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Bioaccumulation Factors for Radionuclides in Freshwater Biota

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BIOACCUMULATION FACTORS FOR RADIONUCLIDES IN FRESHWATER BIOTA

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This report analyzes over 200 carefully selected papers to provide concise data sets and methodology for estimation of bioaccumulation factors for tritium and isotopes of strontium, cesium, iodine, manganese, and cobalt in major biotic components of freshwater environments. Bioaccumulation factors of different tissues are distinguished where significant differences occur. Since conditions in the laboratory are often unnatural in terms of chemical and ecological relationships, this review was restricted as far as possible to bioaccumulation factors determined for natural systems. Because bioaccumulation factors were not available for some shorter-lived radionuclides, a methodology for converting bioaccumulation factors of stable isotopes to those of shorter-lived radionuclides was derived and utilized.

The bioaccumulation factor for a radionuclide in a given organism or tissue may exhibit wide variations among bodies of water that are related to differences in ambient concentrations of stable-element and carrier-element analogues. To account for these variations, simple models are presented that relate bioaccumulation factors to stable-element and carrier-element concentrations in water. The effects of physicochemical form and other factors in causing deviations from these models are discussed. Bioaccumulation factor data are examined in the context of these models, and bioaccumulation factor relations for the selected radionuclides are presented.

SUMMARY

BIOACCUMULATION FACTOR CONCEPTS

The bioaccumulation factor for an organism or tissue i is the steady-state ratio of radionuclide concentration in the organism or tissue to that in water:

$$BF(R)_i = [R]_i/[R]_w$$

where

$BF(R)_i$ = bioaccumulation factor for radionuclide R in organism or tissue i,

$[R]_i$ = radionuclide concentration (μCi/g fresh weight) in organism or tissue i, and

$[R]_w$ = radionuclide concentration in water (μCi/g), a constant.

Bioaccumulation factors are used to predict radionuclide concentrations in whole organisms or their tissues from knowledge of the radionuclide concentration in water for chronic releases of radionuclides. Bioaccumulation factors for radionuclides were related to ambient concentrations of their stable or non-isotopic carrier element analogues according to three idealized patterns. The first pattern is that the bioaccumulation factor for radionuclide R in organism or tissue i, $BF(R)_i$, is constant regardless of stable-element or carrier-element concentration:

$$BF(R)_i = \text{const.} \tag{1}$$

The second pattern applies to an element that is homeostatically

maintained at a constant concentration in organism or tissue i:

$$BF(R)_i = \Sigma_i/[C]_w \ , \qquad (2)$$

where

Σ_i = concentration of stable element in organism or tissue i, a constant (μg/g fresh weight), and

$[C]_w$ = concentration of corresponding stable element in water (μg/g).

The third pattern applies to a radioisotope and its non-isotopic carrier element which is homeostatically maintained at a constant concentration in i:

$$BF(R)_i = \frac{q_i \Sigma_i^*}{[C^*]_w} \ , \qquad (3)$$

where

q_i = discrimination coefficient,

Σ_i^* = concentration of non-isotopic carrier element in organism or tissue i, a constant (μg/g fresh weight), and

$[C^*]_w$ = concentration of non-isotopic carrier element in water (μg/g).

The discrimination coefficient, q_i, is the ratio $([R]_i/[C^*]_i)/([R]_w/[C^*]_w)$, where $[C^*]_i$ is the carrier element concentration in i. Equation (3) is often rewritten in the form:

$$\ln BF(R)_i = \ln q_i\Sigma_i^* - \ln[C^*]_w \qquad (4)$$

Radionuclides exist in a wide variety of physicochemical forms in natural waters, and their different forms have different availabilities to the food chain. Sediments, too, may be source of radionuclides to biota, and sediment type can influence the availability. For those elements that are homeostatically maintained at constant concentrations in a given organism, the concentration of stable element in the organism is independent of concentration of stable element in water or its availability in prey, sediment, and different physicochemical forms in the water. In contrast, differences in availability of radionuclides in different sediment types and different physicochemical forms in water can lead to marked deviations from the idealized patterns of Equations (1) and (3).

When bioaccumulation factors were not available for radionuclides, they were estimated from bioaccumulation factors of the corresponding stable elements. Owing to physical decay, the bioaccumulation factor of a short-lived radionuclide is much less than that of the stable element or longer-lived nuclides. Bioaccumulation factors of shorter-lived radionuclides were estimated from bioaccumulation factors of the stable element according to the relation:

$$BF(R)_i = \frac{k}{k+\lambda} \quad BF(C)_i \ , \tag{5}$$

where

k = elimination coefficient of C in i (day^{-1}),

λ = radioactive decay constant (day^{-1}), and

$BF(C)_i$ = bioaccumulation factor of corresponding stable element in i.

BIOACCUMULATION FACTORS

Cesium

Potassium is a non-isotopic carrier element for cesium because of their chemical similarities and the greater abundance of potassium in water. Further, since potassium concentration is homeostatically maintained at constant concentrations in animals, the bioaccumulation factor of cesium is given by Equation (3) or (4). Unlike this pattern for animals, potassium concentration of water has only a small effect on the cesium bioaccumulation factor in algae.

The primary mode of accumulation of cesium and potassium in aquatic animals is from the food chain. Absorption efficiency of potassium and cesium from food is high. In animals the excretion coefficient of potassium is about 3 times larger than the excretion coefficient of cesium. As a result, q_i increases by a factor of about 3 with each trophic level. If potassium concentrations in the predators and prey are about equal, the cesium bioaccumulation factor increases by a factor of 3 with each trophic level.

Because cesium is strongly adsorbed by suspended particulate materials, especially clays, the proportion of cesium in the soluble phase decreases with increasing suspended solids concentrations. Potassium, too, is sorbed but to a much lesser degree. Since algae accumulate cesium, potassium, and other elements from the soluble phase, the availabilities to the food chain of cesium and of cesium relative to potassium decrease with increasing suspended solids concentrations. Thus, discrimination coefficients and bioaccumulation factors decrease with increasing suspended solids concentrations.

On the basis of data available, we recommend the bioaccumulation factors given in Table 1.

Strontium

Calcium is a non-isotopic carrier element for strontium because of their chemical similarities and the greater abundance of calcium in water. Further, since calcium concentration is homeostatically maintained at constant concentrations in animals, the bioaccumulation factor of strontium is given by Equation (3) or (4). Unlike this pattern for animals, calcium concentration of water has only a small effect on the strontium bioaccumulation factor in algae. The primary mode of uptake of strontium and calcium in animals (as well as plants) is from water. As a result, trophic level has little effect on the discrimination coefficient and the bioaccumulation factor of strontium. Further, the discrimination coefficient is relatively independent of calcium concentration in water. Calcium and strontium concentrations in bones and shells are higher than in other tissues of animals.

Strontium has physicochemical properties similar to calcium and, like calcium, appears mainly as free ions in water. As a result of this and the fact that the discrimination coefficient is independent of calcium concentration in water, the discrimination coefficient varies little among sites. This implies that the product $q_i \Sigma_i^*$

Table 1. Recommended bioaccumulation factor relations for long-lived isotopes of cesium[a]

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factor Relation
Piscivorous fishes	All tissues	Clear waters[b]	$1.5 \times 10^4/[K]_w$ [c]
Piscivorous fishes	All tissues	Turbid waters[d]	$3 \times 10^3/[K]_w$
Non-piscivorous fishes	All tissues	Clear waters	$5 \times 10^3/[K]_w$
Non-piscivorous fishes	All tissues	Turbid waters	$1 \times 10^3/[K]_w$
Algae	Whole	All waters	10^3
Aquatic vascular plants	Whole	All waters	10^3
Emergent vascular plants	Whole	All waters	10^3
Molluscs	Shell	All waters	10^2
	Soft tissues	All waters	10^3
Invertebrates other than molluscs	Whole	All waters	10^3
Amphibians	All tissues	All waters	10^4
Waterfowl	All tissues	All waters	3×10^3

[a]To convert to bioaccumulation factors of ^{136}Cs use conversion factors in Table 3.1.1.

