L of anesthetic solution and held in place with sponges and a plastic sponge mop head (additional sponges were placed between the left side of the fish and the roof of the container). The container containing the fish was placed on top of a NaI(Tl) solid scintillator detector (dashed line), with the right side of the fish against the sensitive volume, and whole-body counted for 300 s to measure whole-animal ¹³⁷Cs activity.

The background count rate (s⁻¹) was determined by counting uncontaminated bass (collected off the SRS) in the same geometry for the same length of time. An earlier study (Peters and Brisbin 1988) had determined that contributions by natural background radionuclides of similar energies to the ¹³⁷Cs background (especially from the 609-keV gamma photon of ²¹⁴Pb at secular equilibrium with naturally-occurring radium in the skeleton) were negligible for aquatic turtles (*Trachemys scripta*) from this region. These turtles have a much greater percentage of bone (up to 40% of body mass) than do teleosts, and it was therefore assumed that the same was true for the bass. It was also assumed that the fish were not cross-contaminating each other with their excreted ¹³⁷Cs, as uncontaminated catfish maintained in closed tanks together with fish labeled with much greater amounts of ¹³⁷Cs did not show detectable uptake after 10 wk.[#]

The background count rate was subtracted from the gross count rate (the count rate of the Pond B fish plus natural background) to obtain the net count rate (s⁻¹) for each individual. The net count rates for the amount of radionuclide lost due to physical decay since the beginning of the measurement period were corrected by dividing them by $e^{-k_p t}$, where k_p is the fraction of the total radioactivity lost per unit time (defined as $\ln 2/T_p$, where T_p is the radioactive half-time of ¹³⁷Cs), and t represented the elapsed time since the beginning of the study. The corrected net count rates, R_{net} , were then regressed against time. The appropriateness of single and

⁸ Rubbermaid Commercial Products, Inc., Winchester, VA.

⁹ LS 700, Frigid Units, Toledo, OH.

¹¹ Canberra, Meriden, CT.

[#] E. L. Peters, unpublished data.

multiple-exponential models in fitting the data was judged using methods described by Landlaw and DiStefano (1984), including testing the normality of the residual distributions, *F*-tests of the multiple-partial correlation coefficients (*R*²) of the functions (Kleinbaum et al. 1988), and visual inspections of the fits (Boxenbaum et al. 1974). Other statistical procedures were as described in Sokal and Rohlf (1969) and Steel and Torrie (1980).

The slopes of elimination relationships were used to estimate the biological elimination rate constant (*k_b*, the fraction of the total body burden eliminated per unit time) and biological half-time (*T_b* = ln 2/*k_b*). To minimize error in estimating *k_b*, twelve whole-body measurements were made of each fish over the duration of the measurement period (Peters and Brisbin 1988).

Count yield

To estimate the count yield for whole-body (*Y_B*) and muscle (*Y_M*), both in counts s⁻¹ Bq⁻¹ ¹³⁷Cs, 34 Pond B bass (85 to 534 g) were captured (16 during 27 September to 26 October 1995, 18 during 30 May to 6 June 1996). These fish were killed immediately upon collection and whole-body measurements of these fish were subsequently made using the same procedures as for the live bass. An approximately 20-g white muscle sample was then removed and counted to a 1% 2-σ error in an automated 7.6-cm-wide × 8.3-cm-high well-type NaI(Tl) gamma counter.** The muscle sample was then homogenized, together with the remaining carcass, and sufficient water was added to dilute the homogenate to a 1-L volume. The homogenate was then counted in the same geometry as a 1-L ¹³⁷Cs standard (*A_{std}* = 3.162 kBq, which was almost identical to the activity concentration of the fish), using a 900 s count time for both. The count yield for the whole fish was estimated from the count yield of the standard (*Y_{std}*, counts s⁻¹ Bq⁻¹), i.e.

$$Y_{std} = \frac{R_{net(std)}(s^{-1})}{A_{std}(Bq)} \quad (1)$$

$$\text{Whole fish activity } (A_{fish}, Bq) = \frac{R_{net(homogenate)}(s^{-1})}{Y_{std}} \quad (2)$$

$$Y_B(\text{counts s}^{-1} \text{ Bq}^{-1}) = \frac{R_{net(whole fish)}(s^{-1})}{A_{fish}(Bq)} \quad (3)$$

Similar procedures were used to obtain the muscle count yield from the muscle samples. The whole-body and muscle count yields were plotted against fish body mass, and the appropriate regression functions for estimating the whole-body and muscle ¹³⁷Cs activities of live bass from their whole-body *R_{net}* measurements were determined using methods described above.

