

Bioaccumulation of Radiocesium by Fish: the Influence of Physicochemical Factors and Trophic Structure

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Although many measurements have been made on radiocesium levels in water and aquatic biota, no agreement has been reached regarding the factors affecting bioaccumulation of these radionuclides. With monitoring data from countries that operate nuclear facilities and data from the primary literature, we explored the chemical and ecological factors that determine the bioaccumulation of radiocesium. Using log-linear regression we found that the bioaccumulation of ^{137}Cs by fish was a negative function of both dissolved potassium and suspended sediment concentration, and a positive function of temperature. Important ecological factors were the trophic level of the fish (piscivores bioaccumulate more than planktivores and benthivores), and the length of the food chain as reflected by the ratio of piscivore yield relative to net primary production. Fish from softwater drainages, which make up a large portion of northern Europe and Canada, are more vulnerable to radiocesium contamination than fish from hardwater sedimentary drainages, because these waters are extremely low in potassium and suspended sediment, and their watersheds are less efficient in retaining radiocesium. High dissolved potassium, short food chains, and the much greater volume and mixing potential of the ocean make marine fish less vulnerable to releases of radiocesium.

Bien que l'on ait effectué de nombreux dosages des concentrations de césium radioactif dans l'eau et les biotes aquatiques, on ne s'entend pas sur les facteurs en cause dans la bioaccumulation de ces radionucléides. À l'aide de données de surveillance provenant de pays qui exploitent des installations nucléaires et de données fondamentales, nous avons examiné les facteurs d'ordre chimique et écologique qui déterminent la bioaccumulation du césium radioactif. Au moyen d'une régression log-linéaire, nous avons montré que la bioaccumulation de ^{137}Cs par les poissons était une fonction négative des concentrations de potassium dissous et des sédiments en suspension et qu'elle était une fonction positive de la température. Des facteurs écologiques importants étaient le niveau trophique des poissons (la bioaccumulation est plus importante chez les piscivores que chez les planctivores et les benthivores) et la longueur de la chaîne alimentaire établie par le rapport entre le rendement des piscivores et la production primaire nette. Les poissons des eaux douces de ruissellement, qui constituent une grande partie du nord de l'Europe et du Canada, sont plus susceptibles à la contamination par le césium radioactif que les poissons des eaux dures sédimentaires de ruissellement, puisque ces eaux ont des teneurs extrêmement faibles en potassium et en sédiments en suspension et que leurs bassins versants retiennent moins efficacement le césium radioactif. La concentration élevée de potassium dissous, les courtes chaînes alimentaires ainsi que le volume et le potentiel de mélange beaucoup plus considérables des océans rendent les poissons marins moins susceptibles aux rejets de césium radioactif.

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Radiocesium isotopes are abundant fission products that decay slowly (^{137}Cs and ^{134}Cs have half-lives of 30.17 years and 2.07 years, respectively) and accumulate in biota by virtue of their high assimilation and slow clearance from living tissue (Kolehmainen 1972; Ugedal et al. 1992). Interest in these radionuclides dates back to the 1950s when aboveground nuclear weapon tests resulted in contamination of the stratosphere. Atmospheric weapon tests along with the operation of nuclear facilities such as generating stations, fuel reprocessing plants, and waste disposal areas have led to continual releases of these contaminants to the environment. The combined effect of these

emissions has resulted in both point source and global pollution. Recently, the nuclear accident at Chernobyl has renewed interest in these cesium (Cs) radioisotopes (e.g., Hakanson et al. 1992; Elliott et al. 1992). Although many measurements have been made on radiocesium levels in water and in virtually any type of aquatic biota, no agreement seems to have been reached regarding the factors that affect the bioaccumulation of these radionuclides, and this lack of ecological understanding has hindered the development of robust dose models.

The fate and cycling of radioactive isotopes in ecosystems has long been a subject of vital interest to applied

ecologists and health physicists (Davis and Foster 1958; Krumholz 1956); however, it is also of fundamental significance in biogeochemistry and trophodynamics (Hutchinson and Bowen 1950; Rigler 1956; King 1964; Marples 1966; Odum 1965). Although Cs is itself not biologically essential, the pathways that cycle this element within the ecosystem, and the processes that lead to its bioconcentration are the fundamental physiological and energetic pathways and processes that dominate ecosystem function.

While some authors have emphasized the role of dissolved cations as nonisotopic carriers (especially potassium) in reducing biouptake of radiocesium (Williams and Swanson 1958; Davis 1963; Kolehmainen et al. 1967; Preston et al. 1967), others have denied these effects (Mailhot et al. 1988). Food chain factors are equally problematic. While there is general agreement that food is the major route of radiocesium uptake by fish, some authors claim varying degrees of biomagnification (Gustafson 1967; Eyman and Kevern 1975; Hammar et al. 1991), some deny food-chain effects (Nelson 1969; Reichle et al. 1970; Kolehmainen 1972; Thomann 1981), and some even claim that radiocesium undergoes "biodiminution" in aquatic food chains (Mailhot et al. 1988). Clearly both the biochemical and ecological aspects of radiocesium biouptake remain shrouded in controversy.

Although the problem of radiocesium bioaccumulation is of major ecological and radiological importance, its analysis is impeded by the fact that most of the monitoring data have been collected for the purpose of calculating site-specific doses to be used by local regulatory agencies, and have not been published in the primary scientific literature. The relative inaccessibility of most of this "gray" government and industrial literature has prevented a thorough review of the subject, and those reviews that have been attempted have used only a fraction of the data that have been collected by monitoring agencies (Reichle et al. 1970; Vanderploeg et al. 1975; Blaylock 1982; Mailhot et al. 1988). The purpose of this study was to collect as much of the monitoring data as possible from countries that operate nuclear facilities, to combine these with the information available in the primary literature, and, using statistics, to explore the chemical and ecological factors that determine the bioaccumulation of radiocesium. The empirical models generated in this analysis can be used to evaluate the potential contribution to risk associated with the consumption of fish from sites near nuclear reactors. Although most of the radiocesium monitoring data that we have compiled in this study come from sites near nuclear power stations, there are sufficient data available from remote sites in Canada to permit a comparison with sites near nuclear stations. These remote sites include Lake Superior, the Ottawa River, and the three basins of Great Slave Lake.

Hypotheses

Besides collecting data on radiocesium concentrations in fish and their bioaccumulation factor (relative to water), we compiled data on a number of physicochemical and ecological factors that we hypothesized to be important determinants of the partitioning of radiocesium between water and biota.

The first hypothesis that we test can be termed the "potassium carrier" hypothesis; we expect to see a reduction in the radiocesium bioaccumulation factor with increasing

potassium (K) concentration of the water. Although the primary route of radiocesium and K uptake in fish is through food, and food varies little in its K content, we expect to see this inverse relationship, because the major route of radiocesium entry into the food chain is through direct absorption by plants from water. Plants absorb cations through an active transport mechanism that clears H^+ ions from the cell, making the cell interior negative, and resulting in indiscriminate uptake of cations with radii small enough to pass through membrane channels. Thus, rubidium and Cs enter the plant cell along with the more abundant and essential K ions (MacRobbie 1974; Kochlan and Lucas 1982). While the flux of ions into the cell increases with the external concentration (Kochlan and Lucas 1982) the flux relative to the external concentration decreases, and thus the uptake of K (and therefore radiocesium) relative to its external concentration will diminish with increasing K concentration. Thus, algal cells bioaccumulate radiocesium about two to three orders of magnitude when K is about 10 μM , and by only about one order of magnitude when K is about 10 mM (Williams and Swanson 1958; Williams 1960). Thus, most of the radiocesium bioaccumulation in the food chain occurs at the primary producer level; therefore, we expect the K dilution effect seen in plants to be manifested in fish at all trophic levels.

As radiocesium is known to be efficiently assimilated (Kolehmainen 1972; Forseth et al. 1992) and cleared very slowly by fish (biological half-life in the hundreds of days; Kolehmainen 1972; Ugedal et al. 1992), most theoretical models presume that radiocesium will be biomagnified to some degree by aquatic food chains (Forseth et al. 1991). This has been well documented for a wide variety of persistent environmental contaminants with long biological half-lives such as organochlorines (Thomann 1981; Rasmussen et al. 1990; Borgmann and Whittle 1992; Rowan and Rasmussen 1992) and mercury (Borgmann and Whittle 1992; Cabana et al. 1994). Although we expect to find significantly higher radiocesium levels in piscivorous fish than in planktivorous or benthivorous fish, the effect has never been rigorously quantified by empirical analyses.

Uncharacteristically high or low bioaccumulation factors have been estimated for fish in Scandinavia in the few years following the pulse release of radiocesium from Chernobyl in 1986 (Sundblad et al. 1991). This results from the time lag associated with the clearance of a pulse through long food chains, and underscores the need to concentrate on equilibrated systems, when attempting to estimate bioaccumulation factors empirically. We have excluded from our analysis data collected in Europe after 1986, as the detailed time series required to integrate across the pulse for each fish species over time were not available. While we have restricted our analysis to "quasi-equilibrated" sites, pulses are to some extent inevitable because releases from weapon tests and from nuclear facilities change from year to year. Fish low in the food chain would tend to track such variations closely, and thus we should expect to see linear relationships (log-log slopes near 1) between planktivore radiocesium levels and levels in water (bioaccumulation of the isotope in fish relative to water (BAF) independent of water concentrations). Piscivores, on the other hand, are not likely to track variations as closely, and thus we would expect to see log-log slopes <1 ; that is BAFs estimated at high water concentrations might be somewhat lower than those estimated

at low water concentrations because of their tendency to lag environmental variation. Therefore, in spite of our exclusion of post-Chernobyl pulse data, we expect to see some evidence of disequilibrium, and we expect to see this more pronounced in piscivores than in planktivores or benthivores.

The assimilation efficiency of animals for ^{137}Cs is known to be high except when food is rich in clay minerals to which ^{137}Cs strongly absorbs (Kolehmainen 1972; Eyman and Kitchings 1975). Its clearance rate in animal tissues also decreases with increasing animal size. Thus, we would expect that steady-state tissue concentrations will generally exceed concentrations found in food except when assimilation efficiency is abnormally low, or growth rate is high enough to bring about significant growth dilution. High concentrations of suspended clay minerals in water would be expected to reduce bioaccumulation factors for planktivores and benthivores by reducing the assimilation efficiency for ^{137}Cs through direct competition between the ingested clay and the gut membrane. These hypotheses will be tested by multiple linear regression on the global data set that we have compiled from a wide range of freshwater and marine sources.

Methods

Data on Cs in fish (either muscle or whole body) and Cs in water (dissolved or total) were obtained from primary literature as well as government monitoring reports, together with our own data for the Ottawa River. Concentrations of Cs in whole fish should be slightly lower than concentrations in muscle (hard tissues tend to be lower in Cs but comprise a small proportion of body mass), but these differences should not have a significant effect on the analysis because variability among systems spans orders of magnitude. For systems with relatively low concentrations of suspended sediment, there will be little difference between dissolved and total concentrations of Cs in water (e.g., in Lake Erie, a fairly turbid system, about 90% of the ^{137}Cs is dissolved). However, for most of the turbid systems (e.g., western basin of Great Slave Lake and White Oak Lake), dissolved concentrations were reported. Most data were for ^{137}Cs measured by γ -spectrometry using NaI or Ge crystals; however, some data for stable ^{133}Cs were also included. These were measured by γ -spectrometry following neutron activation. The isotope of Cs measured is coded in Table 1. For sites with data spanning several years, we averaged both water and fish Cs values over the same time span. For freshwater sites cation chemistry was obtained from the literature (usually determined by atomic absorption spectrometry) and for most marine sites cation chemistry was estimated by multiplying salinity measurements by the mean composition of seawater given in Broecker and Peng (1982). Concentrations of suspended inorganic sediments were obtained from the literature when available, but generally were calculated using the allochthonous sediment mass-balance approach outlined by Rowan et al. (1992). Fish yields and net primary production were obtained from the literature sources identified in Table 2, and represent averages from time intervals spanning from several years to decades. A detailed discussion on the statistical aspects of the ratio of piscivore yield to primary production was given by Rowan and Rasmussen (1992). Fish were placed in trophic (piscivore

and nonpiscivore) and thermal habitat (epilimnetic vs. hypolimnetic) categories using Scott and Crossman (1973) for North American freshwater fish; Phillips and Rix (1985) and Wheeler (1969) for European freshwater fish; and Leim and Scott (1966), Carcasson (1977), Burgess and Axelrod (1973), and Smith and Heemstra (1986) for marine fishes. When lengths and/or weights were not included in the data sources, they were obtained from Bykov (1984) or the preceding sources. Mean annual temperature was obtained from climatic atlases as outlined in Rasmussen and Kalff (1987).