[b]Suspended solids concentration less than 50 ppm.

[c]$[K]_w$ = stable potassium concentration of water in ppm.

[d]Suspended solids concentration greater than 50 ppm.

appearing in the numerator of Equation (3) may be treated as a constant and that a regression of ln $BF(Sr)_i$ versus $\ln[Ca]_w$, where $[Ca]_w$ is calcium concentration in water, will have a slope near -1 and a high correlation coefficient.

On the basis of the data available, we recommend the strontium bioaccumulation factors given in Table 2 and shown in Figure 1.

Tritium

Tritium was included in this report because of a concern that the relative kinetics of tritium and protium resulting from tritium's heavier mass ("isotope effect") might lead to a preferential accumulation of tritium over protium. Experiments in aquatic systems indicate that this does not occur and that the bioaccumulation factor for tritium is less than or about equal to the bioaccumulation factor for protium, which has a bioaccumulation factor approximately equal to 1. We recommend that a bioaccumulation factor of 1 be used for all tissues of all aquatic biota (Table 3).

Iodine

Iodine is highly concentrated by the thyroid tissue of vertebrates. As a result, the bioaccumulation factor for iodine in the thyroid tissue of fishes is very high. Recommended bioaccumulation factors are given in Table 4.

Manganese

Manganese is homeostatically maintained at constant concentrations in vertebrates and some invertebrates. Thus, the bioaccumulation

Table 2. Recommended bioaccumulation factor relations for strontium

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factor Relation
Fishes	Bone	All waters	Figure 1
	Flesh	All waters	Figure 1
Algae	Whole	All waters	$2x10^3$
Vascular plants (aquatic and emergent)	Whole	All waters	$2x10^2$
Molluscs	Shell	All waters	$6.8x10^4/[Ca]_w$ [a]
	Soft tissues	All waters	$3x10^2$

[a] $[Ca]_w$ = stable calcium concentration of water in ppm.

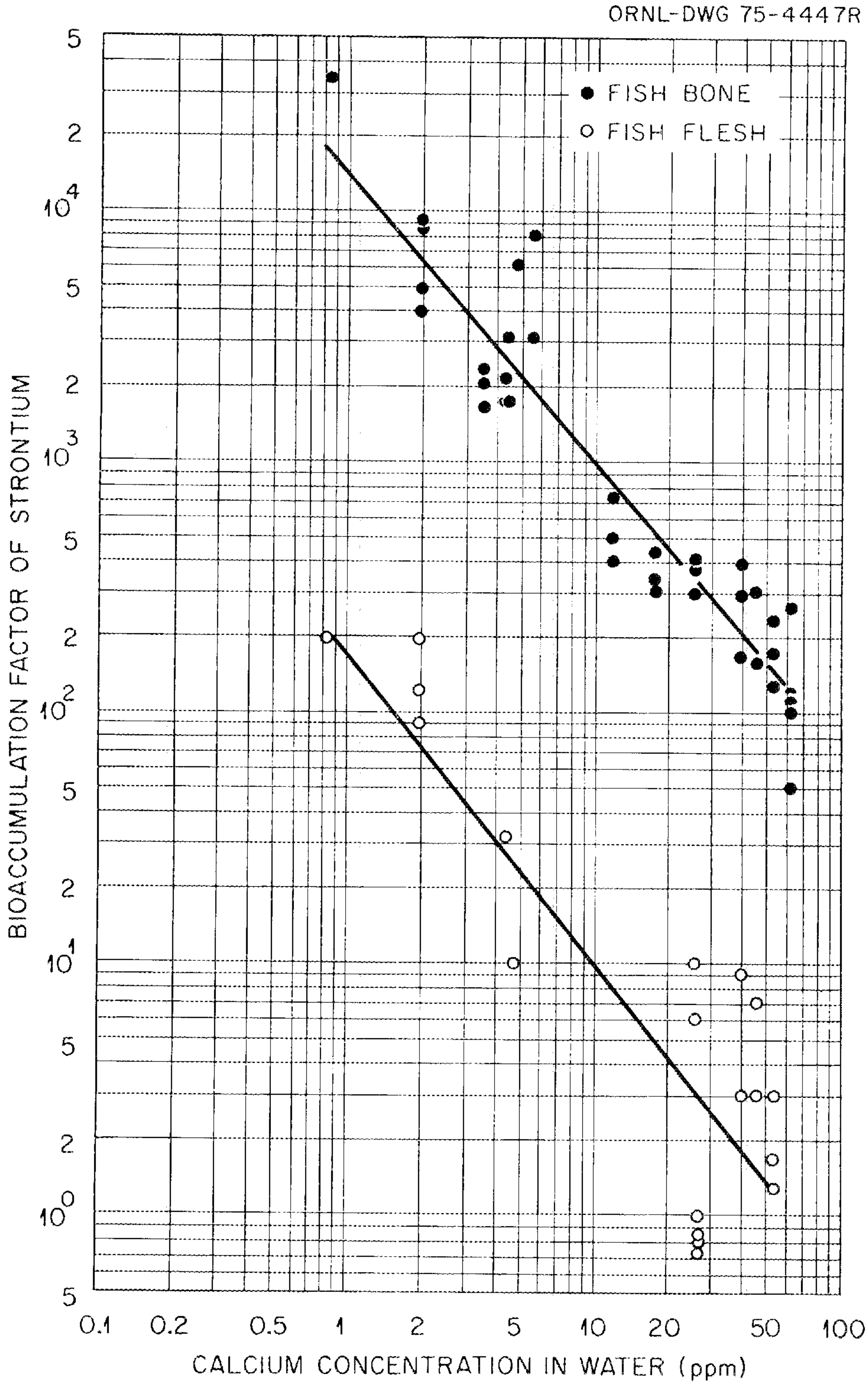

Figure 1 Bioaccumulation factors for strontium in freshwater fishes as a function of calcium concentration in water.

Table 3. Recommended bioaccumulation factor for tritium

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factor
All organisms	All tissues	All waters	1

Table 4. Recommended bioaccumulation factors for stable iodine, iodine-129, and iodine-131

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factor Relation
	Stable Iodine and Iodine-129		
Fishes	Muscle	All waters	50
	Ovary (eggs)		800
	Thyroid tissue		290,000
Crustacea		All waters	340
Phytoplankton		All waters	800
	Iodine-131		
Fishes	Muscle	All waters	40
	Ovary (eggs)		800[a]
	Thyroid tissue		110,000
Aquatic insect larvae	Whole	All waters	400
Molluscs	Soft tissues	All waters	50
	Shell		400
Algae	Whole	All waters	260
Macrophytes	Whole	All waters	120

[a]Default value based on bioaccumulation factor of stable iodine.

factor of manganese in fishes is inversely proportional to manganese concentration in water [Equation (2)]. Recommended bioaccumulation factors are given in Table 5.

Cobalt

Cobalt in solution has a strong tendency to form complexes with dissolved organic matter. Since metals complexed with organic molecules have significantly lower availabilities to plants and animals than free ions, cobalt bioaccumulation factors would be expected to decrease with increasing eutrophy of water. Cobalt bioaccumulation factors, which in this report are based on cobalt concentrations in the soluble phase of water, conform to this expected pattern.

Absorption efficiency of cobalt from food is very low in fishes. This may explain the relatively low bioaccumulation factors of cobalt in fishes. Recommended bioaccumulation factors for cobalt are given in Table 6.