Time-dependent muscle concentrations

At the end of the elimination experiment, the bass were killed by overdose of MS-222 and the ¹³⁷Cs concentrations of muscle and whole body activities were determined using the methods described above. The ratio of muscle to whole-body ¹³⁷Cs concentrations was compared with those of the bass used to estimate the muscle and whole body count yield. The otoliths of the elimination treatment fish were removed and examined to determine their approximate ages.

RESULTS

By the end of the acclimation period and the beginning of the elimination measurements, the body masses of the bass increased by at least 15% over their mass at capture. By regulating food intake, we were able to maintain the bass at relatively constant mass during the measurement period (Table 2). The body masses of the bass increased only slightly during the elimination measurements, with a mean (95% CI) percent increase of 1.6% (1.3 to 1.9%) at 15 °C, 1.1% (0.9 to 1.4%) at 20 °C, and 1.1% (0.9 to 1.4%) at 26 °C (*n* = 12 measurements per fish in all cases).

¹³⁷Cs elimination

By adjusting the measurement interval for the temperature of the fish, it was possible to maintain a consistent fraction of decrease in whole-body burdens (about 4% between each measurement) for each group of fish. After twelve counts (231 d at 15 °C, 154 d at 20 °C, and 92 d at 26 °C), the whole-body activity of all three groups had declined to 62% (95% CI: 60 to 65%) of the first body burden measurement taken at the end of the acclimation period (50 to 55% of the body burden at the time of capture).

Based on the results of the statistical criteria described above, whole body elimination from each individual bass was best described by a single-component exponential model, and elimination data for all bass were therefore fitted to the model: ¹³⁷Cs concentration (Bq, corrected for physical decay) = *a e^{β time (d)}* (Table 3). The predictive power of each individual fish's elimination equation was very high (all *r*² > 0.90), and the β values for each regression were used to estimate *k_b*. The model *k_b* = *a e^{β T} m^c* [where *T* was the temperature (°C), and *m* was the average body mass (g) of each fish] was then fitted to the estimated half-time from all bass. This model was highly significant (*F*_{2,25} = 150.4, *p* < 0.0001), and had a high degree of predictability (*R*² = 0.92). The parameter estimates for this model were *a* = 2.71 × 10⁻³ (95% CI: 1.08 × 10⁻³ to 6.73 × 10⁻³), *b* = 0.070 (95% CI: 0.061 to 0.080) and *c* = -0.224 (95% CI: -0.370 to -0.077). The *T_b*'s estimated from these *k_b* values averaged 322 d (95% CI: 311 to 333 d) at 15 °C, 225 d (95% CI: 220 to 230 d) at 20 °C, and 140 d (95% CI: 137 to 143 d) at 26 °C (Fig. 2). Across the range of size and temperatures used in this study, the difference between our observations and the predicted values of Rowan and Rasmussen (1995) increased with both

** Packard Auto-gamma® 5530, Packard Instrument Company, Meriden, CT.

Table 2. Growth of caged largemouth bass during ^{137}Cs elimination measurements.

Fish	Body mass (g)		% Change	% Change d^{-1}
	Initial	Final		
15 °C				
1	250	269	7.60	0.033
2	253	306	20.9	0.091
3	259	278	7.34	0.032
4	279	333	19.4	0.084
5	307	346	12.7	0.055
6	333	371	11.4	0.049
7	350	380	8.57	0.037
8	468	383	-18.2	-0.079
Mean (95% C.I.)	312 (262-363)	333 (302-364)	8.72 (0.39-17.1)	0.038 (0.002-0.074)
20 °C				
9	141	154	9.22	0.055
10	152	188	23.7	0.142
11	174	218	25.3	0.151
12	223	244	9.42	0.056
13	239	292	22.2	0.133
14	263	277	5.32	0.032
15	288	326	13.2	0.079
16	296	311	5.07	0.030
17	313	343	9.58	0.057
18	320	354	10.6	0.064
Mean (95% C.I.)	241 (200-282)	271 (229-313)	13.4 (8.67-18)	0.08 (0.052-0.108)
26 °C				
19	130	136	4.62	0.050
20	176	209	18.8	0.204
21	185	189	2.16	0.024
22	243	262	7.82	0.085
23	252	263	4.37	0.047
24	275	289	5.09	0.055
25	290	331	14.1	0.154
26	291	304	4.47	0.049
27	336	334	-0.60	-0.006
28	448	471	5.13	0.056
Mean (95% C.I.)	263 (207-318)	279 (221-336)	6.60 (3.04-10.1)	0.072 (0.033-0.110)

temperature and body mass: T_b 's were 1.7 to 2.5 \times longer than would be expected for acutely contaminated (non steady-state) fish and 1.2 to 1.8 \times longer than those predicted for steady-state conditions (Fig. 3).