Data were analyzed by linear regression (SYSTAT, MGLH; Wilkinson 1989) using \log_{10} -transformed variables. To test for the effectiveness of the log transformations in removing heteroscedasticities and nonlinearities, the residuals from our models were examined to ensure that they were normally distributed about the predicted values, and that residuals were uncorrelated with the predicted Y values. No problems of this sort were found. The regressions were examined for serious collinearity effects using the condition index and the variance decomposition (Belsley et al. 1980, in Wilkinson 1989), and for significant leverage effects using the Cook statistic as outlined in SYSTAT (Wilkinson 1989). Thus, the models obtained fit the underlying assumptions of linear regression analysis.

Results and Discussion

Statistical Analyses

Because most of the data used in this analysis have not been published previously in readily accessible journals, we have included them in summary form (Tables 1–3).

The global data set (Table 1) was analyzed by log-linear regressions relating the concentration of ^{137}Cs in fish ($\text{mBq} \cdot \text{kg}^{-1}$) to concentrations of ^{137}Cs ($\text{mBq} \cdot \text{L}^{-1}$) and K ($\mu\text{M} \cdot \text{L}^{-1}$) in water for each trophic category (piscivore = 1, nonpiscivore = 0). Concentrations of ^{137}Cs in fish increased linearly with concentrations of ^{137}Cs in water, and the BAF values decreased significantly with the K concentration of the water. The BAF was significantly higher for piscivores than nonpiscivores, and the slope of the concentration of ^{137}Cs in fish relative to water was greater for nonpiscivores than for piscivores. The models obtained were

$$(1) \log[^{137}\text{Cs}]_{\text{fish}} = 3.956(0.053) + [0.915(0.027) - 0.083(0.038)(\text{trophic level})] \log[^{137}\text{Cs}]_{\text{water}} - 0.541(0.016)\log[K]_{\text{water}} + 0.382(0.054)(\text{trophic level})$$

$$R^2 = 0.876, \text{SE}_{\text{est}} = 0.305, N = 362$$

and

$$(2) \log ^{137}\text{Cs BAF} = 3.863(0.045) - 0.164(0.028)(\text{trophic level})\log[^{137}\text{Cs}]_{\text{water}} - 0.549(0.016)\log[K]_{\text{water}} + 0.488(0.043)(\text{trophic level})$$

$$R^2 = 0.817, \text{SE}_{\text{est}} = 0.309, N = 362.$$

A more detailed analysis of the effects of trophic structure of the pelagic food web and other physicochemical factors was possible for a large portion of the global data set. Although Equations (1) and (2) clearly show the importance of the trophic position of fish, the lower levels of the food web that link forage fish to primary production differ greatly among systems. This aspect of trophic structure is difficult

TABLE 1. Sampling location, fish species, trophic level, ^{137}Cs concentration of fish, ^{137}Cs and K concentration of water, and data source. (For trophic level, piscivores = 1 and nonpiscivores = 0.) * [^{133}Cs] in fish ($\mu\text{g}\cdot\text{kg}^{-1}$) and water $\text{ng}\cdot\text{L}^{-1}$).

Location and species	Trophic level	^{137}Cs fish ($\text{Bq}\cdot\text{kg}^{-1}$)	^{137}Cs water ($\text{mBq}\cdot\text{L}^{-1}$)	K ($\mu\text{m}\cdot\text{L}^{-1}$)	References ^a
<i>Freshwater</i>					
Cable Lake, Michigan, USA					
<i>Micropterus salmoides</i>	1	33.7	5.5	90	1
Clinch River, Tennessee, USA					
<i>Aplodinotus grunniens</i>	1	8.7*	25.0*	33	2
<i>Ictalurus punctatus</i>	1	4.1*	25.0*	33	2
<i>Lepomis macrochirus</i>	0	3.4*	25.0*	33	2
<i>Morone chrysops</i>	1	16.0*	25.0*	33	2
<i>Pomoxis annularis</i>	1	12.9*	25.0*	33	2
<i>Pomoxis annularis</i>	1	17.6	29.4	33	2
Dewey Lake, Michigan, USA					
<i>Micropterus salmoides</i>	1	26.3	1.9	64	1
East Twin Lake, Colorado USA					
<i>Oncorhynchus mykiss</i>	0	1988.2*	580.0*	46	3
Elbe River, Germany					
<i>Abramis brama</i>	0	2.4	8.2	512	4
<i>Anguilla anguilla</i>	1	4.7	8.2	512	4
<i>Leuciscus idus</i>	0	1.8	8.2	512	4
<i>Leuciscus rutilus</i>	0	2.6	8.2	512	4
Lake Erie, central basin, Canada–USA					
<i>Micropterus dolomieu</i>	1	2.2	0.6	33	5–8
<i>Morone chrysops</i>	1	2.1	0.6	33	5–8
<i>Oncorhynchus kisutch</i>	1	0.9	0.5	33	5–8
<i>Stizostedion vitreum</i>	1	1.0	0.6	33	5–8
Lake Erie, eastern basin, Canada–USA					
<i>Oncorhynchus mykiss</i>	1	1.4	0.3	33	5–8
Lake Erie, western basin, Canada–USA					
<i>Oncorhynchus kisutch</i>	1	1.3	0.9	31	5–8
<i>Oncorhynchus mykiss</i>	1	1.9	0.8	31	5–8
<i>Osmerus mordax</i>	0	0.5	1.3	31	8,9
<i>Perca flavescens</i>	1	1.1	1.3	31	8,9
<i>Stizostedion vitreum</i>	1	0.9	0.8	31	5–8
Fair Lake, Michigan, USA					
<i>Micropterus salmoides</i>	1	41.1	8.9	243	1
Fine Lake, Michigan, USA					
<i>Micropterus salmoides</i>	1	22.2	5.2	83	1
Georgian Bay, Ontario, Canada					
<i>Salvelinus namaycush</i>	1	10.2	1.8	18	5,10–13,60
<i>Stizostedion vitreum</i>	1	43.6	4.1	18	5,10–13,60
Gleaston Beck, UK					
<i>Salmo trutta</i>	0	2.0	7.4	102	14
Great Slave Lake, Christie Bay, Canada					
<i>Catostomus catostomus</i>	0	1.3	1.9	22	15–18
<i>Coregonus artedii</i>	0	3.1	1.9	22	15–18
<i>Coregonus clupeaformis</i>	0	1.8	1.9	22	15–18
<i>Esox lucius</i>	1	4.4	1.9	22	15–18
<i>Lota lota</i>	1	2.9	1.9	22	15–18
<i>Prosopium cylindraceum</i>	0	2.0	1.9	22	15–18
<i>Salvelinus namaycush</i>	1	5.7	1.9	22	15–18
Great Slave Lake, McLeod Bay, Canada					
<i>Coregonus artedii</i>	0	19.3	3.6	11	15–18
<i>Esox lucius</i>	1	22.9	3.6	11	15–18
<i>Prosopium cylindraceum</i>	0	6.7	3.6	11	15–18
<i>Salvelinus namaycush</i>	1	23.8	3.6	11	15–18

TABLE 1. (Continued)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (µm·L ⁻¹)	References ^a
Great Slave Lake, western basin, Canada					
<i>Coregonus artedii</i>	0	0.4	1.3	26	15,17,18
<i>Coregonus clupeaformis</i>	0	0.5	1.3	26	15,17,18
<i>Esox lucius</i>	1	1.8	1.3	26	15,17,18
<i>Lota lota</i>	1	1.1	1.3	26	15,17,18
<i>Stenodus leucichthys</i>	1	1.1	1.3	26	15,17,18
Lake Huron, northern basin, Canada–USA					
<i>Alosa pseudoharengus</i>	0	3.7	1.5	22	6,9–13,19
<i>Lota lota</i>	1	5.8	1.5	22	6,9–13,19,20
<i>Osmerus mordax</i>	0	5.9	1.5	22	6,9–13,19
<i>Perca flavescens</i>	1	9.2	1.5	22	6,9–13,19
Lake Huron, southern basin, Canada–USA					
<i>Alosa pseudoharengus</i>	0	3.7	1.8	22	6,9,10,11,19,21
<i>Ambloplites rupestris</i>	1	7.2	1.2	22	5,7,10,11,21–28
<i>Catostomus commersoni</i>	0	2.7	1.2	22	5–7,9–11,19,21,22,25,28–38
<i>Coregonus clupeaformis</i>	0	2.5	1.1	22	5,11,21,31,33
<i>Coregonus hoyi</i>	0	2.7	1.5	22	6,9,10,11,19,21
<i>Cyprinus carpio</i>	0	1.0	1.2	22	5,10,11,21,24,31
<i>Esox lucius</i>	1	4.6	1.3	22	5,7,10,11,21–23,30,32–34,37,39
<i>Ictalurus nebulosus</i>	1	6.0	1.3	22	5,7,10,11,21–27
<i>Ictalurus punctatus</i>	1	3.9	1.2	22	5,11,19,21,28,29,34,37,38
<i>Lota lota</i>	1	3.1	1.8	22	5,10,11,19,21
<i>Micropterus dolomieu</i>	1	5.4	1.1	22	5,7,10,11,19,21,23,24,26,29,31,32,34–38
<i>Morone chrysops</i>	1	3.5	1.1	22	5,11,21,38
<i>Oncorhynchus kisutch</i>	1	8.5	1.2	22	5,11,19,21
<i>Oncorhynchus mykiss</i>	1	5.2	1.8	22	10,11,21,39
<i>Perca flavescens</i>	1	8.0	1.1	22	5,7,10,11,21,23–26,28,30–33
<i>Stizostedion vitreum</i>	1	6.2	1.2	22	5,19,11,21,28–33,35–38
Inarinjarvi, Finland					
<i>Coregonus clupeaformis</i>	0	50.5	55.5	8	39–42
<i>Esox lucius</i>	1	139.6	55.5	8	39–42
<i>Lota lota</i>	1	155.8	55.5	8	40–42
<i>Perca fluviatilis</i>	1	106.2	55.5	8	39–42
<i>Salmo trutta</i>	1	145.8	55.5	8	39–42
<i>Salvelinus alpinus</i>	1	121.2	55.5	8	39–42
<i>Salvelinus salvelinus</i>	1	116.9	55.5	8	39–42
<i>Thymallus vulgaris</i>	0	55.8	55.5	8	39–42
Kytajarvi, Finland					
<i>Abramis blicca</i>	0	4.8	10.4	47	40–42
<i>Abramis brama</i>	0	9.2	10.4	47	40–42
<i>Esox lucius</i>	1	16.3	10.4	47	40–42
<i>Leuciscus rutilus</i>	0	12.4	10.4	47	40–42
<i>Lota lota</i>	1	14.1	10.4	47	40–42
<i>Lucioperca lucioperca</i>	1	28.9	10.4	47	40–42
<i>Perca fluviatilis</i>	1	11.0	10.4	47	40–42
Lupton Beck, UK					
<i>Salmo trutta</i>	0	3.7	4.0	49	14
Lower Three Runs Creek, South Carolina, USA					
<i>Aphredoderus sayanus</i>	0	213.1	296.0	36	43
<i>Esox niger</i>	1	268.8	296.0	36	43
<i>Gambusia affinis</i>	0	263.7	296.0	36	43
<i>Lepomis auritus</i>	0	394.9	296.0	36	43
<i>Lepomis marginatus</i>	0	204.5	296.0	36	43
<i>Micropterus salmoides</i>	1	829.7	296.0	36	43
<i>Notropis petersoni</i>	0	323.2	296.0	36	43