Table 5. Recommended bioaccumulation factor relations for manganese

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factor Relation
Fishes	Whole	All waters	$6.7/[Mn]_w$[a]
	Flesh	All waters	$0.32/[Mn]_w$
Algae	Whole	All waters	10^4
Submerged macrophytes	Whole	All waters	10^4
Floating-leaf macrophytes	Whole	All waters	10^3
Emergent vascular plants	Whole	All waters	10^3
Molluscs:			
Bivalves	Soft	All waters	10^5
	Shell		$3x10^4$
Snails	Soft		$2x10^3$
	Shell		10^4
	Whole		10^4
Crustaceans	Whole	All waters	10^4
Insect larvae	Whole	All waters	10^3

[a] $[Mn]_w$ = stable manganese concentration in water in ppm.

Table 6. Recommended bioaccumulation factors for cobalt (based on isotope concentrations in water after filtration to remove particulates)

Taxon/ Functional Group	Tissue	Environment	Recommended Bioaccumulation Factors
Fishes	Flesh[a]	Mesotrophic and oligotrophic waters	320
	Whole		440
	Flesh[a]	Eutrophic waters	27
	Whole		44
Algae	Whole	All waters	10^4
Emergent vascular plants	Whole	All waters	10^3
Submerged-leaf macrophytes	Whole	Mesotrophic and oligotrophic waters	10^4
		Eutrophic waters	$2x10^3$
Floating-leaf macrophytes	Whole	Mesotrophic and oligotrophic waters	10^3
		Eutrophic waters	$4x10^2$
Molluscs: Bivalves	Soft	Mesotrophic and oligotrophic waters	10^4
		Eutrophic waters	$4x10^2$
Snails	Soft	All waters	10^4
Insect larvae	Whole	All waters	10^4
Tubificid worms	Whole	Eutrophic waters	$5x10^2$
Crustaceans	Whole	All waters	10^3

[a]Bioaccumulation factors of other tissues of fishes may be estimated from relative concentrations of cobalt given in Table 3.6.2.

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1. INTRODUCTION

1.1 Scope

This report analyzes over 200 carefully selected papers to provide concise data sets and methodology for estimation of bioaccumulation factors for selected radionuclides in major biotic components of freshwater environments. The papers were critically reviewed from the standpoint of experimental techniques for each of the selected radionuclides. Since conditions in the laboratory are often unnatural in terms of chemical and ecological relationships, this review is restricted as far as possible to bioaccumulation factors determined for natural systems. Cesium, strontium, iodine, manganese, and cobalt were chosen based on an ORNL survey of releases from 16 light-water-cooled nuclear power stations. This survey showed that the major radiological impact on the aquatic biota and man comes from these radionuclides (Kaye, 1973). Although tritium contributes only a minor fraction of the dose from liquid effluents, it is nonetheless treated too because of a concern that it may be concentrated in passage through food chains. Bioaccumulation factors for freshwater environments are of immediate concern since most nuclear plants (operable, under construction, or planned) are located adjacent to bodies of freshwater.

1.2 Objectives

The objective of this report is to present bioaccumulation factors for the selected radionuclides in the different tissues of major biotic components of freshwater systems. The bioaccumulation factor of a radionuclide in an organism or tissue may exhibit wide variations among sites that are related to differences in stable-element and carrier-element concentrations and other limnological variables. For this reason, bioaccumulation factor relations rather than single bioaccumulation factors are presented to account for site-specific variations.

1.3 Format

General discussions of important aspects of element bioaccumulation are presented in Section 2. We present simple models to illustrate the relation between bioaccumulation factors for radionuclides and the concentrations of their stable-element and carrier-element analogues. The effects of physicochemical form and other factors in causing deviations from these models are also discussed along with methodology for converting stable-element bioaccumulation factors to bioaccumulation factors for shorter-lived radionuclides. Section 3 reviews the literature on bioaccumulation factors for the six selected radionuclides and gives recommended bioaccumulation factor relations based on the literature values.

Reference

Kaye, S. V., Assessing potential radiological impacts to aquatic biota in response to the National Environmental Policy Act (NEPA) of 1969. IN Symposium on Environmental Behavior of Radionuclides Released in the Nuclear Industry, IAEA Symposium, Aix-en-Provence, France, 14-18 May 1973.

2. BIOACCUMULATION FACTOR CONCEPTS

The bioaccumulation factor of an organism or tissue i is the steady-state ratio of radionuclide concentration in the organism or tissue to that in water:

$$BF(R)_i = [R]_i/[R]_w$$

where

$BF(R)_i$ = bioaccumulation factor for radionuclide R in organism or tissue i,

$[R]_i$ = radionuclide concentration (μCi/g fresh weight) in organism or tissue i, and

$[R]_w$ = radionuclide concentration in water (μCi/g), a constant.

Bioaccumulation factors are used to predict radionuclide concentrations in whole organisms or their tissues from knowledge of the radionuclide concentration in water for chronic releases of radionuclides. Bioaccumulation factors also appear in expressions that are used to predict the transient dynamics of radionuclide concentration in organisms (Vanderploeg et al., 1974). Therefore, an understanding of variables that affect bioaccumulation factors is central to understanding steady-state and transient dynamics of radionuclide concentration in aquatic organisms.

2.1 Idealized Bioaccumulation Factor Patterns

Bioaccumulation factors for radionuclides are often portrayed as being related to ambient concentrations of their stable or non-isotopic carrier element analogues according to three idealized patterns (e.g., Fleishman, 1973; Peterson, 1970). The first pattern is that the bioaccumulation factor for radionuclide R in organism or tissue i, $BF(R)_i$, is constant regardless of stable-element or carrier-element concentrations:

$$BF(R)_i = \text{const.} \qquad (2.1.1)$$

The second pattern is that the concentration of an element in an organism or tissue is homeostatically maintained at a constant concentration despite different concentrations of the element in water. This implies that the bioaccumulation factor of a homeostatically controlled stable element is inversely proportioned to its concentration in the water:

$$BF(C)_i = \Sigma_i/[C]_w \quad , \qquad (2.1.2)$$

where Σ_i = concentration of stable element in organisms or tissue i, a constant (μg/g fresh weight), and

$[C]_w$ = concentration of stable element in water (μg/g).

Assuming that the specific activity, that is, the ratio of radionuclide concentration to the stable element concentration (μCi/g stable element) in i is equal to that of the water, the bioaccumulation factor for the radionuclide, $BF(R)_i$, is also inversely proportional to the concentration of the stable element:

$$BF(R)_i = \Sigma_i/[C]_w \quad . \qquad (2.1.3)$$

Taking the natural logarithm of each side of Equation (2.1.2) and (2.1.3) results in the linear relation:

$$\ln BF(R)_i = \ln BF(C)_i = \ln \Sigma_i - \ln[C]_w \ . \qquad (2.1.4)$$

Equation (2.1.4) implies that the plot of $\ln BF(R)_i$ versus $\ln[C]_w$ has a slope of -1. The third pattern is that the bioaccumulation factor for the radionuclide, $BF(R)_i$, is inversely proportional to the concentration of a non-isotopic carrier element in water, a non-isotopic carrier element being nearly chemically similar to but occurring in higher concentrations than the stable-element analogue. The derivation of this relationship follows.

The bioaccumulation factor for radionuclide R is related to $BF(C^*)_i$, the bioaccumulation factor for the carrier element C*, by

$$BF(R)_i = BF(C^*)_i \frac{BF(R)_i}{BF(C^*)_i} \ , \qquad (2.1.5)$$

which may be written as

$$BF(R)_i = BF(C^*)_i \frac{(R/C^*)_i}{(R/C^*)_w} \ , \qquad (2.1.6)$$

where $(R/C^*)_i$ and $(R/C^*)_w$ are ratios of radionuclide concentration to carrier element concentration found in the organism or tissue and in the water, respectively. Assume that the non-isotopic carrier element is homeostatically controlled in the organism, that is:

$$BF(C^*)_i = \Sigma_i^*/[C^*]_w \ , \qquad (2.1.7)$$

where

Σ_i^* = concentration of non-isotopic carrier element in organism or tissue i, a constant (μg/g fresh weight), and

$[C^*]_w$ = concentration of non-isotopic carrier element in water (μg/g).