As noted in previous studies (e.g., Ugedal et al. 1992), temperature exerted a much greater influence ($F_{1,25} = 227.6$, $p < 0.0001$) on T_b 's than did body mass ($F_{1,25} = 9.805$, $p < 0.0044$). The Q_{10} for ^{137}Cs elimination (also estimated from the k_b values) was 2.14, and the 95% CI for this value (1.93 to 2.37) enclosed the expected value of 2 based on the presumed influence of physical, chemical, and biological processes on metabolic rate. Including the age of the fish, as determined from otolith annuli, did not significantly improve the fits of any of the regression models tested.

Count yield

There was no significant seasonal difference between the whole-body ^{137}Cs concentrations of the 34 bass used in the muscle and whole-body count yield determinations. These data were therefore combined, and a significant and highly predictive relationship between body mass and whole-body ^{137}Cs activity (A_{fish} , Bq) was observed, with $A_{\text{fish}} = a m^b$ ($F_{1,32} = 179.5$, $p < 0.0001$, $r^2 = 0.85$), where m was the wet mass of the fish (g), a

was 5.997 (95% CI: 2.878 to 12.50), and b was 0.866 (95% CI: 0.734 to 0.997). The mass-dependent whole-body count yield (Y_B) was best estimated by an exponentially decreasing function (Fig. 4): $Y_B = ae^{bm}$ ($F_{1,32} = 82.85$, $p < 0.0001$, $r^2 = 0.71$), where m was the wet mass of the fish (g), a was 0.119 (95% CI: 0.112 to 0.126), and b was -8.6×10^{-4} (95% CI: -1.06×10^{-3} to -6.71×10^{-4}). The mass-dependent muscle count yield (Y_M) was best estimated by a power function (Fig. 4): $Y_M = a m^b$ ($F_{1,32} = 601.3$, $p < 0.0001$, $r^2 = 0.95$), where m was the wet mass of the fish (g), a was 3.16 (95% CI: 2.24 to 4.44), and b was -0.739 (95% CI: -0.800 to -0.677).

Time-dependent muscle concentrations

The muscle to whole-body ^{137}Cs ratios of the three elimination groups were indistinguishable (Scheffe's S tests, all p -values > 0.43) from those of the freshly-caught bass (Fig. 5). As mentioned above (and depicted in Fig. 2), the whole-body ^{137}Cs of all three elimination groups had decreased approximately 38% from their post-acclimation period levels at the time that these ratios were measured (at least a 40-50% decline from their body burdens at capture).

Table 3. Elimination of ¹³⁷Cs by caged largemouth bass. All bass were fitted to the model: ¹³⁷Cs Concentration (Bq, corrected for physical decay) = $\alpha e^{\beta \text{time(d)}}$. The elimination rate constant $k_b = \beta$. *P*-values for all regression model fits were ≤ 0.0001 .