TABLE 1. (Continued)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (μm·L ⁻¹)	References ^a
Melkutin, Finland					
<i>Coregonus clupeaformis</i>	0	51.8	16.7	24	40-42
<i>Esox lucius</i>	1	253.1	16.7	24	40-42
<i>Leuciscus rutilus</i>	0	86.2	16.7	24	40-42
<i>Perca fluviatilis</i>	1	250.1	16.7	24	40-42
Lake Michigan, USA					
<i>Alosa pseudoharengus</i>	0	5.9	2.5	29	6,7,9,12,13,19, 44-50
<i>Catostomus commersoni</i>	0	8.7	3.4	29	12,13,44,46,48-50
<i>Coregonus clupeaformis</i>	0	8.9	2.2	29	12,13,45,47-50
<i>Coregonus hoyi</i>	0	8.9	2.5	29	6,7,9,12,13,19,44-50
<i>Cottus cognatus</i>	0	7.2	2.0	29	6,7,9,12,13,19,45,47-50
<i>Cyprinus carpio</i>	0	8.5	3.4	29	12,13,44,46,48-50
<i>Micropterus dolomieu</i>	1	13.7	3.4	29	12,13,44,46,48-50
<i>Oncorhynchus kisutch</i>	1	18.6	2.8	29	12,13,44-50
<i>Oncorhynchus mykiss</i>	0	9.3	3.4	29	12,13,44,46,48-50
<i>Oncorhynchus mykiss</i>	1	11.3	2.6	29	6,7,9,12,13,19,44,46,48-50,76
<i>Oncorhynchus tshawytscha</i>	1	16.0	2.6	29	6,7,9,12,13,19,44,46,48-50,76
<i>Osmerus mordax</i>	0	7.5	2.5	29	6,7,9,12,13,19,44-50
<i>Perca flavescens</i>	1	15.4	2.6	29	6,7,9,12,13,19,44-50
<i>Salmo trutta</i>	1	16.3	2.6	29	6,7,9,12,13,19,44,46,48-50,76
<i>Salvelinus namaycush</i>	1	12.9	2.5	29	6,7,9,12,13,19,44-50
Muggelsee, Germany					
<i>Abramis brama</i>	0	3.4	4.8	102	4
<i>Anguilla anguilla</i>	1	3.1	4.8	102	4
<i>Lucioperca lucioperca</i>	1	10.0	4.8	102	4
<i>Perca fluviatilis</i>	1	7.0	4.8	102	4
Niemenjarvi, Finland					
<i>Abramis blicca</i>	0	3.0	30.3	94	40-42
<i>Abramis brama</i>	0	3.7	30.3	94	40-42
<i>Esox lucius</i>	1	11.1	30.3	94	40-42
<i>Leuciscus rutilus</i>	0	5.4	30.3	94	40-42
<i>Lota lota</i>	1	14.1	30.3	94	40-42
<i>Perca fluviatilis</i>	1	8.3	30.3	94	40-42
North Channel, Ontario, Canada					
<i>Acipenser fluvescens</i>	0	3.7	2.1	16	5,11,19,60,61
<i>Salvelinus namaycush</i>	1	8.2	2.3	16	5,11,19,60,61
<i>Stizostedion vitreum</i>	1	9.8	2.1	16	5,11,19,60,61
Northburn Beck, UK					
<i>Salmo trutta</i>	0	2.3	2.5	96	14
Oder River, Germany					
<i>Anguilla anguilla</i>	1	8.7	10.5	179	4
<i>Aspius rapax</i>	1	10.4	10.5	179	4
<i>Esox lucius</i>	1	11.4	10.5	179	4
<i>Leuciscus rutilus</i>	0	5.2	10.5	179	4
<i>Tinca vulgaris</i>	0	6.7	10.5	179	4
Lake Ontario, Canada-USA					
<i>Alosa pseudoharengus</i>	0	1.8	1.3	43	6-9,12,19,29,51-53
<i>Ambloplites rupestris</i>	1	9.1	1.2	43	5-9,12,19,51-53,79
<i>Anguilla americanus</i>	1	5.3	1.2	43	5-9,12,19,51-53,79
<i>Aplodinotus grunniens</i>	1	5.4	1.3	43	8,12,51-53
<i>Catostomus commersoni</i>	0	1.0	0.8	43	5,8,12,19,27,29,30-38,53-57
<i>Coregonus clupeaformis</i>	0	0.7	0.8	43	5,8,12,19,29,30,33-38,53-57
<i>Cottus cognatus</i>	0	1.9	1.3	43	8,12,51-53
<i>Cyprinus carpio</i>	0	1.3	1.0	43	5,8,12,22,24,26,27,51-54
<i>Dorosoma cepedianum</i>	0	3.8	1.1	43	5,8,12,25,51-54
<i>Esox lucius</i>	1	2.6	1.2	43	5,8,12,51-54
<i>Salvelinus fontinalis</i> × <i>S. namaycush</i>	1	2.1	0.8	43	5,8,12,19,29,53,54

TABLE 1. (Continued)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (μm·L ⁻¹)	References ^a
<i>Ictalurus natalis</i>	1	1.8	1.0	43	5-9,12,19,23-27,51-54,58,79
<i>Ictalurus punctatus</i>	1	4.7	0.9	43	5,8,12,25,53
<i>Lepisosteus osseus</i>	1	4.5	1.3	43	8,12,51-53
<i>Maxostoma macrolepidotum</i>	0	4.0	1.3	43	8,12,51-53
<i>Micropterus dolomieu</i>	1	2.4	0.8	43	5,7,8,12,23,26-32,36,38,53-57
<i>Micropterus salmoides</i>	1	7.8	1.3	43	8,12,51-53
<i>Morone americana</i>	1	2.1	0.6	43	5,8,12,24,26,37,38,53,54,56,57
<i>Morone chrysops</i>	1	3.8	1.0	43	5-9,12,19,22,24,25,51-54
<i>Oncorhynchus kisutch</i>	1	4.2	1.2	43	5-9,12,19,51-53,79
<i>Oncorhynchus mykiss</i>	1	2.0	0.9	43	5,7,8,12,23-25,27,29,30,53-55
<i>Oncorhynchus tshawytscha</i>	1	5.9	1.3	43	8,12,51-53
<i>Osmerus mordax</i>	0	2.4	1.2	43	8,12,27,51-54
<i>Perca flavescens</i>	1	4.3	1.0	43	5-9,12,19,23,25-34,51-58,79
<i>Salmo trutta</i>	1	1.7	0.9	43	5,8,12,19,29,30,33-38,53-57
<i>Salvelinus namaycush</i>	1	1.5	0.9	43	5,8,12,26,28-30,32-38,53-57
<i>Stizostedion vitreum</i>	1	2.4	0.6	43	5,8,12,27,37,38,53,54,56,57
Ottawa River, Canada					
<i>Catostomus commersoni</i>	0	11.7	3.5	17	26,28-34,57,59-62
<i>Coregonus clupeaformis</i>	0	5.6	2.0	17	57,59,63
<i>Esox lucius</i>	1	26.9	3.5	17	28,31-33,57,59,61-63
<i>Ictalurus punctatus</i>	1	27.0	3.4	17	28,30,31,57,59,61
<i>Lota lota</i>	1	29.4	3.9	17	26,27,30,57,59-61
<i>Moxostoma anisurum</i>	0	9.6	3.2	17	28,57,59,60,63
<i>Micropterus dolomieu</i>	1	21.1	2.9	17	28,31,57,59,61,63
<i>Perca flavescens</i>	1	21.4	3.0	17	29,33,57,59,60,62,63
<i>Stizostedion canadense</i>	1	41.3	3.6	17	30,57,59,61
<i>Stizostedion vitreum</i>	1	54.9	3.6	17	26-34,57,59-63
Par Pond, South Carolina, USA					
<i>Ictalurus natalis</i>	1	1406.0	1221.0	36	64,65
<i>Lepomis macrochirus</i>	0	999.0	1221.0	36	64,65
<i>Micropterus salmoides</i>	1	1295.0	1221.0	36	64,65
Po River, Italy					
<i>Barbus barbus</i>	0	14.3	112.0	66	66
<i>Chondrostoma soetta</i>	0	50.3	112.0	66	66
<i>Chondrostoma toxostoma</i>	0	43.8	112.0	66	66
<i>Leuciscus caphalus</i>	0	21.3	112.0	66	66
Pond B, South Carolina, USA					
<i>Dorosoma cepedianum</i>	0	459.0	760.0	36	67
<i>Gambusia affinis</i>	0	1750.0	760.0	36	67
<i>Ictalurus natalis</i>	1	332.0	760.0	36	67
<i>Labidesthes sicculus</i>	0	45.0	760.0	36	67
<i>Lepomis gulosus</i>	0	149.0	760.0	36	67
<i>Lepomis macrochirus</i>	0	246.0	760.0	36	67
<i>Micropterus salmoides</i>	1	381.0	760.0	36	67
<i>Pomoxis nigromaculatus</i>	1	431.0	760.0	36	67
Red Lake, Minnesota, USA					
<i>Aplodinodus grunniens</i>	1	52.6	37.0	179	68
<i>Catostomus commersoni</i>	0	22.7	37.0	179	68
<i>Coregonus clupeaformis</i>	0	40.7	37.0	179	68
<i>Esox lucius</i>	1	133.9	37.0	179	68
<i>Lota lota</i>	1	44.2	37.0	179	68
<i>Perca flavescens</i>	1	27.8	37.0	179	68
<i>Perca flavescens</i>	0	15.1	37.0	179	68
River Gary, UK					
<i>Salmo trutta</i>	0	136.9	48.7	8	14
Schnable Lake, Michigan, USA					
<i>Micropterus salmoides</i>	1	3.7	3.0	90	1