Combining Equations (2.1.6) and (2.1.7) and letting $q_i = (R/C^*)_i/(R/C^*)_w$,

$$BF(R)_i = \frac{q_i \Sigma_i^*}{[C^*]_w} \quad . \tag{2.1.8}$$

Taking the natural logarithm of each side of Equation (2.1.8),

$$\ln BF(R)_i = \ln q_i \Sigma_i^* - \ln[C^*]_w \quad . \tag{2.1.9}$$

Since Σ_i^* is a constant, a plot of $\ln BF(R)_i$ versus $\ln[C^*]_w$ will have a slope of -1 if q_i is constant.

If water were the immediate source of the radionuclide and carrier element to organism or tissue i, q_i would be called the "observed ratio" (Comar et al., 1956). Since water may not necessarily be the direct source of an element to an organism, we will designate q_i the discrimination coefficient.

On the basis of empirical data on ^{137}Cs and its carrier element potassium, Fleishman (1973) suggested that q_i is not a constant but rather a function of $[C^*]_w$. It can be shown mathematically that if an animal accumulates the carrier and nuclide from water and food, q_i becomes a function of $[C^*]_w$ (see Appendix).

Because the number of sites where steady-state bioaccumulation factors can be determined from the release of radionuclides is limited, bioaccumulation factors for the radionuclides are often estimated from distributions of their stable element analogues. Therefore, the requirement of equal specific activities in organisms and water that we have for Equation (2.1.3) applies as well to Equations (2.1.1) and (2.1.8) or any relation when stable element data are used to estimate bioaccumulation factors.

2.2 Physicochemical Forms of Radionuclides and Availability to Biota

Radionuclides exist in a wide variety of physicochemical forms in natural waters and their different physicochemical forms have different availabilities to aquatic biota. The major physicochemical forms of trace metals that Gibbs (1973) reports for rivers illustrate this diversity:

1. Dissolved ionic species and inorganic associations,
2. Complexes with organic molecules in solution,
3. Adsorbed to solids,
4. Precipitates and coprecipitates on solids (metallic coatings),
5. Incorporated in solid biological materials, and
6. Incorporated in crystalline structures.

Both unicellular and multicellular algae, which generally form the base of the aquatic food web, accumulate elements from the soluble phase. Likewise, some radionuclides are directly accumulated by aquatic animals from the soluble phase. Aquatic vascular plants, which may be important as food to consumers in some systems, accumulate elements from the soluble phase of water and also from the interstitial water of sediment. It has also been demonstrated that radionuclides and metals complexed with organic molecules may have significantly lower availabilities to algae and animals than free ions (Timofeeva et al., 1960).

Aquatic animals, especially filter feeders, may accumulate radionuclides from the suspended phase. The availability of different forms of radionuclides in the suspended phase has not been studied. It can be said that adsorbed (exchangeable) radionuclides and radionuclides in solid biological materials are available to some aquatic consumers. Elements in the crystal structure of clay minerals and other lithogenous detritus are almost completely unavailable.

Radionuclides enter food webs not only from the water but also from bottom sediments, and availability of radionuclides to the food web varies with sediment type. Benthic invertebrates accumulate radionuclides from bottom sediments. Fishes may accumulate radionuclides indirectly from bottom sediment by ingestion of these invertebrates and also directly by incidental ingestion of sediment with prey (Gallegos et al., 1970). Absorption of radionuclides from ingested sediment varies with the nature of the sediment (Gallegos et al., 1970; Eyman and Kitchings, 1974). In addition, different sediments have different capacities for sorption of radionuclides (Friend, 1963; Lomenick and Gardiner, 1965).

2.3 Bioaccumulation Factors and Availability of Radionuclides

For those elements that are homeostatically maintained at constant concentrations in a given organism, the concentration of stable element in the organism is independent of concentrations of stable element in water or its availability from prey, sediment, and different physicochemical forms of the water. If the radionuclide in the organism has the same specific activity as that in water, the bioaccumulation factor for the radionuclide will be given by Equation (2.1.3). In contrast, differences in availability of radionuclides in different sediments and different physicochemical forms in water can lead to marked deviations from the idealized pattern of a constant bioaccumulation factor, Equation (2.1.1).

For the bioaccumulation factor pattern of Equation (2.1.8), we pointed out that q_i is a function of $[C^*]_w$. We note too that if sediment is an important source of radionuclide to the animal, the concentration of carrier in sediment relative to food and water will have an influence on q_i (see Appendix). Moreover, variations in the proportions of radionuclide and carrier element appearing in various physicochemical forms can lead to variations in q_i since the radionuclide or carrier could be preferentially accumulated by the food web.

2.4 Bioaccumulation Factors for Short-Lived Radionuclides

Because of radioactive decay, the bioaccumulation factor for a short-lived radionuclide is expected to be less than the corresponding bioaccumulation factor of the stable element. Since the bioaccumulation factors of some stable elements have been more commonly measured than the bioaccumulation factors of short-lived radionuclides, a methodology is presented to predict the bioaccumulation factor for the short-lived radionuclides from the bioaccumulation factor of the stable element, $BF(C)_i$. The methodology requires knowledge of the elimination coefficient, k, in organism or tissue i.

Treating organism or tissue i as a single compartment, the rate of change in the amount of radionuclide R in i, R_i, (μCi) is

$$\frac{dR_i}{dt} = I_T - R_i\ (k + \lambda), \qquad (2.4.1)$$

and the steady-state amount of stable element C is given by

$$C_i = \frac{I_T'}{k} \qquad (2.4.2)$$

where

I_T = rate of input of radionuclide from all sources (μCi/day),

I_T' = rate of input of the corresponding stable element from all sources (μg/day),

k = elimination coefficient at steady-state concentration of C in i (day^{-1}), and

λ = radioactive decay constant (day^{-1}).

Assuming constant weight of the organism or tissue, the steady-state concentration of radionuclide R and stable element C are given by

$$[R]_i = \frac{I_T}{B\ (k + \lambda)} \qquad (2.4.3)$$

$$[C]_i = \frac{I_T'}{Bk} \qquad (2.4.4)$$

where B = biomass or fresh weight of the organism or tissue i (g).

Assuming all uptake is from water,

$$[R]_i = \frac{a\ [R]_w}{B\ (k + \lambda)} \qquad (2.4.5)$$

$$[C]_i = \frac{a\ [C]_w}{Bk} \quad , \tag{2.4.6}$$

where a = uptake coefficient for element C from water (day^{-1}),

$[R]_w$ = radionuclide concentration in water (μCi/g), and

$[C]_w$ = stable element concentration in water (μg/g).

Dividing Equation (2.4.5) by $[R]_w$ and Equation (2.4.6) by $[C]_w$ gives

$$BF(R)_i = \frac{a}{B\ (k + \lambda)} \tag{2.4.7}$$

$$BF(C)_i = \frac{a}{Bk} \tag{2.4.8}$$

which implies that

$$BF(R)_i = \frac{k}{k + \lambda} \qquad BF(C)_i \tag{2.4.9}$$

for uptake from water. The derivation of Equation (2.4.9) parallels that given by Peterson (1970). Thus if the radioactive decay constant, λ, is significant relative to the elimination coefficient, k, $BF(R)_i$ will be significantly less than $BF(C)_i$. If there are intervening steps in the food chain between the organism or tissue and water, radioactive decay will occur at each step and further diminish $BF(R)_i$. Since the elimination coefficients of prey (or forage) are generally much larger than the elimination coefficients of their respective predators, Equation (2.4.9) will generally not greatly overpredict $BF(R)_i$. Since Equation (2.4.9) will not greatly overpredict the $BF(R)_i$ and since information on food web structure is needed to derive a more accurate $BF(R)_i$, we suggest that Equation (2.4.9) be used to convert bioaccumulation factors for stable elements or long-lived nuclides to bioaccumulation factors for short-lived radionuclides. Tables of k and $[\frac{k}{k+\lambda}]$ values are given when the bioaccumulation factors for shorter-lived isotopes must be estimated from the bioaccumulation factors for stable elements.