Fish	¹³⁷ Cs concentration (Bq g ⁻¹)		% ¹³⁷ Cs remaining	k_b (95% C.I.) $\times 10^{-3}$, d ⁻¹	T_b (95% C.I.), d	$F_{1,10}$	r^2
	Initial	Final					
15 °C (231 d)							
1	2.59	1.53	59.1	2.01 (1.68–2.34)	345 (296–413)	181.2	0.95
2	1.54	0.82	53.0	2.44 (2.02–2.86)	284 (242–343)	165.9	0.94
3	2.08	1.17	56.4	2.30 (2.00–2.60)	301 (267–347)	295.4	0.97
4	2.55	1.27	49.9	2.27 (2.04–2.51)	305 (276–340)	474.5	0.98
5	1.84	1.05	57.3	2.3 (1.82–2.78)	301 (249–381)	114.9	0.92
6	3.51	1.95	55.5	2.32 (2.1–2.54)	299 (273–330)	556.1	0.98
7	2.62	1.69	64.4	1.63 (1.12–2.15)	425 (322–619)	90.56	0.91
8	2.48	1.73	69.7	2.05 (1.81–2.28)	338 (304–383)	367.2	0.97
Mean (95% C.I.)	2.40 (1.99–2.82)	1.40 (1.14–1.67)	58.2 (53.8–62.6)	2.17 (1.99–2.34)	320 (296–349)		
20 °C (167 d)							
9	2.84	1.23	43.4	4.00 (3.04–4.96)	173 (140–228)	86.04	0.90
10	2.03	1.07	52.7	2.60 (2.03–3.17)	267 (219–341)	102.4	0.91
11	2.04	0.99	48.4	3.10 (2.67–3.54)	224 (196–260)	250.4	0.96
12	1.96	1.10	55.8	3.12 (2.75–3.49)	222 (199–252)	354.2	0.97
13	2.07	1.16	56.1	2.92 (2.37–3.48)	237 (199–292)	138.4	0.93
14	2.61	1.42	54.2	3.28 (2.89–3.66)	211 (189–240)	361.5	0.97
15	1.90	1.03	54.3	2.84 (2.45–3.24)	244 (214–283)	256.6	0.96
16	1.78	1.03	58.0	3.11 (2.76–3.45)	223 (201–251)	392.9	0.98
17	1.97	1.11	56.1	3.03 (2.65–3.41)	229 (203–262)	315.4	0.97
18	1.65	0.90	54.7	3.14 (2.9–3.39)	221 (204–239)	809.7	0.99
Mean (95% C.I.)	2.09 (1.86–2.31)	1.1 (1.02–1.19)	53.4 (50.7–56.1)	3.07 (2.97–3.16)	226 (219–233)		
26 °C (92 d)							
19	2.61	1.68	64.3	3.28 (2.89–3.66)	211 (189–240)	90.7	0.90
20	2.62	1.45	55.2	5.52 (4.83–6.20)	126 (112–144)	361.5	0.97
21	2.44	1.46	59.8	5.48 (4.39–6.57)	126 (106–158)	125.8	0.93
22	2.39	1.39	58.0	5.21 (4.30–6.12)	133 (113–161)	163.2	0.94
23	2.45	1.44	58.9	5.01 (4.15–5.87)	138 (118–167)	168.1	0.94
24	2.52	1.66	66.1	4.78 (4.33–3.66)	145 (189–160)	563.8	0.98
25	2.52	1.57	62.3	4.55 (4.08–5.02)	152 (138–170)	461.1	0.98
26	2.32	1.20	51.6	4.51 (3.45–5.16)	154 (134–201)	124.8	0.93
27	3.36	2.02	60.3	4.41 (3.68–5.14)	157 (135–188)	180.6	0.95
28	1.77	1.20	67.8	5.73 (4.79–6.67)	121 (104–145)	223.6	0.97
Mean (95% C.I.)	2.50 (2.26–2.74)	1.51 (1.35–1.66)	60.4 (57.4–63.5)	4.96 (4.63–5.29)	140 (131–150)		

DISCUSSION

The cesium T_b 's observed in this study were at least twice as long as would be expected for acutely contaminated fish, and were longer than those predicted by the steady-state model of Rowan and Rasmussen (1995). In that model, the authors estimated that fish at steady-state with the environment eliminated cesium at approximately 72% of the rate of fish that were not at steady state. The primary difference between these predictions and the results of this study appeared to be in the value of the intercept of the response surface, rather than in the regression parameters for the dependent variables (which were statistically indistinguishable from the same parameters estimated in earlier models). The combined difference in these parameters is nonetheless important: the mean k_b 's predicted from the Rowan and Rasmussen model exceeds the mean k_b 's predicted by our model in every case (which is, as should be expected, not the case for our fitted model). In most cases, the 95% CIs for k_b of the model of Rowan and Rasmussen barely include the upper 95% confidence limit of the observed k_b values. For 22 of 28 fish, the lower bound of the 95% CI of the

Rowan and Rasmussen estimates are greater than the mean observed k_b value. The difference in the two models is accentuated when these k_b values are converted into half-times. Thus, even where the full confidence limits of the predictions of the Rowan and Rasmussen model are taken into account, the result is a statistically significant overestimate of k_b (underestimate of T_b). Comparisons with this model should be made cautiously, however, in that the data that comprise this model were obtained from only five studies (almost all of which were cold-water fish), using the unweighted mean values for temperature/mass groups with unequal numbers of individuals, and with one study of brown trout (Ugedal et al. 1992) accounting for a majority of the model data.