TABLE 1. (Continued)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (μm·L ⁻¹)	References ^a
Stechlinsee, Germany					
<i>Anguilla anguilla</i>	1	132.5	74.0	51	4
<i>Coregonus albula</i>	0	118.4	74.0	51	4
<i>Esox lucius</i>	1	186.5	74.0	51	4
<i>Leuciscus rutilus</i>	0	76.5	74.0	51	4
<i>Perca fluviatilis</i>	1	228.3	74.0	51	4
St. Lawrence River, Quebec, Canada					
<i>Catostomus commersoni</i>	0	1.5	1.1	40	8,12,53,54,56,57,59,62,63,69
<i>Cyprinus carpio</i>	0	1.2	1.0	40	8,12,53,54,56,57,62,63,69
<i>Esox lucius</i>	1	2.3	1.2	40	8,12,53,54,56,57,62,63,69
<i>Ictalurus punctatus</i>	1	2.6	1.3	40	8,12,53,54,56,57,62,63,69
<i>Micropterus dolomieu</i>	1	2.9	1.2	40	8,12,53,54,56,57,62,63,69
<i>Moxostoma anisurum</i>	0	1.3	1.0	40	8,12,53,54,56,57,62,63,69
<i>Perca flavescens</i>	1	2.9	1.2	40	8,12,53,54,56,57,62,63,69
<i>Stizostedion vitreum</i>	1	2.6	1.2	40	8,12,53,54,56,57,62,63,69
Suolijarvi, Finland					
<i>Coregonus albula</i>	0	53.3	29.2	22	40-42
<i>Esox lucius</i>	1	143.2	29.2	22	40-42
<i>Leuciscus rutilus</i>	0	65.9	29.2	22	40-42
<i>Lota lota</i>	1	80.5	29.2	22	40-42
<i>Perca fluviatilis</i>	1	100.1	29.2	22	40-42
Lake Superior, Canada-USA					
<i>Osmerus mordax</i>	0	7.4	2.8	14	6,9-13,19
<i>Salvelinus namaycush</i>	1	12.7	2.4	14	5,10-13,70
<i>Salvelinus siscowet</i>	1	14.3	2.3	14	5,10-13,70
Sutton Bingham, UK					
<i>Salmo trutta</i>	0	0.5	1.3	103	14
Trawsfynydd, UK					
<i>Perca fluviatilis</i>	1	92.5	18.9	9	14
<i>Perca fluviatilis</i>	1	960.0	228.0	10	14,71-73
<i>Salmo trutta</i>	0	473.3	228.0	10	14,71-73
<i>Salmo trutta</i>	0	62.9	18.9	9	14
Ullswater, UK					
<i>Salmo trutta</i>	0	10.7	7.9	12	14
White Oak Lake, Tennessee, USA					
<i>Carassius auratus</i>	0	19.5*	28.4*	45	74
<i>Carassius auratus</i>	0	1277.6	2153.4	45	74
<i>Dorosoma cepedianum</i>	0	12.9*	28.2*	45	2,74
<i>Dorosoma cepedianum</i>	0	1740.1	2153.4	45	74
<i>Ictalurus melas</i>	1	7.6*	28.0*	45	2
<i>Lepomis gulosus</i>	0	12.1*	28.4*	45	74
<i>Lepomis gulosus</i>	0	1357.5	2153.4	45	74
<i>Lepomis macrochirus</i>	0	10.1*	28.2*	45	2,74
<i>Lepomis macrochirus</i>	0	1480.4	2153.4	45	74
<i>Lepomis microlophus</i>	0	9.0*	28.4*	45	74
<i>Lepomis microlophus</i>	0	993.5	2153.4	45	74
<i>Micropterus salmoides</i>	1	14.3*	28.2*	45	2,74
<i>Micropterus salmoides</i>	1	1951.8	2153.4	45	74
<i>Notemigonus crysoleucas</i>	0	14.5*	28.4*	45	74
<i>Notemigonus crysoleucas</i>	0	2316.6	2153.4	45	74
Lake Winnipeg, Manitoba, Canada					
<i>Coregonus clupeaformis</i>	0	0.9	1.2	57	15,16
<i>Stizostedion canadense</i>	1	2.9	1.2	57	15,16
Wintergreen Lake, Michigan, USA					
<i>Amia calva</i>	1	8.2	10.2	249	2,75
<i>Amia calva</i>	1	5.2*	23.0*	249	2,75
<i>Erimyzon succeta</i>	0	7.7	10.2	249	2,75
<i>Erimyzon succeta</i>	0	6.4*	23.0*	249	2,75

TABLE 1. (Continued)

Location and species	Trophic level	^{137}Cs fish (Bq·kg ⁻¹)	^{137}Cs water (mBq·L ⁻¹)	K (µm·L ⁻¹)	References ^a
<i>hybrid sunfish</i>	0	5.4*	23.0*	249	2,75
<i>hybrid sunfish</i>	0	4.9	10.2	249	2,75
<i>Ictalurus natalis</i>	1	9.1	10.2	249	2,75
<i>Ictalurus natalis</i>	1	5.3*	23.0*	249	2,75
<i>Ictalurus nebulosus</i>	1	4.7	10.2	249	2,75
<i>Ictalurus nebulosus</i>	1	6.7*	23.0*	249	2,75
<i>Micropterus salmoides</i>	1	12.0*	23.0*	249	2,75
<i>Micropterus salmoides</i>	1	16.2	10.1	249	2,75
<i>Perca flavescens</i>	1	11.4	10.2	249	2,75
<i>Perca flavescens</i>	1	33.8*	23.0*	249	2,75
<i>Marine</i>					
Atlantic Ocean, Faroe Islands					
<i>Gadus callarias</i>	1	0.3	3.5	10030	77–85
<i>Melanogrammus aeglefinus</i>	0	0.2	3.5	10030	77–85
Atlantic Ocean, Greenland					
<i>Anarrhichas lupus</i>	1	0.5	5.8	8840	84
<i>Gadus morhua</i>	1	0.5	5.0	8840	82,84,86–88
<i>Hippoglossus vulgaris</i>	1	0.3	6.5	8840	85
<i>Mallotus villosus</i>	0	0.2	7.2	8840	84,86,88,89
<i>Salmo salar</i>	1	0.3	5.7	8840	82,84,86,87
<i>Salmo trutta</i>	1	0.8	6.2	8840	83
Atlantic Ocean, Iceland					
<i>Gadus morhua</i>	1	0.5	3.6	10002	71–73,90
<i>Melanogrammus aeglefinus</i>	0	0.2	3.5	10002	72,90
<i>Pleuronectes platessa</i>	0	0.2	3.8	10002	73,90
Atlantic Ocean, Norway					
<i>Gadus morhua</i>	1	2.2	45.0	9492	72,90
<i>Melanogrammus aeglefinus</i>	0	2.6	45.0	9492	72,90
Atlantic Ocean, Spain					
<i>Micromesistius poutassou</i>	0	0.3	3.9	9917	91
<i>Trachurus trachurus</i>	1	0.4	3.9	9917	91
<i>Trisotropis luscus</i>	1	0.3	3.9	9917	91
Baltic Sea, Belt Sea, Baresback, Sweden					
<i>Anguilla anguilla</i>	1	2.9	20.4	2774	92–98
<i>Gadus morhua</i>	1	5.0	20.4	2774	92–98
<i>Limanda limanda</i>	0	2.3	20.4	2774	92–98
<i>Pleuronectes flesus</i>	0	2.2	20.4	2774	92–98
<i>Pleuronectes platessa</i>	0	2.6	20.4	2774	92–98
Baltic Sea, Belt Sea, Ringhals, Sweden					
<i>Anarrhichas lupus</i>	1	4.4	40.2	5287	92–98
<i>Anguilla anguilla</i>	1	2.4	40.2	5287	92–98
<i>Gadus morhua</i>	1	4.0	40.2	5287	92–98
<i>Limanda limanda</i>	0	2.9	40.2	5287	92–98
<i>Limanda</i> sp.	0	2.4	40.2	5287	92–98
<i>Myoxocephalus quadricornis</i>	0	4.1	40.2	5287	92–98
<i>Pleuronectes flesus</i>	0	1.3	40.2	5287	92–98
<i>Pleuronectes platessa</i>	0	2.5	40.2	5287	92–98
Baltic Sea, Bothnian Bay, Finland					
<i>Clupea harengus</i>	0	3.0	12.1	878	99–102
<i>Esox lucius</i>	1	7.8	12.1	878	99–102
Baltic Sea, Gottland Basin, Okarshamn, Sweden					
<i>Anguilla anguilla</i>	1	2.5	17.0	1983	92,101
<i>Esox lucius</i>	1	2.1	17.0	1983	92,101
Baltic Sea, Greifswalder Bodden, Germany					
<i>Anguilla anguilla</i>	1	7.7	47.3	1790	4
<i>Clupea harengus</i>	0	2.4	47.3	1790	4

TABLE 1. (Continued)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (μm·L ⁻¹)	References ^a
<i>Esox lucius</i>	1	17.6	47.3	1790	4
<i>Leuciscus rutilus</i>	0	2.9	47.3	1790	4
<i>Lucioperca lucioperca</i>	1	12.3	47.3	1790	4
<i>Perca fluviatilis</i>	1	12.9	47.3	1790	4
<i>Pleuronectes flesus</i>	0	4.0	47.3	1790	4
Baltic Sea, Gulf of Bothnia, Forsmark, Sweden					
<i>Anguilla anguilla</i>	1	3.1	14.0	1558	92,101
<i>Esox lucius</i>	1	4.9	14.0	1558	92,101
<i>Perca fluviatilis</i>	1	3.9	14.0	1558	92,101
Baltic Sea, Gulf of Bothnia, Okiluoto, Finland					
<i>Clupea harengus</i>	0	1.5	10.8	1674	99-102
<i>Esox lucius</i>	1	4.1	12.7	1674	99-102
<i>Perca fluviatilis</i>	1	7.0	14.0	1674	99-102
<i>Rutilus rutilus</i>	0	1.6	14.0	1674	99-102
Baltic Sea, Gulf of Bothnia, Seili, Finland					
<i>Clupea harengus</i>	0	1.9	13.0	1898	99-102
<i>Esox lucius</i>	1	4.9	13.0	1898	99-102
Baltic Sea, Gulf of Bothnia, Vassa, Finland					
<i>Clupea harengus</i>	0	2.7	15.6	1643	99-102
<i>Esox lucius</i>	1	7.1	12.3	1643	99-102
Baltic Sea, Gulf of Finland, Lovissa, Finland					
<i>Clupea harengus</i>	0	1.7	8.6	1224	99-102
<i>Esox lucius</i>	1	3.3	8.6	1224	99-102
<i>Leuciscus rutilus</i>	0	1.5	12.0	1224	99-102
<i>Perca fluviatilis</i>	1	6.5	12.0	1224	99-102
Baltic Sea, Gulf of Finland, Tvarmine, Finland					
<i>Clupea harengus</i>	0	2.5	14.3	1757	99-102
<i>Esox lucius</i>	1	4.7	13.2	1757	99-102
Bering Sea, Alaska, USA					
<i>Oncorhynchus keta</i>	1	1.0	18.5	8783	106
Gulf of California, Mexico					
<i>Anisotremus davidsoni</i>	0	36.0*	320.0*	10200	103
<i>Bairdiella icistius</i>	0	54.0*	320.0*	10200	103
<i>Cynoscion xanthulus</i>	1	39.0*	320.0*	10200	103
<i>Mugil cephalus</i>	0	51.0*	320.0*	10200	103
<i>Thunnus alalunga</i>	1	40.5*	320.0*	9492	103
Irish Sea, Chapelcross, UK					
<i>Pleuronectes flesus</i>	0	263.0	3993.3	8783	71-73
<i>Salmo trutta</i>	1	96.0	1815.0	8783	73
Irish Sea, west, Ireland					
<i>Clupea harengus</i>	0	86.0	1000.0	9633	104,105
<i>Gadus morhua</i>	1	75.0	1000.0	9633	104,105
<i>Merlangus merlangus</i>	1	97.0	1000.0	9633	104,105
<i>Pleuronectes platessa</i>	0	35.0	1000.0	9633	104,105
Irish Sea, Wylfa, UK					
<i>Pleuronectes platessa</i>	0	22.0	854.0	9633	71
<i>Pleuronectes flesus</i>	0	38.0	611.0	9633	72
North Sea, middle, UK					
<i>Clupea harengus</i>	0	13.0	123.9	9803	71,72,90
<i>Gadus morhua</i>	1	9.7	100.9	9803	71-73,90
<i>Melanogrammus aeglefinus</i>	0	6.7	100.9	9803	71-73,90
<i>Merlangus merlangus</i>	1	17.2	100.9	9803	71-73,90
<i>Pleuronectes platessa</i>	0	4.7	100.9	9803	71-73,90
North Sea, north, UK					
<i>Clupea harengus</i>	0	3.9	59.9	9917	71,72,90
<i>Gadus morhua</i>	1	6.5	59.8	9917	71-73,90

TABLE 1. (Concluded)