2.5 Experimental Error in Determination of Bioaccumulation Factors

In earlier sections we treated the effects of variations in certain environmental conditions on bioaccumulation factors. Variations also result from experimental error, that is, errors resulting from artifacts of analysis, evaluation and presentation of data. Below we discuss some important sources of experimental error and the precautions which can be taken to minimize them. These important sources of experimental error are:

a. Analytical error,

b. Use of literature values for water concentrations,

c. Filtration of water samples,

d. Determination of organism concentration based on dry weight,

e. Averaging isotopic concentrations of different tissues, and

f. Determinations made under nonsteady-state conditions.

2.5.1 Analytical Error

The concentrations of radionuclides or stable elements in most environmental samples are generally so low that serious problems can be encountered in their measurement, especially in water. Eight laboratories participated in an intercalibration study involving spiked freshwater samples (Maletskos, 1972). The results were given as a ratio of the reported values to the "correct" values ± the standard deviation. The results were: ^{54}Mn = 0.91 $\pm$ 0.20, ^{60}Co = 0.97 $\pm$ 0.40, ^{137}Cs = 1.05 $\pm$ 0.27, and ^{90}Sr = 1.09 + 0.46. Iodine and tritium were not included in this study. Although it is not likely that errors in analysis of biological material would parallel those occurring in water analysis, it is obvious that analytical error can result in a significant variation in measured bioaccumulation factors. Significant error may also be associated with measuring stable element concentrations in water, especially for cesium, cobalt and manganese.

Unfortunately, it is not generally possible to evaluate the accuracy of measurements from particular studies. In deriving our bioaccumulation factor relations, we implicitly assumed that variations in bioaccumulation factors arising from analytical error were small in relation to variations caused by environmental variables and that deviations caused by analytical error were randomly distributed.

2.5.2 Use of Literature Values for Water Concentrations

Some of the variation in reported bioaccumulation factors can be attributed to the use of average literature values for stable isotope concentration in water. This is especially true in freshwater environments where element concentrations are more variable than in marine environments. Thompson et al. (1972) derived some average bioaccumulation factors by determining the average stable element concentration in organisms from a number of freshwater habitats and then dividing this average concentration by the average stable element concentration of a number of freshwater habitats. Often the water values were taken from studies different from those giving the concentrations in the organisms. Thus, some error may have been introduced into the average bioaccumulation factor values.

In selecting or calculating bioaccumulation factors for this report only those studies that collected water and organism samples from the same locality are included.

2.5.3 Filtration of Water Samples

Some experimenters have reported bioaccumulation factors based on concentrations of isotopes from filtered as well as unfiltered water samples. Since a significant fraction of some elements in water may be in the suspended phase - in some cases greater than 90% - bioaccumulation factors based on concentrations in filtered samples may be much larger than bioaccumulation factors based on unfiltered concentrations.

To understand the significance of filtered or unfiltered bioaccumulation factors for prediction at proposed nuclear power plants, it is necessary to discuss the nature of the effluent and how radionuclide concentrations in the water at the proposed reactor sites are estimated. Radionuclides in the reactor effluent are in the soluble phase. Concentrations in the body of water of concern are usually estimated from predicted concentrations in the effluent and from water turnover rates. Actual concentrations in the water may be somewhat lower because the bottom sediment may serve as a sink for radionuclides. Since radionuclides may become distributed between soluble and suspended phases, the "predicted concentration" approximates the unfiltered concentration of the radionuclide in the water. Assuming that the specific activity of the radionuclide in the soluble and the suspended phases are equal, an unfiltered bioaccumulation factor will most closely predict the radionuclide concentration in the organism. Generally, filtered bioaccumulation factors will overpredict the radionuclide concentration in organisms.

2.5.4 Determination of Organism Concentration Based on Dry Weight

Bioaccumulation factors expressed in terms of dry weight of the organism are higher than those calculated using wet weight concentrations. Thus, care is taken to ensure that bioaccumulation factors listed in this report are on a fresh weight basis. When dry weight bioaccumulation factors were given in a publication, they were converted to fresh weight bioaccumulation factors if conversion factors between fresh weight and dry weight were available in the same publication. In some cases where conversion factors were not available in the same publication, conversion factors from other sources were used.

2.5.5 Averaging Isotope Concentrations of Different Tissues

In reporting or evaluating bioaccumulation factors care must be taken to specify what tissue or tissues of the organism were analyzed. Certain elements have an affinity for different tissues. For example, strontium concentrates to a higher degree in bony tissues, and iodine concentrates to a higher degree in the thyroid than in other tissues of the organism. Thus, any bioaccumulation factor is incomplete without reference to the specific tissue analyzed. The whole-body bioaccumulation factors, which are often reported, may be different from those of any particular tissue.

Another potential source of uncertainty associated with whole-body bioaccumulation factors is that the organism's gut may contain sediment when the whole-body bioaccumulation factor is determined. The concentration of metals, such as cesium, manganese, cobalt, and strontium in sediment may be significantly higher than the concentrations found in the tissues of the organism. Sediment contamination is most likely a significant factor for benthic invertebrates.

2.5.6 Determinations Made Under Nonsteady-State Conditions

Measurements of organism concentrations in situations where a steady-state condition is not present can result in low or high bioaccumulation factors depending on the circumstance. If an organism is transferred from an uncontaminated environment to an environment with a fixed level of a radioisotope, the time necessary for the establishment of a nearly steady-state concentration is a function of the biological half-time of that element in the organism and the radioactive half-life. Obviously, organism concentrations analyzed prior to the establishment of a steady state will result in low bioaccumulation factors. If the environmental concentration is subjected to sudden variations, the subsequent changes in the organism concentrations may lag behind that of the environment. Low bioaccumulation factors would be expected if determinations are made shortly after a sudden rise in environmental concentration, or after an organism moves from an area of low isotope concentration to an area of high concentration. The opposite situations would result in high bioaccumulation factors.

2.6 Use of Bioaccumulation Factors in Radiation Dose Estimation

The two principal uses of bioaccumulation factors for aquatic biota are: (1) calculation of radiation dose to biota from radionuclides deposited in tissues, and (2) calculation of radiation dose to man from consumption of contaminated biota. In both situations the radiation exposure is within the body and not exterior to it.

2.6.1 Dose to Biota

The internal absorbed dose (rads/year) to organism i from an internally deposited radionuclide is estimated from the energy deposited per gram of tissue and can be expressed mathematically as:

$$D_i = [R]_w \, BF(R)_i \, E \, K \text{ (rads/year)} \qquad (2.6.1)$$

where

D_i = internal absorbed dose (rads/year) to organism i from a radionuclide uniformly deposited in its tissues,

$[R]_w$ = radionuclide concentration in water (μCi/g),

$BF(R)_i$ = bioaccumulation factor for radionuclide R in organism i,

E = effective absorbed energy (MeV) of the radionuclide for the physical dimensions appropriate for the organism i, and

K = conversion factor to convert μCi/g to rads/year.

There is no radioactive decay correction in the above equation because the concentration of the radionuclide in water is assumed to remain constant and the concentration in the biota is in a steady-state with the water. In actual practice, the average annual concentration of the radionuclide in the receiving water at the point of interest is used in dose calculations.