Hewett and Jeffries (1976) reported that decreased radiocesium T_b 's could be induced as a result of handling stress. This fact was used to justify the protocol used by Ugedal et al. (1992), i.e., serial sampling from a cohort of acutely-dosed brown trout to estimate ¹³⁴Cs elimination. If such a stress-related decrease in T_b were the case in this study, then the actual T_b 's would be even longer than

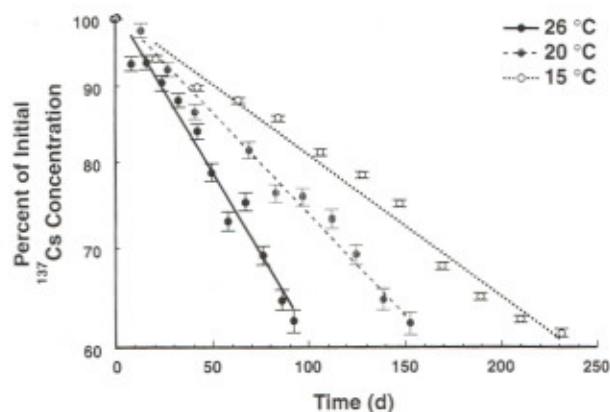


Fig. 2. Whole-body elimination of ^{137}Cs from largemouth bass at 15 °C ($n = 10$, counting interval 21 d), 20 °C ($n = 8$, counting interval 14 d), or 26 °C ($n = 10$, counting interval 8 d). Symbols indicate the means for each group, error bars indicate the 95% CIs for the means. Regression lines are fits to all individual data points within each treatment group.

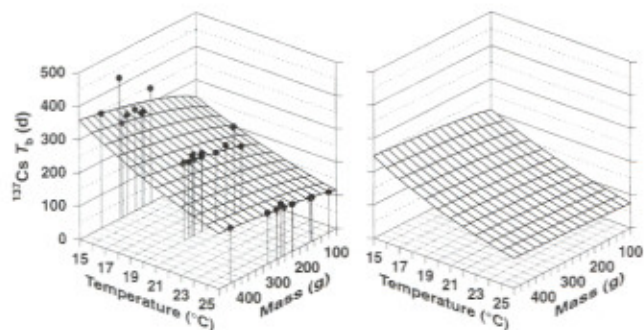


Fig. 3. Comparison of observed measurements (●) of ^{137}Cs elimination half-times (T_b , d) of largemouth bass at 15, 20, and 26 °C and the response surface for predicted T_b as a function of temperature and body mass (left) with the predicted elimination (Rowan and Rasmussen 1995) for chronically-contaminated (steady-state) fish at equilibrium with their environment (right).

those observed. The fish in our study appeared to rapidly become accustomed to the minimal handling required, however, and would feed readily as soon as 15 min after recovery from anesthesia. It is also unlikely that potential effects of repeated anesthesia on metabolic rate were responsible for increasing the T_b 's of the fish in this study. Acutely-dosed channel catfish (*Ictalurus punctatus*) maintained under nearly identical conditions to this study had ^{137}Cs T_b 's that were about 50% as long as the bass in this study,^{††} which is nearly identical to the T_b 's expected for nonsteady-state fish (Rowan and Rasmussen 1995).

The estimated power coefficient of body mass was fairly broad, and undoubtedly reflects the relatively small size range (0.325 orders of magnitude) of the fish in this study. The 95% CI for our mass coefficient (-0.370 to -0.077) enclosed both the value of -0.176 obtained by

^{††} E. L. Peters, unpublished data.

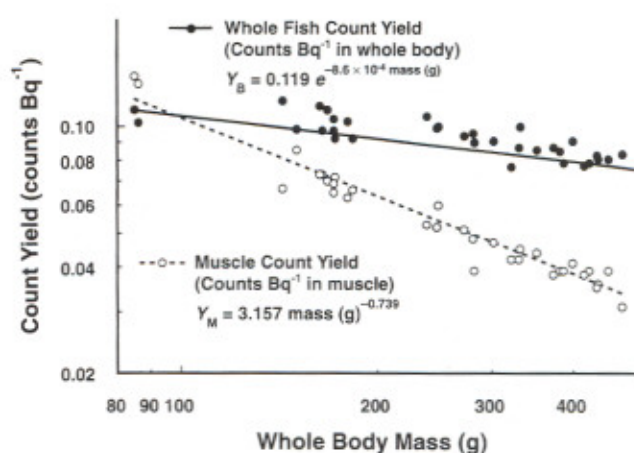


Fig. 4. Mass-dependent whole-body and muscle count yields of largemouth bass.