Location and species	Trophic level	¹³⁷ Cs fish (Bq·kg ⁻¹)	¹³⁷ Cs water (mBq·L ⁻¹)	K (µm·L ⁻¹)	References ^a
<i>Melanogrammus aeglefinus</i>	0	4.6	59.8	9917	71-73,90
<i>Pleuronectes platessa</i>	0	4.0	59.8	9917	71-73,90
<i>Pollachius virens</i>	1	3.3	56.9	9917	72,73,90
<i>Pout</i>	0	1.2	54.2	9917	72,90
North Sea, southern, UK					
<i>Gadus morhua</i>	1	7.1	39.4	9633	71-73,90
<i>Merlangus merlangus</i>	1	4.4	39.4	9633	71-73,90
<i>Pleuronectes platessa</i>	0	2.7	39.4	9633	71-73,90
Pacific Ocean, California, USA					
<i>Thunnus alalunga</i>	1	3.3	24.0	9492	107
Pacific Ocean, Gulf of Alaska, USA					
<i>Oncorhynchus keta</i>	1	0.9	18.5	8925	106
<i>Oncorhynchus kisutch</i>	1	1.0	25.3	9492	106
<i>Oncorhynchus kisutch</i>	1	1.2	18.5	9067	106
<i>Oncorhynchus nerka</i>	1	1.0	18.5	8925	106
Pacific Ocean, Hawaii, USA					
<i>Thunnus alalunga</i>	1	2.1	12.9	9917	107
Pacific Ocean, Japan					
<i>Argyrosomus argantatus</i>	1	0.3	12.2	9237	108
<i>Beryx splendens</i>	1	1.0	12.2	9237	108
<i>Chrysophrys major</i>	0	0.4	13.9	9237	109
<i>Ditrema temmincki</i>	1	1.1	17.8	9237	109
<i>Engraulis</i> sp.	0	0.2	12.2	9237	110
<i>Hyperoglyphe japonica</i>	0	0.4	12.2	9237	108
<i>Katsuwonus pelamis</i>	1	0.5	12.2	9237	108
<i>Konosirus punctatus</i>	0	0.5	12.5	9237	109
<i>Lateolabrax japonicus</i>	1	1.6	12.2	9237	108
<i>Lotella maximowiczi</i>	1	0.6	12.2	9237	108
<i>Myllo macrocephalus</i>	0	0.6	11.5	9237	109
<i>Paracaesio caeruleus</i>	1	0.5	12.2	9237	108
<i>Paralichthys olivaceus</i>	0	0.4	12.2	9237	108
<i>Pleuronectes platessa</i>	0	0.3	12.2	9237	110
<i>Sardinops melanosticta</i>	0	1.1	19.8	9237	109
<i>Scombrops boops</i>	1	1.2	12.2	9237	108
<i>Sebastes matsubarai</i>	0	0.4	12.2	9237	108
<i>Sebastes guntherii</i>	0	0.7	20.4	9237	109
<i>Seriola quinqueradiata</i>	1	0.5	13.3	9237	109
<i>Thunnus alalunga</i>	1	2.7	14.8	9633	107
Pacific Ocean, Samoa					
<i>Thunnus alalunga</i>	1	0.5	3.1	10058	107
Sea of Japan, Japan					
<i>Gadus macrocephalus</i>	1	1.3	16.6	9237	109
<i>Limanda irrdorum</i>	0	0.5	13.6	9237	109
<i>Mugil cephalus</i>	0	0.9	15.5	9237	109
<i>Scomber japonicus</i>	0	0.6	24.6	9237	109
<i>Sillago sihama</i>	0	1.4	16.9	9237	109
<i>Trachurus japonicus</i>	0	0.7	14.9	9237	109

^a(1) KeVERN and Spigarelli 1971; (2) Nelson 1969; (3) Hakonson and Whicker 1975; (4) Ettenhuber and Rohnsch 1975; (5) Joshi 1984; (6) Wahlgren and Nelson 1974; (7) Alberts and Wahlgren 1981; (8) Weiler and Chawla 1968; (9) Marshall et al. 1975; (10) Tracy and Prantl 1983; (11) Basch et al. 1980; (12) Kramer 1964; (13) Weiler and Chawla 1969; (14) Preston et al. 1967; (15) Elliot et al. 1981; (16) Armstrong and Schindler 1971; (17) Fee et al. 1985; (18) Rawson 1950; (19) Joshi 1991; (20) Armstrong and Lutz 1977a; (21) Smith et al. 1977; (22) Gorman and Walsh 1976; (23) Gorman and Walsh 1977; (24) Gorman and Walsh 1978; (25) Gorman and Walsh 1979; (26) Pilar and Walsh 1980; (27) Pilar 1981; (28) Neil 1984; (29) Neil 1982; (30) Neil 1983; (31) Neil 1985; (32) Neil 1986; (33) Neil 1987; (34) Neil 1988; (35) Neil 1989; (36) Neil 1990; (37) Neil 1991; (38) Christiani 1992; (39) Hasanen and Miettinen 1963; (40) Kolehmainen et al. 1966; (41) Kolehmainen et al. 1967; (42) Jaakkola et al. 1967; (43) Shure and Gottschalk 1976; (44) Nelson et al. 1972; (45) Waller et al. 1974; (46) Bowen 1972; (47) Yaguchi et al. 1974; (48) Torrey 1976; (49) Klein 1975; (50) Risley and Fuller 1965; (51) Bowen and Noshkin 1973; (52) Bowen 1974; (53) Kramer 1962; (54) Roy et al. 1990; (55) Baweja et al. 1987; (56) Neal 1991; (57) Atomic Energy of Canada, unpublished data; (58) Knight and Wong 1974; (59) Merritt 1975; (60) Lambert and Gibson 1983; (61) Nowell 1985; (62) Lim 1988; (63) Lim 1991; (64) Harvey 1964; (65) Newman and Brisbin 1990; (66) Smedile and Queirazza 1976; (67) Whicker et al. 1989; (68) Gustafson 1967; (69) Hydro Quebec; (70) Armstrong

to quantify and model, but has important implications for contaminant biomagnification. Rowan and Rasmussen (1992) characterized these differences using a trophic efficiency measure, the ratio of the sustained commercial yield of piscivorous fish (i.e., the top of the food chain) to the net primary production of the phytoplankton (base of the food chain) (PY/PP), and showed that among-lake variation in the bioaccumulation of persistent organochlorines in Great Lakes fish was largely due to variation in the length of pelagic food webs. Thus we hypothesized that the bioaccumulation of ^{137}Cs by fish will increase with increasing food-chain length (decreasing PY/PP). For many of the systems in our data set we were able to obtain data on the mean annual commercial yield of piscivorous fish ($\text{g C} \cdot \text{m}^{-2} \cdot \text{yr}^{-1}$) as well as mean annual primary production ($\text{g C} \cdot \text{m}^{-2} \cdot \text{yr}^{-1}$) (Table 2). We were also able to characterize fish according to their thermal zone (i.e., epilimnetic = 1 vs. hypolimnetic = 0), and mean annual temperature ($^{\circ}\text{C}$) of the site. We expected that the temperature regime might affect the bioaccumulation of ^{137}Cs because it has been shown to influence the clearance rate of Cs isotopes from body tissues, in addition to its overall effect on metabolism. Body size, another important determinant of ^{137}Cs clearance and metabolism was also included in this analysis.

In addition, for this data set we were able to calculate allochthonous sediment mass balances, and the suspended sediment concentrations that we estimated ranged widely, from $71 \text{ mg} \cdot \text{L}^{-1}$ in the highly turbid western basin of Great Slave Lake to as low as $0.2 \text{ mg} \cdot \text{L}^{-1}$ in basins from Canadian shield drainage, and as much as 20-fold lower in the open ocean (Table 2). This allowed us to test the hypothesis that we outlined above concerning the negative effect of suspended clay on the ^{137}Cs BAF.

The models obtained from the log-linear regressions were

$$(3) \log[^{137}\text{Cs}]_{\text{fish}} = 2.743(0.225) + [1.012(0.033) - 0.105(0.035)(\text{trophic level})] \log[^{137}\text{Cs}]_{\text{water}} - 0.546(0.040) \log[\text{K}]_{\text{water}} + 0.362(0.041)(\text{trophic level}) - 0.249(0.035)(\log \text{PY/PP}) + 0.016(0.003)(\text{mean annual temperature}) + 0.103(0.040)(\text{thermal zone}) - 0.133(0.030)(\log[\text{suspended sediment}])$$

$$R^2 = 0.897, \text{SE}_{\text{est}} = 0.192, N = 209$$

$$(4) \log ^{137}\text{Cs BAF} = - [0.099(0.031)(\text{trophic level})] \log[^{137}\text{Cs}]_{\text{water}} - 0.541(0.037) \log[\text{K}]_{\text{water}} + 0.356(0.037)(\text{trophic level}) - 0.251(0.034)(\log \text{PY/PP}) + 0.016(0.003)(\text{mean annual temperature}) + 0.104(0.040)(\text{thermal zone}) - 0.135(0.030)(\log[\text{suspended sediment}])$$

$$R^2 = 0.946, \text{SE}_{\text{est}} = 0.191, N = 209.$$

Body sizes (length or weight or their logarithms) were not significant in any of the regressions that we tested (Equations (1)–(4)).

For this data set, we were able to obtain data on other major ions, and on specific conductivity. Although we hypothesized on the basis of the chemical similarity of the hydrated ions of Cs and K, that K should be a more effective isotropic carrier than other ions, the strong correlations that exist among ions make it possible that effective predictive models could be established using a wide variety of chemical parameters. We first used stepwise elimination (SYSTAT backward elimination; critical $t = 1.5$ for variable entry and leaving) with all chemical parameters (Table 2) tested. This analysis showed that K was in fact the best chemical parameter selected, and that no significant improvement in predictive power could be obtained from the inclusion of more chemical constituents. However, owing to the strong correlation among different chemical constituents, strong predictive models could be developed using any of the parameters.

Comparison of ^{137}Cs at Reactor Sites to Some Remote Sites in Canada

While our analyses show that environmental concentrations of ^{137}Cs are an important determinant of equilibrium levels in fish, the data indicate that ^{137}Cs levels in fish at reactor sites need not be higher than in fish from remote sites influenced solely by long-range atmospheric deposition. For example, the mass-balance calculations of Joshi (1991) suggest that about 50% of the ^{137}Cs in the waters of Lakes Ontario and Huron is of reactor origin, yet these waters have among the lowest ^{137}Cs concentrations in Canada (Table 3). This suggests that among-system variability in ^{137}Cs concentrations of water and fish resulting from catchment and lake processes is much greater than the reactor-derived increment. (Table 3).

Systems with catchments on the Canadian Shield have strikingly higher ^{137}Cs concentrations in their water than systems with catchments on sedimentary rock. This can be seen by comparing the Ottawa River, MacLeod Bay, and Lake Superior (shield catchments) to Lake Ontario, Lake Winnipeg, and the west basin of Great Slave Lake (mainly sedimentary catchments). Within Great Slave Lake, a lake far removed from reactor inputs, the on-shield (MacLeod Bay) to off-shield (west basin) gradient is most striking, with Christie Bay, a mixed drainage, having intermediate concentrations of ^{137}Cs in its water (Table 3). While catchment retention of radiocesium is always very high, it would appear that shield catchments retain a lower fraction of the ^{137}Cs fallout than do sedimentary catchments.

The Ottawa River, with a very large contribution from shield drainage, contains the highest concentrations of ^{137}Cs in both water and fish of all of the Canadian sites. These levels are high throughout the river, and the values obtained

and Lutz 1977b; (71) Hunt 1982; (72) Hunt 1984; (73) Hunt 1986; (74) Kolehmainen 1972; (75) Eyman and Kevern 1975; (76) Spigarelli and Edwards 1975; (77) Aarkrog et al. 1979a; (89) Aarkrog et al. 1981b; (79) Walsh and Wong 1975; (80) Aarkrog et al. 1982b; (81) Aarkrog et al. 1983b; (82) Aarkrog et al. 1985a; (83) Aarkrog et al. 1985c; (84) Aarkrog et al. 1987b; (85) Aarkrog et al. 1992; (86) Aarkrog et al. 1981c; (87) Aarkrog et al. 1982c; (88) Aarkrog et al. 1983c; (89) Aarkrog et al. 1979b; (90) Camplin and Aarkrog 1989; (91) Mitchell et al. 1988; (92) Notter 1986; (93) Aarkrog et al. 1982a; (94) Aarkrog et al. 1984; (95) Aarkrog et al. 1987a; (96) Aarkeog et al. 1981a; (97) Aarkrog et al. 1985b; (98) Aarkrog et al. 1983a; (99) Institute of Radiation Protection 1979; (100) Institute of Radiation Protection 1980; (101) Institute of Radiation Protection 1984; (102) Finnish Centre for Radiation and Nuclear Safety 1987; (103) Young and Folsom 1979; (104) McAulay and Doyle 1985; (105) Jefferies and Steele 1989; (106) Jenkins 1969; (109) Young et al. 1975; (108) Nakahara et al. 1980; (109) Suzuki et al. 1973; (110) Kurabayashi et al. 1980.