The variable K in the above equation is defined as a conversion factor, but it can represent any type of internal dosimetry model. The variables $[R]_w$, $BF(R)_i$, and E are always required, no matter how specific or general the conversion factor K.

2.6.2 Dose to Man

A bioaccumulation factor is used to estimate the concentration of a radionuclide in the aquatic food (organism or tissue i) consumed by man. The intake rate of a radionuclide (μCi/day) by man is calculated from

$$I_i = [R]_w \, BF(R)_i \, G_i \ (\mu Ci/day) \qquad (2.6.2)$$

where

I_i = man's daily dietary intake rate of a radionuclide (μCi/day) contained in aquatic food organism or tissue i,

$[R]_w$ = radionuclide concentration in water (μCi/g),

$BF(R)_i$ = bioaccumulation factor for radionuclide R in organism or tissue i, and

G_i = man's daily intake rate of organism or tissue i (g/day).

In most exposure situations each organism is contaminated by more than one radionuclide, and man's dietary intake usually includes more than one type of aquatic biota. Therefore, a summation must be performed over all radionuclides and biota to estimate man's total intake of radioactivity.

The internal dose to a reference organ or to the total body of man is calculated using the estimated radionuclide intake rate (μCi/day) for each radionuclide in addition to the following dosimetry factors: fraction of the intake deposited in the organ of reference, the effective elimination constant for each radionuclide in the organ of reference, the effective absorbed energy of each radionuclide in the organ of reference, and the mass of the reference organ.

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3. BIOACCUMULATION FACTORS FOR CESIUM, STRONTIUM, TRITIUM, IODINE, MANGANESE AND COBALT

3.1 CESIUM

3.1.1 Cesium Metabolism

Because of their chemical similarities, cesium and potassium are metabolized somewhat similarly in organisms. In freshwater ecosystems, stable cesium, ^{133}Cs, is an element having concentrations ranging between 0.011 and 1.2 ppb for lakes and rivers (Copeland and Ayers, 1970; Kolehmainen, 1972; Kolehmainen and Nelson, 1969; Merlini et al., 1967; Nelson, 1967). Stable potassium is more abundant in freshwater than cesium and concentrations usually range between 0.2 ppm and 10 ppm.

The bioaccumulation factor for cesium in plants and animals is independent of cesium concentrations found in natural bodies of water (Gertz, 1973; King, 1964; Kolehmainen et al., 1968; Kolehmainen, 1972; Williams and Swanson, 1958). On the other hand, the bioaccumulation factor for potassium in animals is inversely proportional to the potassium concentration in water because the potassium concentration in animals is homeostatically maintained (Kolehmainen, 1972; Kolehmainen and Nelson, 1969; Peterson, 1970). Because of the chemical similarities and relative abundances of cesium and potassium, the bioaccumulation factor for cesium in animals is related to the potassium concentration in water according to Equation (2.1.8). Unlike this pattern for animals, the potassium concentration in water has only a small effect on the cesium bioaccumulation factor in some algae (Williams, 1970) and no effect in other algae (Gertz, 1973).

Cesium elimination from a plant or an animal may be approximated by a single elimination-rate coefficient, k. Table 3.1.1 gives approximate

Table 3.1.1 Biological half-times and elimination coefficients for cesium and [k/(k+λ)] values for ^{136}Cs

Taxon/functional category	Biological half-time (days)	k (days^{-1})	$\frac{k}{k+\lambda}$ for ^{136}Cs	Reference
Unicellular algae	1	0.69	0.93	1.50
Multicellular algae	2	0.35	0.88	1.47
Aquatic vascular plants:				
floating	20	0.035	0.40	1.37
rooted	60[a]	0.012	0.18	1.37
Zooplankton	5	0.14	0.74	1.46
Benthic insect larvae	7	0.10	0.67	1.25
Clams	40	0.017	0.24	1.20
Fishes	100	0.0069	0.12	1.13, 1.23, 1.24, 1.46

[a]Since rooted plants may accumulate part of their burden of cesium from interstitial water of bottom sediments, time to steady-state may be much greater than if uptake were entirely from water above sediments.

biological half-times and elimination rate coefficients values for cesium in different taxa or functional groups. These coefficients must be considered as approximate since they may be influenced by such factors as temperature, size, growth rates, feeding rates, and species differences (Vanderploeg et al., 1974). Most bioaccumulation factors, except those for algae, were derived entirely from stable cesium or from ^{137}Cs (30-year radioactive half-life) concentrations. Cesium-134, having a 2.2-year half-life, will not have an appreciably different bioaccumulation factor from that of stable cesium or ^{137}Cs. Cesium-136, which has a 13-day radioactive half-life, will have an appreciably lower bioaccumulation factor than those of the longer-lived isotopes in some biota; therefore, Table 3.1.1 includes a set of approximate $[k/(k+\lambda)]$ values to convert bioaccumulation factors for stable cesium, or ^{137}Cs, to bioaccumulation factors for ^{136}Cs.

The primary mode of accumulation of cesium and potassium in fishes is generally thought to be via absorption from food (Kolehmainen, 1972); absorption from food may also be a primary mode of uptake for many aquatic invertebrates. However absorption of cesium from ingested sediments may be significant in some systems (Gallegos et al., 1970). In fishes (Kolehmainen, 1972) and other animals (Pendleton et al., 1965), the absorption efficiency of potassium and cesium from food varies but is often high--nearly 100 percent for some foods. The excretion coefficient, k^*, of potassium is on the order of 3 to 4.5 times greater than the excretion coefficient, k, of cesium in fishes (Kolehmainen, 1972) and in terrestrial animals (Fujita et al., 1966, Pendleton et al., 1965). Pendleton pointed out that this will result in a cesium to potassium ratio in the predator that is higher than that of its prey by a factor of about three, assuming that absorption efficiency of cesium is the same as that for potassium. Assuming that the potassium concentration in the predator is the same as that in its prey, the bioaccumulation factor for cesium should increase by about a factor of three with each trophic level. Note that the variable k*/k appears in our general expression for q_i (see Appendix).

3.1.2 Environmental Cesium

In solution, cesium and potassium, like other alkali metals, exist primarily as free ions and do not form inorganic complexes. Because cesium is strongly adsorbed by suspended particulate materials, especially clays, these materials may remove it from the soluble phase to varying degrees. Potassium, too, is sorbed, but its distribution coefficient, K_d, on suspended sediment (that is, the concentration of element sorbed to sediment per concentration in water) is many times smaller than that of cesium (Reynolds and Gloyna, 1964).

Table 3.1.2 shows that the percentage of cesium in the particulate phase increases with clay particle concentration. For freshwater ecosystems the fraction of cesium in water in the suspended phase ranges between 19 and 92 percent, and this fraction appears to be roughly correlated with concentration of suspended solids (Table 3.1.3) . Because potassium is less strongly sorbed than cesium, the percentages of potassium appearing in the suspended phase are very much smaller than those for cesium in these bodies of water. Since algae accumulate cesium, potassium, and other elements from the soluble phase, the availabilities to the food chain of cesium and of cesium relative to potassium become a function of suspended solids concentration and the distribution coefficients of cesium and potassium on

Table 3.1.2 Sorption of ^{134}Cs to various concentrations of clay suspended in distilled water (Garder and Skullberg 1964)

Concentration of clay (ppm)	Percent of cesium sorbed	K_d[a]
16	20	15,600
32	31	13,600
64	43	11,800
128	58	10,800
256	65	7,300

[a]Concentration of element (cesium) sorbed to clay per concentration in water.