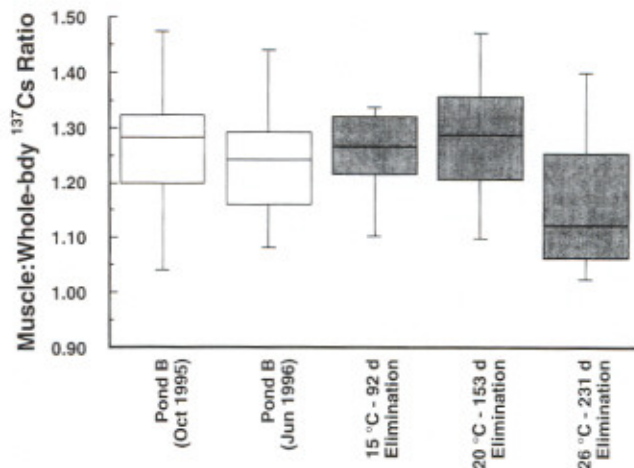


Fig. 5. Muscle to whole-body ratios of ^{137}Cs in largemouth bass. The figure compares two groups of freshly-captured bass from Pond B (sampled during autumn and spring, unshaded boxes) with three groups of Pond B bass held at 15, 20, and 26 °C (shaded boxes). At the time that the measurements were made, all three of the elimination groups had eliminated at least 40% of their initial ^{137}Cs whole-body burdens.

Ugedal et al. (1992) for brown trout over a 1.3 order of magnitude size range and the value of -0.111 obtained by Rowan and Rasmussen (1992) over a greater (3.6 orders of magnitude) size range.

Earlier studies (Baptist and Price 1962; Hewett and Jeffries 1976, 1978) found that radiocesium uptake by fish muscle was slower than for other tissues. Muscle also comprises the largest tissue compartment for cesium after acute intravascular dosage (about 80% in channel catfish).^{‡‡} If cesium elimination kinetics from muscle were comparable to uptake, then this would be consistent with the hypothesis (Rowan and Rasmussen 1995) that slower loss from muscle by chronically-contaminated

^{‡‡} E. L. Peters, unpublished data.

fish might account for their longer whole-body ¹³⁷Cs T_b 's. It is therefore reasonable to expect that the ¹³⁷Cs muscle to whole-body ratio of the bass in this study should also have increased, given the longer elimination time. However, the results of this study indicate that the elimination rate from bass muscle was indistinguishable from that of the body as a whole. None of the elimination treatment fish showed any significant differences in their muscle to whole-body ratios compared with freshly-caught bass (Fig. 5). In fact, the only discernible difference was a (non-significant) decrease in this ratio in the 26 °C group. This suggests that the kinetics of ¹³⁷Cs in muscle dominate the kinetics of the body as a whole, and that potential effects of slow incorporation of radiocesium into skeleton or other connective tissues (which exchange with body tissue fluids only at limited sites) might be less likely to be responsible for this effect. As the fish grows larger and bone thickness increases, a concomitant decrease in bone surface-to-volume ratio would be expected, resulting in an increase in T_b in these tissues. However, the potential size of these compartments appears to be quite small, so unless elimination rates are extremely slow (perhaps increasing with age as increasingly greater quantities of ¹³⁷Cs become sequestered), they are unlikely to exert a discernible influence on apparent elimination until nearly all of the fish's soft tissue burden has been eliminated. If this is the case, then chronically-contaminated fish such as those found in Pond B are likely to retain some ¹³⁷Cs (albeit mostly in inedible tissues) for their entire lives, even after muscle concentrations decline to background levels.

Our results suggested that the conditions of acute vs. chronic exposure might result in significant and systematic differences in the apparent ¹³⁷Cs elimination half-times of fish. These differences appear to persist even when care is taken to thoroughly acclimate the animals and to delay the onset of elimination measurements to avoid the periods of rapid elimination characteristic of acute doses. It is therefore difficult to determine whether observed differences in ¹³⁷Cs elimination rates are the result of experimental conditions or reflect species-specific elimination rates. In either case, the explanation for differences in ¹³⁷Cs retention would appear to lie within physiological mechanisms of uptake, compartmentalization, and elimination within the animal. These processes may include disequilibria between the uptake and loss of ¹³⁷Cs into cells by active and passive transport mechanisms (e.g., Na⁺/K⁺ ATPases, gated and leakage channels for K⁺ in cell membranes), or between uptake and retention of intracellular ¹³⁷Cs in membrane-bound organelles. Understanding these processes may provide insight into uptake and elimination kinetics of the whole organism and may lead to more general predictions of ¹³⁷Cs behavior in fish in the environment, thus decreasing the need for repetitive empirical studies to assess the effects of environmental releases on these animals.