TABLE 2. Production, suspended sediment concentration, and ion chemistry.

Site	Primary production (g C·m ⁻² ·yr ⁻¹)	Piscivore fish yield (g C·m ⁻² ·yr ⁻¹)	Total fish yield (g C·m ⁻² ·yr ⁻¹)	Piscivore fish yield/primary production (×10 ³)	Total fish yield/primary production (×10 ³)	Suspended sediment (mg·L ⁻¹)	Salinity (‰)	Ca (μM·L ⁻¹)	K (μM·L ⁻¹)	Mg (μM·L ⁻¹)	Na (μM·L ⁻¹)	References ^a
<i>Freshwater</i>												
Lake Erie												
Central Basin	224	0.040 50	0.078 25	0.180 80	0.349 33	23.56		953	33	326	472	1-4
Eastern Basin	176	0.015 40	0.019 72	0.087 50	0.112 05	14.98		988	33	349	497	1-4
Western Basin	340	-0.102 10	0.174 43	0.300 29	0.513 03	11.12		858	31	281	332	1-4
Georgian Bay	80	0.000 47	0.002 69	0.005 88	0.033 63	0.21		578	18	279	141	1,3,5-7
Great Slave Lake												
Christie Bay	112	0.003 54	0.004 40	0.031 59	0.039 25	1.87		547	23	207	262	8-11
McLeod Bay	59	0.001 14	0.001 29	0.019 41	0.021 85	0.23		74	11	42	44	8-11
Western Basin	136	0.004 46	0.012 44	0.032 82	0.091 46	70.70		610	26	225	287	8-11
Lake Huron												
Northern Basin	90	0.000 20	0.002 74	0.002 22	0.030 44	2.70		636	22	310	145	1,3,5-7
Southern Basin	90	0.003 59	0.007 23	0.039 89	0.080 33	4.12		717	22	325	195	1,3,5,12
Lake Michigan	144	0.001 00	0.019 98	0.006 94	0.138 75	1.40		847	29	446	185	1,3,6,7,13-15
North Channel	72	0.000 89	0.004 51	0.012 36	0.062 64	1.03		549	16	263	139	1,5
Lake Ontario	180	0.004 19	0.006 60	0.023 28	0.036 67	7.92		988	43	336	483	1,6,7
Lake Superior	50	0.000 27	0.005 03	0.005 40	0.100 60	1.39		332	14	118	52	1,3,5-7
Lake Winnipeg	169	0.016 72	0.023 39	0.098 96	0.138 41	4.91		499	57	354	261	11,16-19
<i>Marine</i>												
Atlantic Ocean												
Faroe Islands	200	0.240 00	0.500 00	1.200 00	2.500 00	0.04	35.4	10 133	10 030	52 110	462 158	20-29
Spain	125	0.142 00	0.469 00	1.136 00	3.752 00	0.07	35.0	10 019	9 917	51 521	456 936	20,21
Iceland	250	0.407 00	1.230 00	1.628 00	4.920 00	0.05	35.3	10 105	10 002	51 963	460 853	20,21
Greenland	125	0.386 00	0.560 00	3.088 00	4.480 00	0.28	31.2	8 931	8 840	45 927	407 326	20,21,26-33
Norway	150	0.093 00	0.273 00	0.620 00	1.820 00	0.15	33.5	9 589	9 492	49 313	437 353	20,21
Baltic Sea												
Belt Sea, Baresback	158	0.218 00	0.410 00	1.379 75	2.594 94	1.11	9.8	2 802	2 774	14 411	127 812	21,34-42
Belt Sea, Ringhals	158	0.218 00	0.410 00	1.379 75	2.594 94	1.11	18.7	5 341	5 287	27 468	243 612	21,34-42
Bothnian Bay	71	0.008 57	0.040 00	0.120 70	0.563 38	0.42	3.1	887	878	4 563	40 471	21,34,35,43-46
Gottland Basin, Okarshamn	101	0.016 33	0.170 00	0.161 68	1.683 17	0.26	7.0	2 004	1.983	10 304	91 387	21,35,36,45
Gulf of Bothnia, Forsmark	81	0.008 57	0.040 00	0.105 80	0.493 83	0.34	5.5	1 574	1 558	8 096	71 804	21,35,36,45
Gulf of Bothnia, Okiluoto	81	0.008 57	0.040 00	0.105 80	0.493 83	0.34	5.9	1 692	1 674	8 700	77 157	21,34,35,43-46
Gulf of Bothnia, Seili	81	0.008 57	0.040 00	0.105 80	0.493 83	0.34	6.7	1 918	1 898	9 863	87 471	21,34,35,43-46
Gulf of Bothnia, Vassa	81	0.008 57	0.040 00	0.105 80	0.493 83	0.34	5.8	1 660	1 643	8 538	75 721	21,34,35,43-46
Gulf of Finland, Lovissa	71	0.026 66	0.310 00	0.375 49	4.336 20	0.33	4.3	1 237	1 224	6 359	56 399	21,34,35,43-46
Gulf of Finland, Tvärmine	71	0.026 66	0.310 00	0.375 49	4.366 20	0.33	6.2	1 775	1 757	9 127	80 943	21,34,35,43-46

TABLE 2. (Concluded)

Site	Primary production (g C·m ⁻² ·yr ⁻¹)	Piscivore fish yield (g C·m ⁻² ·yr ⁻¹)	Total fish yield (g C·m ⁻² ·yr ⁻¹)	Piscivore fish yield/ primary production (×10 ³)	Total fish yield/ primary production (×10 ³)	Suspended sediment (mg·L ⁻¹)	Salinity (‰)	Ca (μM·L ⁻¹)	K (μM·L ⁻¹)	Mg (μM·L ⁻¹)	Na (μM·L ⁻¹)	References ^a
Bering Sea	200	0.111 00	0.339 00	0.555 00	1.695 00	0.29	31.0	8 874	8 783	45 633	404 715	20,21
Gulf of California	100	0.211 00	0.889 00	2.110 00	8.890 00	0.01	36.0	10 305	10 200	52 993	469 992	20,21
Irish Sea												
Chaple cross	200	0.113 00	0.260 00	0.565 00	1.300 00	0.29	34.0	8 874	8 783	45 633	404 715	20,21
Western-Wylfa	200	0.113 00	0.260 00	0.565 00	1.300 00	0.12	31.0	9 732	9 633	50 049	443 881	20,21
North Sea												
Middle	150	0.168 00	0.521 00	1.120 00	3.473 33	0.09	34.6	9 904	9 803	50 932	451 714	20,21
Northern	150	0.168 00	0.521 00	1.120 00	3.473 33	0.07	35.0	10 019	9 917	51 521	456 936	20,21
Southern	150	0.168 00	0.521 00	1.120 00	3.473 33	0.12	34.0	9 732	9 633	50 049	443 881	20,21
Pacific Ocean												
California	100	0.211 00	0.889 00	2.110 00	8.890 00	0.15	33.5	9 589	9 492	49 313	437 353	20,21
Gulf of Alaska	200	0.090 00	0.584 00	0.450 00	2.740 00	0.26	31.5	9 017	8 925	46 369	411 243	20,21
Hawaii	75	0.052 00	0.261 00	0.693 33	3.480 00	0.07	35.0	10 019	9 917	51 521	456 936	20,21
Japan	114	0.750 00	1.878 00	6.578 95	16.473 68	0.20	32.6	9.332	9.237	47.988	425.603	20,21
Samoa	75	0.052 10	0.261 00	0.694 67	3.480 00	0.04	35.5	10 162	10 058	52 257	463 464	20,21
Sea of Japan	106	0.514 00	0.753 00	4.849 06	7.103 77	0.20	32.6	9 332	9 237	47 988	425 603	20,21

^a(1) Rowan and Rasmussen 1992; (2) Kramer 1961; (3) Schelske and Roth 1973; (4) Weiler and Chawla 1968; (5) Basch et al. 1980; (6) Kramer 1964; (7) Weiler and Chawla 1969; (8) Fee et al. 1985; (9) Keleher 1972; (10) Rawson 1950; (11) Armstrong and Schindler 1971; (12) Smith et al. 1977; (13) Klein 1975; (14) Risley and Fuller 1965; (15) Torrey 1976; (16) Brunskill et al. 1979; (17) Brunskill and Graham 1979; (18) Brunskill et al. 1980; (19) Manitoba Department of Natural Resources, unpublished data; (20) Gulland 1971; (21) Broecker and Peng 1982; (22) Aarkrog et al. 1979a; (23) Aarkrog et al. 1981a; (24) Aarkrog et al. 1982a; (25) Aarkrog et al. 1983a; (26) Aarkrog et al. 1985a; (27) Aarkrog et al. 1985b; (28) Aarkrog et al. 1987a; (29) Aarkrog et al. 1992; (30) Aarkrog et al. 1979b; (31) Aarkrog et al. 1981b; (32) Aarkrog et al. 1982b; (33) Aarkrog et al. 1983b; (34) Thurow 1978; (35) Zenkevitch 1963; (36) Nottter 1986; (37) Aarkrog et al. 1981c; (38) Aarkrog et al. 1982c; (39) Aarkrog et al. 1983c; (40) Aarkrog et al. 1984; (41) Aarkrog et al. 1985c; (42) Aarkrog et al. 1987b; (43) Institute of Radiation Protection 1979; (44) Institute of Radiation Protection 1980; (45) Institute of Radiation Protection 1984; (46) Finnish Centre for Radiation and Nuclear Safety 1987.

TABLE 3. Levels of ¹³⁷Cs in water (mBq·L⁻¹) and fish (Bq·kg⁻¹) from several Canadian sites with (*) and without nuclear facilities (1978–80). See Table 1 for references.