Table 3.1.3 Percentage of cesium in water in the particulate phase in some freshwater ecosystems

Location	Mean percent in particulate phase	Suspended solids (ppm) or nature of water body	Separation method	Reference
White Oak Lake, Tennessee	69	Turbid reservoir	Ultracentrifugation: Particles > 0.7 μ	1.44
Clinch River, Tennessee[a]	82-92	25-185	Ultracentrifugation: Particles > 0.7 μ	1.44
Tennessee River, Tennessee[a]	19-30	9-22	Ultracentrifugation: Particles > 0.7 μ	1.44
Lake Maggiore, Italy	28	Mesotrophic Lake	0.7-μ filter	1.35
Experimental pond[b]	58	21	0.45-μ filter	1.10

[a]Range of means of three river stations.

[b]Based on five water values sampled between 24 and 80 days after introduction of radionuclides.

the solids. Thus, the q_i values and bioaccumulation factors of cesium become functions of suspended-solids concentrations and the distribution coefficients. Since the proportions of cesium and of cesium relative to potassium in the soluble phase are expected to vary greatly among environments, the bioaccumulation factors for cesium in algae and the q_i values and bioaccumulation factors of animals may vary greatly among bodies of water having the same potassium concentrations. We shall see in the following sections that much of the scatter in bioaccumulation factors for cesium is apparently related to the proportion of cesium in the particulate phase.

The food web accumulates cesium not only from the soluble phase of water but also from suspended and bottom sediments. Filter feeders may accumulate cesium adsorbed to particulate matter. Benthic invertebrates obtain cesium adsorbed to ingested bottom sediments. Fishes may accumulate cesium from bottom sediment by ingestion of these invertebrates and also by ingestion of sediment with prey (Eyman and Kitchings, 1974; Gallegos et al., 1970). Gallegos et al. (1970) reported that concentration of ^{137}Cs in trout in a montane lake was higher than could be explained by the trouts' accumulation from their prey. Their experiments indicated that this difference could be explained by the trout ingesting small amounts of sediment. In fishes, absorption efficiency of cesium from ingested clay varies with clay type: 8% for illite, 65% for kaolinite, and 85% for montmorillonite (Eyman and Kitchings, 1974). Since cesium is available from ingested sediments, differences in sediment type among water bodies may be another contributor to the variance in bioaccumulation factors among bodies of water.

3.1.3 Review of Cesium Bioaccumulation Factors

This section presents bioaccumulation factors and bioaccumulation factor relations for the following categories of aquatic biota:

1. Fishes
2. Algae
3. Macrophytes (vascular aquatics)
4. Emergent vascular plants
5. Invertebrates--molluscs, insects, and crustaceans
6. Amphibians
7. Waterfowl

Except for algae, all bioaccumulation factors for cesium were derived from distributions of stable cesium or from steady-state distributions of ^{137}Cs in natural or semi-natural bodies of water. Since ecological relationships in aquaria do not completely represent the natural system, bioaccumulation factors derived from laboratory experiments have limited predictive value. Moreover, in Table 3.1.1, it is seen that the biological half-times of ^{137}Cs in some biota range between approximately 40 and 100 days. Laboratory experiments have not been run for a sufficient time period in many cases for these longer-term components to attain steady state.

3.1.3.1 Cesium Bioaccumulation Factors for Fishes

Bioaccumulation factors have usually been reported for flesh or whole bodies of fishes. Since concentrations of cesium and potassium do not vary greatly among organs (Nelson, 1967), flesh or whole-body bioaccumulation factors apply to other organs as well.

Potassium concentration in freshwater fishes is about 3 mg/g fresh weight and varies little among species (Kolehmainen, 1972; Peterson, 1970). The range of potassium concentrations in fishes from both freshwater and marine environments is from 2.1 to 4.5 mg/1, (Fleishman, 1973;

Kolehmainen, 1972; Kolehmainen and Nelson, 1969). The range of potassium concentrations in freshwater and marine fishes is small considering that potassium concentration in the water range between 0.2 and 380 ppm. Clearly, the potassium concentrations are homeostatically maintained. Thus, the bioaccumulation factor for potassium is given by Equation (2.1.7). Since the bioaccumulation factor for potassium is given by Equation (2.1.7), the bioaccumulation factor for cesium is given by Equation (2.1.8). Owing to the variable availabilities of cesium relative to potassium in different environments, the value q_i varies from one environment to another.

Bioaccumulation factors calculated from concentrations of fallout-derived ^{137}Cs in water and fishes collected in Finnish lakes (Kolehmainen et al., 1967; Kolehmainen et al., 1968) are given in Table 3.1.4. Within lakes, the ratio of highest bioaccumulation factor to lowest bioaccumulation factor ranges between 2.4 and 7.1. As expected, the highest bioaccumulation factors are seen for high trophic-level fishes, namely, perch and pike, which are piscivores. The highest bioaccumulation factors are seen for the oligotrophic lake having a potassium concentration in water of 1 ppm. Note that the bioaccumulation factors in the oligotrophic lake are very much higher than the bioaccumulation factors in the other two lakes having the same potassium concentration but higher concentrations of organic matter. The difference appears to be related to the lower concentration of organic matter in the oligotrophic lake. Most probably, however, this difference is related to the lower suspended solids concentration (both organic and inorganic solids) expected in the oligotrophic lake. The bioaccumulation factors for the fishes from the lake having a water concentration of 3.5 ppm potassium are more than 25 times lower than that of the oligotrophic lake having a

Table 3.1.4 Mean ^{137}Cs bioaccumulation factors for whole fishes from Finnish Lakes (Kolehmainen et al. 1967, 1968)[a]

Potassium concentration (ppm)	Lake type[b]	$KMnO_4$ consumption (ppm)[c]	Bioaccumulation Factors: Perch (*Perca fluviatilis*)[d]	Pike (*Esox lucius*)	Burbot (*Lota vulgaris*)	Roach (*Leuciscus rutilus*)	Bream (*Abramis brama*)	Whitefish (*Coregonus* spp.)	Highest BF / Lowest BF
0.8	O	32	11,000	5,600	4,400	3,100	----	2,500	4.4
0.9	D-O	65	9,900	5,500	3,300	1,800	----	----	5.5
1.0	O	6	----	14,000	----	5,200	----	3,700	3.8
1.0	D-O	120	----	2,900	2,800	1,200	800	----	3.6
1.0	E	43	1,800	2,100	2,200	900	----	----	2.4
1.8	D-E	45	5,000	2,900	2,500	1,200	700	----	7.1
3.5	E	75	300	430	480	200	75	----	4.0

[a]Based on whole fish concentrations derived from 1964 and 1965 data; ^{137}Cs concentration in water (unfiltered) from 1964 data.

[b]O = oligotrophic; D-O = dystrophic-oligotrophic; E = eutrophic.

[c]A measure of organic content of water.

[d]Large perch.

potassium concentration of 1 ppm. If the value of q_T were constant, the bioaccumulation factors in the 3.5-ppm lake would be 3.5 times lower than the bioaccumulation factors in the 1-ppm lake. Thus, the potassium concentration in the water alone is not very useful for predicting the bioaccumulation factors for cesium in fishes.

The conclusion that the potassium concentration in the water alone is not very useful for predicting cesium bioaccumulation factors is corroborated by the data in Table 3.1.5. The table gives bioaccumulation factors for cesium as related to potassium and suspended solids concentrations. The bioaccumulation factors for cesium in fishes were taken from Table 3.1.6. The highest bioaccumulation factors are seen for bodies of water of low turbidity. The bioaccumulation factors of bluegills and catfish found in the clear water body with a potassium concentration of 1.4 ppm are more than six and seven times the respective bioaccumulation factors of bluegills and catfish from the very turbid water body with a potassium concentration of 1.3 ppm. Also, the highest bioaccumulation factor for largemouth bass is in a lake of high potassium concentration and low turbidity.