Acknowledgments—This research was supported by Financial Assistance Award Number DE-FC09-96SR18546 from the U.S. Department of Energy to the University of Georgia Research Foundation. We gratefully acknowledge the assistance of J. McCloskey, D. Nestle, D. Sugg, and W. Stephens. L. S. Hales, Jr., provided the otolith analyses. This research was conducted under the approval of the Animal Care and Use Committee of the University of Georgia. The bass were collected under Permit No. G-95-03, issued by the South Carolina Wildlife & Marine Resources Department.

REFERENCES

- Alberts, J. J.; Bowling, J. W.; Orlandini, K. A. The effect of seasonal anoxia on the distribution of ²³⁸Pu, ^{239,240}Pu, ²⁴¹Am, ²⁴⁴Cm, and ¹³⁷Cs in pond systems of the southeastern United States. In: Pinder, J. E.; Alberts, J. J.; McLeod, K. W.; Schreckhise, R. G., eds. Environmental research on actinide elements. U.S. Department of Energy CONF-841142; 1987.
- Anspaugh, L. R.; Catlin, R. J.; Goldman, M. The global consequences of the Chernobyl reactor accident. *Science* 242:1513-1519; 1988.
- Ashley, L. R.; Zeigler, C. C. Releases of radioactivity at the Savannah River Plant, 1954 through 1978. Aiken, SC: E. I. DuPont deNemours, Savannah River Laboratory; DPSPU 75-25-1; 1980.
- Baptist, J. P.; Price, T. J. Accumulation and retention of cesium-137 by marine fishes. U.S. Fish and Wildlife Service Fish. Bull. 206:177-187; 1962.
- Boxenbaum, H. G.; Riegelman, S.; Elashoff, R. M. Statistical estimations in pharmacokinetics. *J. Pharmacokin. Biopharmacol.* 2:123-149; 1974.
- Carlsson, S. A model for the turnover of ¹³⁷Cs and potassium in the pike (*Esox lucius*). *Health Phys.* 35:549-554; 1978.
- Evans, S. Application of parameter uncertainty analysis to accumulation of ¹³⁷Cs in fish, with special emphasis on *Pleuronectes platessa* L. *J. Exp. Mar. Biol. Ecol.* 120:57-80; 1988.
- Fagerström, T. Body weight, metabolic rate, and trace substance turnover in animals. *Oecologia* 29:99-104; 1977.
- Forseth, T.; Jonsson, B.; Numa, R.; Ugedal, O. Radioisotope method for estimating food consumption by brown trout (*Salmo trutta*). *Can. J. Fish. Aquat. Sci.* 49:1328-1335; 1992.
- Forseth, T.; Ugedal, O.; Jonsson, B.; Langeland, A.; Njåstad, O. Radiocesium turnover in Arctic charr (*Salvelinus alpinus*) and brown trout (*Salmo trutta*) in a Norwegian lake. *J. Appl. Ecol.* 28:1053-1067; 1991.
- Gallegos, A. F.; Whicker, F. W. Radiocesium retention by rainbow trout as affected by temperature and weight. In: Nelson, D. J., ed. Radionuclides in ecosystems: Proceedings of the Third National Symposium on Radioecology. Springfield, VA: National Technical Information Service; U.S. Atomic Energy Commission Symposium Series; CONF-710501-PI; 1971.
- Häsänen, E.; Kolehmainen, S.; Miettinen, J. K. Biological half-time of Cs-137 in three species of freshwater fish: perch, roach, and rainbow trout. In: Åberg, B.; Hungate, F. P., eds. Radioecological concentration processes, Proceedings of an International Symposium, Stockholm. Oxford: Pergamon Press; 1967.
- Hewett, C. J.; Jeffries, D. F. The accumulation of radioactive caesium from water by the brown trout (*Salmo trutta*) and its comparison with plaice and rays. *J. Fish Biol.* 9:479-489; 1976.