	Great Slave Lake				Lake Winnipeg*	Ottawa River below NDP*	Above NDP*	Lake Superior	Great Slave Lake		
	Lake Ontario*	Lake Huron*	Lake Superior	Ottawa River below NDP*					Christie Bay	MacLeod Bay	Western Basin
Water	0.7	1.1	1.7	4.4	4.1ns	4.4	4.1ns	1.7	1.9	3.6	1.3
Planktivores											
<i>Alosa pseudoharengus</i>	1.1	2.4							3.1	19.3	0.4
<i>Coregonus artedii</i>		2.0	4.5								
<i>Coregonus hoyi</i>	1.5	4.3	4.5						3.1	19.3	0.4
<i>Osmerus mordax</i>	1.3	2.9	4.5						1632	5361	308
Average BAF	1857	2636	2647								
Benthivores											
<i>Catostomus catostomus</i>	0.9	2.4		15.0	13.7ns				1.3		
<i>Catostomus commersoni</i>	0.6	2.4		12.3					1.8		0.5
<i>Coregonus clupeaformis</i>	1.0										
<i>Cottus cognatus</i>	1.0	1.0		13.3							
<i>Cyprinus carpio</i>	2.2								2.0	6.7	
<i>Maxostoma anisurum</i>		1.9		13.5	13.7				1.7	6.7	0.5
<i>Prosopium cylindraceum</i>	1.1	1758		3076	3341				895	1861	385
Average BAF	1629										
Piscivores											
<i>Ambloplites rupestris</i>	5.3	6.5									
<i>Anguilla americana</i>	3.3										
<i>Aplodinotus grunniens</i>	2.9			34.9	30.5ns				4.4	22.9	1.8
<i>Esox lucius</i>	1.6	4.1									
Hybrid Salvelinus	2.0										
<i>Ictalurus natalis</i>	5.0										
<i>Ictalurus nebulosus</i>		1.3									
<i>Ictalurus punctatus</i>	3.9	3.7									
<i>Lota lota</i>		4.2		42.7	28.5ns				2.9		1.1
<i>Micropterus dolomieu</i>	2.1	5.2		28.9	32.7ns						
<i>Micropterus salmoides</i>	4.3										
<i>Morone americana</i>	2.7										
<i>Morone chrysops</i>	2.8	3.4									
<i>Oncorhynchus kisutch</i>	2.6	7.7									
<i>Oncorhynchus mykiss</i>	1.6	3.2									
<i>Oncorhynchus tshawytscha</i>	3.2			30.1	33.9ns						
<i>Perca flavescens</i>	3.1	7.7									
<i>Salmo trutta</i>	1.4										
<i>Salvelinus namaycush</i>	1.3		9.9						5.7	23.8	1.1
<i>Salvelinus siscowet</i>			12.5								
<i>Stenodus leucichthys</i>											
<i>Stizostedion canadense</i>	3.0	5.9		74.1	56.0ns				2.9		
<i>Stizostedion vitreum</i>	2.9	4.8	11.2	42.1	36.3ns				4.3	23.4	1.3
Average BAF	4135	4372	6588	9577	8859				2281	6491	1026

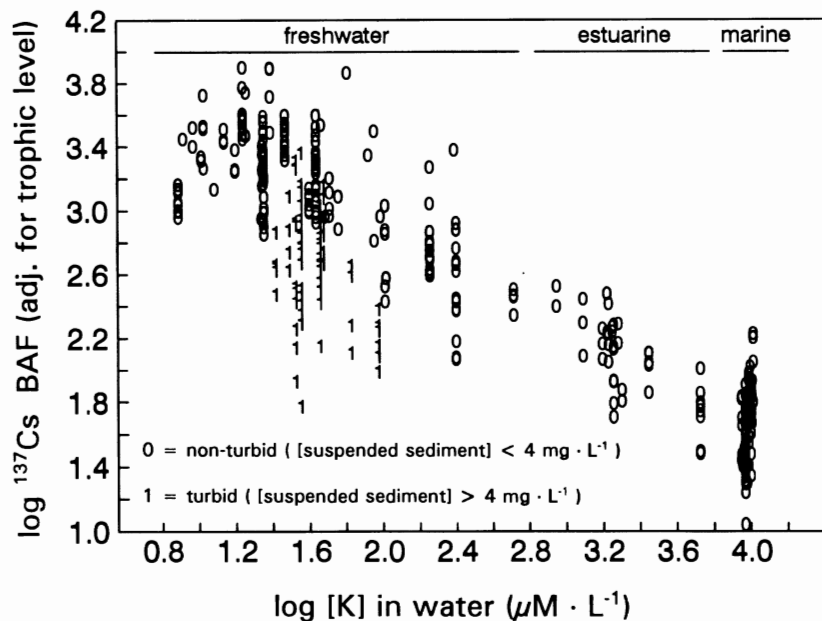


FIG. 1. Bioaccumulation of ^{137}Cs by fish in relation to concentrations of potassium and suspended sediment. BAF values are adjusted to nonpiscivore levels by subtracting 0.3 from piscivore estimates (see Equation (5)).

for fish in Holden Reservoir (upstream of the Rolphton reactor) are not significantly different from those below the dam near the reactor (Table 3). The Rolphton reactor was a small demonstration reactor operated by Ontario Hydro from 1965–83, and monitoring throughout this time showed no upstream–downstream differences. The dam at Rolphton clearly separates upstream from downstream fish populations, showing that high levels in upstream fish are not the result of upstream movements from reactor-influenced sites.

While piscivores in general bioaccumulate ^{137}Cs two- to three-fold relative to levels in forage species, the BAFs vary considerably from system to system. The soft waters of Lake Superior, Ottawa River, and MacLeod Bay have higher BAFs in all trophic categories than do Lake Ontario, Lake Huron, Christie Bay, and western Great Slave Lake (Table 3). This largely reflects the combined importance of the K and suspended sediment concentrations of the water, which were important predictors in Equation (4). BAFs for planktivores and benthivores are the lowest in the western basin of Great Slave Lake where the influence of suspended clay sediment is most pronounced.

Vulnerability of Softwater Systems to Radiocesium Contamination: Biogeochemical Factors

This analysis of global radiocesium data obtained from monitoring programs spanning the last 30 yr strongly supports the hypothesized effects of ions (especially K^+) as isotopic carriers by demonstrating that radiocesium bioaccumulation by fish, which occurs mainly through food, is sharply reduced by environmental K concentrations (Fig. 1). In fresh water, [K] ranges from $<10 \mu\text{M}$ to around 0.5 mM , nearly three orders of magnitude, and in marine habitats ranges from $<1 \text{ mM}$ in estuaries to over 10 mM in the open ocean, about one order of magnitude. For each 10-fold difference in [K], bioaccumulation increases by around 3.5-fold (antilog of the regression coefficient for [K] in Equation (4)), and

variability in [K] accounts for almost 10-fold variability in ^{137}Cs BAFs in freshwater systems. Therefore, fish from the softwater drainages found over large portions of northern Europe and Canada are much more sensitive to radiocesium fallout than those from hardwater, sedimentary drainages which in turn are more sensitive than marine systems.

However, the K-carrier mechanism is not the only factor contributing to high ^{137}Cs levels in fish from soft-water drainages. The low suspended sediment loads in soft waters (as low as $0.1 \text{ mg} \cdot \text{L}^{-1}$ compared with $10\text{--}100 \text{ mg} \cdot \text{L}^{-1}$ in sedimentary drainages) also contributes to about two-fold higher bioaccumulation (Equation 4). Finally, softwater lakes concentrate ^{137}Cs from atmospheric fallout more than lakes with sedimentary watersheds (Table 3). We hypothesize that this is a result of lower watershed retention of radiocesium owing to lower clay content in soils and surficial materials. Thus our analysis points to three important biogeochemical factors that make fish from soft waters much more vulnerable to radiocesium fallout than fish from hardwater sedimentary drainages and marine systems.

Vulnerability of Softwater Systems to Radiocesium Contamination: Trophic Factors

The generality of the biomagnification effect in food chains is further supported by the negative regression coefficient obtained in Equations (3) and (4) for PY/PP. This indicated that the BAF for fish from lakes with short food chains (high piscivore yields relative to primary production), have lower radiocesium BAFs than those from longer food chains (Fig. 2). Piscivore yields range from almost 1% of net primary production to less than 0.001% (Fig. 2; Table 2). Marine systems range from around 1 to 0.01% and thus appear to have, in general, shorter more direct food chains than freshwater systems wherein piscivore yields range from 0.03% (western Lake Erie) to $<0.001\%$ in the oligotrophic Laurentian Great Lakes.

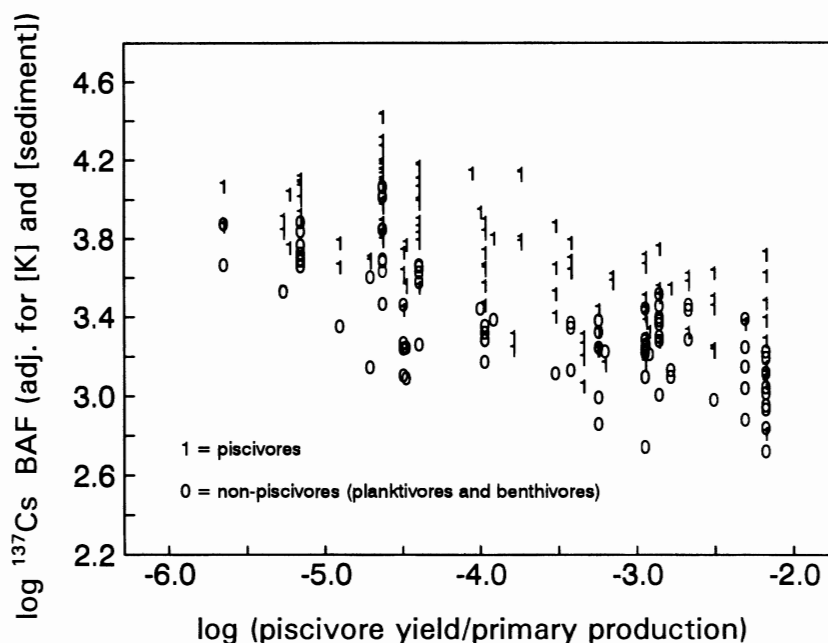


FIG. 2. Bioaccumulation of ^{137}Cs by fish in relation to trophic structure; piscivore yield/net primary production (PY/PP) characterizes the aquatic community in terms of the efficiency with which primary production is converted to fish, a function of food chain length. BAF estimates are adjusted for the effects of the two major physicochemical factors, [K], and [suspended sediment] using the coefficients in Equation (4); estimates were adjusted to $10 \mu\text{M}\cdot\text{L}^{-1}$ K, and $1 \text{ mg}\cdot\text{L}^{-1}$ suspended sediment.

Variability in PY/PP probably reflects variable size structure of phytoplankton and its consequences for the size structure of zooplankton. Picoplankton are frequently processed by a web of microconsumers (the microbial loop), whereas large phytoplankton, such as diatoms, usually support large herbivorous zooplankton, such as *Diatomus* or *Calanus* (Parsons et al. 1984). Plankton communities also vary considerably in the number of trophic levels in the zooplankton, including variation in the importance of macrozooplankton such as euphausiids, chaetognaths, fish and invertebrate larvae, *Chaoborus*, or mysids (Rasmussen et al. 1990). Thus, the efficiency with which phytoplankton is converted to fish food can vary by orders of magnitude, with major consequences for fish productivity (Ryther 1969; Gulland 1970, 1971).

The almost 3000-fold range in PY/PP accounted for about seven-fold variability in the BAF of ^{137}Cs . Such an effect is consistent with the observation that piscivores generally have higher BAFs than forage fish, but in addition, shows that there is great variability in the trophic structure at the base of food webs that link forage fish to primary production, and that this variability has important consequences for radiocesium bioaccumulation. We found a very similar effect on the BAFs for PCBs and DDT in Great Lakes fishes (Rowan and Rasmussen 1992). While PY/PP seems to be a food web descriptor that explains among-system variation, its utility for prediction will be limited to situations where data on the yields from sustained commercial fisheries are available. For most inland waters such data are not available. We, therefore, present a model predicting ^{137}Cs BAFs with no PY/PP term below (Equation (5)).

Softwater lakes tend to have the lowest PY/PP (e.g., Lake Superior, northern Lake Huron, and Georgian Bay; Table 2), and thus long food chains appear to be an additional

contributing factor to high BAFs in fish from soft waters. In the sea, where PY/PP is highest, the shorter more direct food chains, combined with K concentrations of about 10 mM lead to the lowest BAFs recorded. Planktivores from the Japanese coast, the California coast, Greenland, and Iceland have BAFs less than 50, compared with upper Great Lakes values over 2000. Thus, high [K], short food chains, and the much greater volume and mixing potential make marine fish much less vulnerable to releases of ^{137}Cs .

In spite of the resilience of marine systems to radiocesium pollution, some marine sites such as those impacted by the fuel reprocessing plant at Sellafield, UK (e.g., Irish Sea and the adjacent North Sea) receive such high loadings of ^{137}Cs that ^{137}Cs concentrations in water exceed $1 \text{ Bq}\cdot\text{L}^{-1}$, and their fish exhibit levels of contamination on the same level as those from the Ottawa River and Georgian Bay where water concentrations are only a few millibecquerels per litre.