- Hewett, C. J.; Jeffries, D. F. The accumulation of radioactive caesium from food by the plaice (*Pleuronectes platessa*) and the brown trout (*Salmo trutta*). *J. Fish Biol.* 13:143-153; 1978.
- Kelly, M. S.; Pinder, J. E., III. Foliar uptake of ^{137}Cs from the water column by aquatic macrophytes. *J. Environ. Radioact.* 30:271-280; 1995.
- Kevern, N. R. Feeding rate of carp estimated by a radioisotopic method. *Trans. Amer. Fish. Soc.* 95:363-371; 1966.
- Kleinbaum, D. G.; Kupper, L. L.; Muller, K. E. Applied regression analysis and other multivariate methods. Second edition. Boston, MA: PWS-KENT Publishing Company; 1988.
- Kolehmainen, S. E. The balances of Cs-137, stable cesium, and potassium of bluegill (*Lepomis macrochirus*) and other fish in White Oak Lake. *Health Phys.* 23:301-315; 1972.
- Korhonen, R. Modeling transfer of ^{137}Cs fallout in a large Finnish watercourse. *Health Phys.* 59:443-454; 1990.
- Koulikov, A. O.; Ryabov, I. N. Specific cesium activity in freshwater fish and the size effect. *Sci. Tot. Environ.* 112:125-142; 1992.
- Kryshev, I. I. Radioecological consequences of the Chernobyl accident. Moscow: Nuclear Society International; 1992.
- Landlaw, E. M.; DiStefano, J. J. Multiexponential, multicompartamental, and noncompartmental modeling II: data analysis and statistical considerations. *Am. J. Physiol.* 246:R665-R667; 1984.
- Mailhot, H.; Peters, R. H.; Cornett, R. J. The biological half-time of radioactive Cs in poikilothermic and homeothermic animals. *Health Phys.* 56:473-484; 1989.
- Medvedev, Z. A. The legacy of Chernobyl. New York: W. W. Norton; 1990.
- Peters, E. L.; Brisbin, I. L., Jr. Radiocaesium elimination in the yellow-bellied turtle (*Pseudemys scripta*). *J. Appl. Ecol.* 25:461-471; 1988.
- Peters, E. L.; Brisbin, I. L., Jr. Environmental influences on the ^{137}Cs kinetics of the yellow-bellied turtle (*Trachemys scripta*). *Ecol. Monogr.* 66:115-136; 1996.
- Pinder, J. E., III; Alberts, J. J.; Bowling, J. W.; Nelson, D. M.; Orlandini, K. A. The annual cycle of plutonium in the water column of a warm, monomictic reservoir. *J. Environ. Radioact.* 17:59-81; 1992.
- Rowan, D. J.; Rasmussen, J. B. The elimination of radiocaesium from fish. *J. Appl. Ecol.* 32:739-744; 1995.
- Schultz, I. R.; Peters, E. L.; Newman, M. Toxicokinetics and disposition of inorganic mercury and cadmium in channel catfish after intravascular administration. *Toxicol. Appl. Pharmacol.* 140:39-50; 1996.
- Sokal, R. R.; Rohlf, F. J. Biometry. San Francisco, CA: W. H. Freeman and Company; 1969.
- Steel, R. G. D.; Torrie, J. H. Principles and procedures of statistics: a biometrical approach. Second edition. New York: McGraw-Hill; 1980.
- Ugedal, O.; Jonsson, B.; Njåstad, O.; Næumañ, R. Effects of temperature and body size on radiocesium retention in brown trout, *Salmo trutta*. *Freshwater Biol.* 28:165-171; 1992.
- Whicker, F. W.; Schultz, V. Radioecology: nuclear energy and the environment. Volume II. Boca Raton, FL: CRC Press, Inc.; 1982.
- Whicker, F. W.; Hinton, T. G.; Niquette, D. J.; Seel, J. Health risks to hypothetical residents of a radioactively contaminated lake bed. In: ER '93—Meeting the Challenge: Proceedings of the Environmental Remediation Conference at Augusta, Georgia, October 24-28 1993, U.S. Department of Energy, Savannah River Operations Office, Augusta, Georgia; Vol. 1; 1993.
- Whicker, F. W.; Pinder, J. E., III; Bowling, J. W.; Alberts, J. J.; Brisbin, I. L., Jr. Distribution of long-lived radionuclides in an abandoned reactor cooling reservoir. *Ecol. Monogr.* 60:471-496; 1990.