Effect of Clay on the Assimilation of Radiocesium from Food

Although many published studies on the distribution of radiocesium in aquatic communities have failed to demonstrate increases in tissue concentrations at higher trophic levels (Nelson 1969; Reichle et al. 1970; Thomann 1981), and indeed some studies even show lower radiocesium concentrations in fish tissues than in food (even for quasi-steady-state levels of radiocesium pollution) (Kolehmanien 1972; Mailhot et al. 1988), our analysis indicates that piscivores usually have about double the radiocesium concentrations seen in planktivores and benthivores. Although this appears to be true in general, there are a few examples within the data set that do not exhibit this pattern (Table 1). Kolehmanien (1972) presents an example where high clay content in White Oak Lake leads to this type of anomaly. He showed that

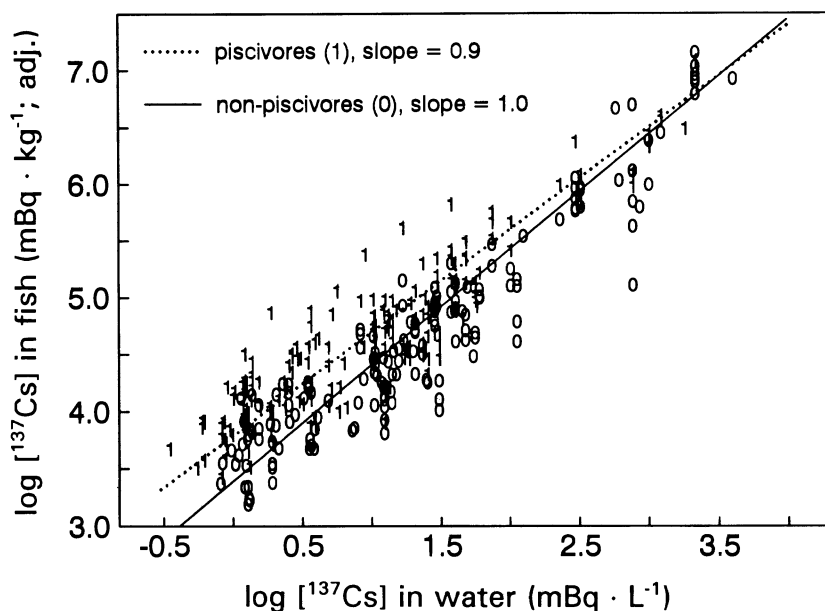


FIG. 3. Relationships between [^{137}Cs] in fish and water. Log-linear slope of 1 for nonpiscivores indicates that concentrations in fish at lower trophic levels track water concentrations closely; for piscivores slope < 1 ($p < 0.01$) suggests that cesium levels in higher trophic level fish lag temporal variations in water concentrations slightly. [^{137}Cs] in fish were adjusted for the effects of the two major physicochemical factors, [K], and [suspended sediment] using the coefficients in Equation (4); estimates were adjusted to $10 \mu\text{M} \cdot \text{L}^{-1}$ K, and $1 \text{ mg} \cdot \text{L}^{-1}$ suspended sediment.

bluegills (*Lepomis macrochirus*) have very low ($< 10\%$) assimilation efficiency for food items rich in clay (e.g., chironomid larvae and detritus) whereas food lacking clay had their radiocesium assimilated by the fish with around 60–70% efficiency. Our analysis of sites where we were able to estimate concentrations of suspended inorganic sediment by mass balance (Equation (3) and (4)), indicated that radiocesium bioaccumulation was much lower in lakes with high suspended sediment concentrations (Fig. 1). This effect was most clearly manifested in the comparison of the sediment-laden western basin of Great Slave Lake to the clearer, but chemically similar, Christie Bay waters of the same lake (Tables 2 and 3).

An alternative explanation for the effect of clay in our regressions is that some of the radiocesium concentrations in water were measured on unfiltered samples, with varying proportions of the radiocesium bound to particulates and unavailable. However, if this was the main factor then the slope of the water concentration term in Equation (1) should be significantly lower than one, and it is not. In addition, most of the water from turbid sites, where this effect would have greater significance, was filtered before analysis for radiocesium (e.g., Kolehmanien 1972; Elliot et al. 1981).

Disequilibrium between Piscivores and Environmental Concentrations

While we have tried to exclude data from pulse releases in this study, because of the biasing effect that disequilibrium between water and biota would have on estimates of BAFs, we predicted that year-to-year variations in releases from nuclear facilities and atmospheric fallout would lead to slight disequilibrium at the top of the food web (piscivores), but that planktivores and benthivores would tend to track environmental variations very closely. The log-linear

regression coefficient relating ^{137}Cs concentrations in nonpiscivorous fish to those in water was not significantly different from 1 (Equation (3); Fig. 3), which shows that BAFs estimated for nonpiscivores at high water concentrations were not significantly different from those estimated under low water concentrations. As hypothesized, fish low in the food chain appear to track variations in environmental ^{137}Cs concentrations closely enough to be essentially in equilibrium. For piscivores (Fig. 3) we observed log-linear slopes slightly but significantly < 1 , which we interpret as evidence of slight disequilibrium between tissue and water concentrations, resulting from the time lag required for ^{137}Cs released into the environment to be transmitted through the food web.

Results from tracer experiments and post-Chernobyl Europe provide more dramatic demonstrations of the disequilibrium associated with pulse inputs to aquatic systems. The fate of ^{134}Cs added to Lake 224 in the Experimental Lakes Area in northwestern Ontario was followed in the water and sediment (Hesslein et al. 1980; Hesslein 1987) and in the fish (Harrison et al. 1990). BAFs for fish in this system (BAFs have little meaning in disequibrated systems) ranged from 3170 for littoral planktivores and 797 for pelagic piscivores 63 d following the addition, to 7040 in the pelagic piscivores after 1 yr (Harrison et al. 1990). Lake 224 is very low in K (about $9.5 \mu\text{M} \cdot \text{L}^{-1}$) and suspended sediment ($< 1 \text{ mg} \cdot \text{L}^{-1}$) (Hesslein 1987) and predictions for planktivores and piscivores from Equation (5) are 4266 and 8356, respectively. As expected, the planktivores approached equilibrium quickly, while there was a substantial lag for the piscivores to reach equilibrium. Similar trends were observed in Europe following Chernobyl, with rapid increases (and decreases) in planktivores and slower, more moderate increases in piscivores (Elliott et al. 1992; Sundblad et al. 1991).

A Model for Equilibrated Systems

For truly equilibrated situations (at present, most North American systems) BAFs calculated for all fish should be independent of [^{137}Cs] in water, and a BAF model calculated without the interaction term between trophic level and [^{137}Cs] in water should apply. A simple model calculated without the interaction term and PY/PP, and including only the most robust predictors is

$$(5) \log^{137}\text{CS BAF} = \\ 4.332(0.062) - 0.718(0.022)\log[\text{K}]_{\text{water}} \\ + 0.292(0.034)(\text{trophic level}) \\ - 0.233(0.027)(\log[\text{suspended sediment}]) \\ R^2 = 0.873, SE_{\text{est}} = 0.275, N = 266.$$

Temperature Effects

The temperature regime under which a fish lives exerts profound effects on all metabolic processes. Clearance rates of radiocesium are known to increase with temperature (Ugedal et al. 1992), but it cannot necessarily be inferred that equilibrium tissue levels will decrease with increasing temperature, because increases in feeding rates and/or basal metabolism could easily counter the effects of enhanced clearance. Thus it would be extremely difficult to predict a priori whether the BAFs for radiocesium would decrease, increase, or be unaffected by temperature. Although detailed data on the temperature regime were unavailable for most sites, our estimates of the mean annual air temperature combined with basic habitat information characterizing the fish as epilimnetic or hypolimnetic provides a reasonable description of the thermal regime experienced by the fish in our data set. Of course, within a region, our estimates of the effects of temperature cannot distinguish between large lakes that have lower mean annual temperatures than small lakes (Fee and Hecky 1992). The regressions (Equation (3) and (4)) suggest that the overall effect is for equilibrium levels of ^{137}Cs to increase with temperature because significant positive regression coefficients were obtained for mean annual temperature, and for the categorical variable describing the thermal zone inhabited by the fish (epilimnetic = 1, hypolimnetic = 0). This then suggests that the tendency for feeding and metabolic rates to increase with temperature overrides the tendency for enhanced clearance. This hypothesis should be further tested in laboratory studies.

Residual Variation and its Relation to Fish Bioenergetics

There is about twofold uncertainty in the predictions of fish radiocesium bioaccumulation in our models, despite their high predictive power (SE_{est} for predicting $\log[^{137}\text{Cs}]$ in fish and $\log(\text{BAF})$ are 0.19). This uncertainty is about equal to the within-system variation observed among fish of the same trophic level (see Tables 1 and 3), variation observed among age-classes of species (D.J. Rowan, unpublished data) and seasonal variation within an age-class (Kolehmanien 1974). The natural variability is also much greater than any bias that may exist in the data owing to the mixture of analyses of whole fish or fish muscle. To explain such residual variation and thereby further increase predictive power, a much more detailed knowledge of the diet and the bioenergetic profile would be required for each age-class of each species at each site. Dynamic models such as that of Thomann

(1981) indicate that the equilibrium tissue concentration of radiocesium depends on the concentration in food, the assimilation efficiency of Cs from the food, the biomass assimilation efficiency, the clearance rate of Cs from the body, the specific rates of food consumption, metabolism, and growth, and less importantly on direct uptake from water. Of these, the metabolic parameters are virtually never available for natural populations of fish, and reasonable variation in these parameters can easily lead to two-fold variation in the predictions made by such dynamic models. For this reason, such dynamic models are generally used for illustrative purposes or for simulation and sensitivity analyses, rather than prediction.

While it is virtually impossible to provide the site-specific ecological parameters necessary to make more accurate predictions of radiocesium bioaccumulation, it should be noted that there is considerable potential for using data on radiocesium levels in fish and their food organisms in combination with clearance models to estimate metabolic parameters, which are required for a wide range of ecological and management purposes. Site-specific estimates of such parameters made by traditional methods are very labor intensive and crude. KeVERN (1966), Kolehmainen (1974), and, more recently, Forseth et al. (1992) have used this approach to estimate daily rations of fish. Furthermore, if growth rates can be supplied for the fish, which is usually possible, both daily rations and daily metabolic costs can be estimated. While this approach has only been employed in lakes with elevated radiocesium levels (KeVERN (1966) and Kolehmainen (1974) studied carp (*Cyprinus carpio*) and bluegills at White Oak Lake near Oak Ridge, TN, and Forseth et al. (1991) studied salmonids in Norwegian lakes experiencing fallout from Chernobyl), modern γ counting technology can provide accurate and reasonably rapid estimates of radiocesium levels even in systems influenced only by long-range atmospheric fallout.

Linking Isotope Studies to Contaminant Bioaccumulation

Much of our recent research effort has been directed toward studying the factors that influence the BAFs for persistent contaminants (PCBs, DDT, methyl mercury, and, most recently, ^{137}Cs) with special emphasis on the effect of food-chain structure (Rasmussen et al. 1990; Rowan and Rasmussen 1992; Cabana et al. 1994). Because detailed characterization and quantitative descriptions of food chains are labour intensive and thus rarely carried out, one of our long-term goals is to use the BAFs for persistent substances as surrogate measures to quantify food-chain length. Radioisotopes such as ^{137}Cs seem to have potential for this type of purpose, when appropriate adjustments for K concentration, suspended sediment, and temperature are made. In addition to radioisotopes, we are also studying stable isotopes (^{15}N and ^{13}C) in the food webs of lakes including the Great Lakes, to investigate their possible use in quantitative description of trophic structure (Cabana and Rasmussen 1994).

We feel that there exists considerable unexplored potential for using steady-state radioisotope tracer approaches in ecological studies, and we are hoping that by analyzing basic patterns in the ecological behaviour of radioisotopes we can contribute both models useful in environmental management, and to broadening interest among ecologists and fisheries managers in radioisotopes as potential research tools.

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