Most investigators have reported bioaccumulation factors that are calculated from concentration of cesium isotopes in unfiltered water samples; however, some have reported bioaccumulation factors based on concentration in the water after particulate material has been filtered off. We shall denote a "filtered" bioaccumulation factor by BF*. Assuming all particulate material is filtered out, a BF* represents an upper-limit bioaccumulation factor for that and other environments of similar potassium concentration and sediment type since cesium on suspended sediments is to

Table 3.1.5 Bioaccumulation factors for cesium as related to potassium and suspended solids concentrations[a]

Potassium concentration in water (ppm)	Suspended solids concentration (ppm)	Bioaccumulation factors of Cesium: Bluegill (*Lepomis macrochirus*)		Largemouth bass (*Micropterus salmoides*)		Catfish (*Ictalurus*)		Reference
		^{137}Cs	Stable Cs	^{137}Cs	Stable Cs	^{137}Cs	Stable Cs	
1.3	Very high (25-185)		140				160	1.36 1.44
1.4	Low (1.3)	900		1200		1200		1.19 1.34
1.8	High	270	310	350	620			1.27
9.8	Low			1600	570			1.7

[a]Based on concentrations of cesium in unfiltered water samples.

Table 3.1.6 Bioaccumulation factors for cesium in fishes

Species	Tissue	K concentration (ppm)	BF, fallout ^{137}Cs	BF*,[a] fallout ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable cesium	BF*, stable cesium	Water treatment method	Feeding habits	Location	Suspended solids concentration (ppm)	Reference
Salmo trutta	F[b]	0.4	3,900								Lake Trawsfynyd, England	Low[c]	1.42
Perca fluviatilis	F	0.4	5,800							P[d]	Lake Trawsfynyd, England	Low	1.42
Salmo trutta	F	0.3	3,000								English river	Low	1.42
Salmo trutta	F	0.4	1,400								English river	Low	1.42
Salmo trutta	F	1.9	920								English river	Low	1.42
Salmo trutta	F	3.8	940								English river	Low	1.42
Salmo trutta	F	4.1	260								English river	Low	1.42
Salmo trutta	F	4.1	380								English river	Low	1.42
Pomoxis annularis	F	1.3					500	6,400	0.7 μ[e]	P	Clinch River, Tennessee	25-185[f]	1.36
Aplodinotus grunniens	F	1.3					350	4,400	0.7 μ		Clinch River, Tennessee	25-185	1.36
Roccus chrysops	F	1.3					640	8,000	0.7 μ	P	Clinch River, Tennessee	25-185	1.36
Ictalurus punctatus	F	1.3					160	2,000	0.7 μ		Clinch River, Tennessee	25-185	1.36
Lepomis macrochirus	F	1.3					140	1,700	0.7 μ		Clinch River, Tennessee	25-185	1.36
Dorosoma cepedianum	W[g]	1.8			310	810	400		0.7 μ		White Oak Lake, Tennessee	High[h]	1.27
Notemigonus crysoleucas	W	1.8			420	1,100	510		0.7 μ		White Oak Lake, Tennessee	High	1.27
Carassius auratus	W	1.8			200	530	690		0.7 μ		White Oak Lake, Tennessee	High	1.27
Lepomis microlophus	W	1.8			180	460	320		0.7 μ		White Oak Lake, Tennessee	High	1.27
Lepomis macrochirus	W	1.8			270	710	310		0.7 μ		White Oak Lake, Tennessee	High	1.27
Chaenobryttus gulosus	W	1.8			240	630	430		0.7 μ		White Oak Lake, Tennessee	High	1.27

Table 3.1.6 (continued)

Species	Tissue	K concentration (ppm)	BF, fallout ^{137}Cs	BF*, fallout ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable cesium	BF*, stable cesium	Water treatment method	Feeding habits	Location	Suspended solids concentration (ppm)	Reference
Micropterus salmoides	W	1.8			350	910	620		0.7 μ	P	White Oak Lake, Tennessee	High	1.27
Salmo gairdneri	F	1.7		12,000					A-ER[i]		East Twin Lake, Colorado	Low	1.12
Micropterus salmoides	W	9.8		1,625			570			P	Wintergreen Lake, Michigan	Low	1.7
Perca flavescens	W	9.8		1,200			1,600			P	Wintergreen Lake, Michigan	Low	1.7
Hybrid sunfish	W	9.8		500			250				Wintergreen Lake, Michigan	Low	1.7
Erimyzon sucetta kennerlii	W	9.8		790			300				Wintergreen Lake, Michigan	Low	1.7
Perca fluviatilis	W	2.1	2,900				1,100			P	Lake Maggiore, Italy		1.2,1.3
Perca fluviatilis	W	2.6	1,400				140			P	Lake Varese, Italy		1.2,1.3
Perca fluviatilis	W	2.1	2,400				660			P	Lake Comabbio, Italy		1.2,1.3
Perca fluviatilis	W	1.3	5,100				930			P	Lake Monate, Italy		1.2,1.3
Scardinius erythrophtalmus	W	2.1	1,300				560				Lake Maggiore, Italy		1.2,1.3
Scardinius erythrophtalmus	W	2.6	890				140				Lake Varese, Italy		1.2,1.3
Scardinius erythrophtalmus	W	2.1	1,000				370				Lake Comabbio, Italy		1.2,1.3
Scardinius erythrophtalmus	W	1.3	1,400				350				Lake Monate, Italy		1.2,1.3

Table 3.1.6 (continued)

Species	Tissue	K concentration (ppm)	BF, fallout ^{137}Cs	BF*, fallout ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable cesium	BF*, stable cesium	Water treatment method	Feeding habits	Location	Suspended solids concentration (ppm)	Reference
Lepomis gibbosis	W	2.1	790				440				Lake Maggiore, Italy		1.2,1.3
Lepomis gibbosis	W	2.6	420				110				Lake Varese, Italy		1.2,1.3
Lepomis gibbosis	W	2.1	690				420				Lake Comabbio, Italy		1.2,1.3
Lepomis gibbosis	W	1.3	2,800				790				Lake Monate, Italy		1.2,1.3
Lepomis macrochirus	F	1.4[j]			900						Par Pond, South Carolina	1.3	1.19
Micropterus salmoides	F	1.4			1,200					P	Par Pond, South Carolina	1.3	1.19
Ictalurus natalis	F	1.4			1,200						Par Pond, South Carolina	1.3	1.19
Oncorhyncus kisutch	F	1.6						2,700	0.45 μ	P	Lake Michigan	Low	1.5
Perca flavescens	F	1.6						3,300	0.45 μ	P	Lake Michigan	Low	1.5
Salvelinus namaycush namaycush	F	1.6						2,400	0.45 μ	P	Lake Michigan	Low	1.5
Salmo trutta	F	1.6						3,400	0.45 μ	P	Lake Michigan	Low	1.5
Coregonus clypeaformis	F	1.6						1,500	0.45 μ		Lake Michigan	Low	1.5
Pomolobus pseudoharengus	W	1.6						1,400	0.45 μ		Lake Michigan	Low	1.5
Osmerus mordax	W	1.6						1,900	0.45 μ		Lake Michigan	Low	1.5
Notropus hudsonius	W	1.6						2,100	0.45 μ		Lake Michigan	Low	1.5

Table 3.1.6 (continued)

Species	Tissue	K concentration (ppm)	BF, fallout ^{137}Cs	BF*, fallout ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable cesium	BF*, stable cesium	Water treatment method	Feeding habits	Location	Suspended solids concentration (ppm)	Reference
Cyprinus carpio	W	2.6					500				Carp pond		1.53

[a]A BF* is a bioaccumulation factor calculated from isotope concentration in water after filtration.

[b]F = flesh.

[c]Low ≤ 30 ppm suspended solids.

[d]P = piscivorous.

[e]Pore size of filter.

[f]Struxness et al. (1967).

[g]W = whole fish.

[h]High > 30 ppm suspended solids.

[i]A-ER = anion-exchange resin.

[j]Personal communication.

some degree available to the food web. Whenever possible, we calculated both unfiltered and filtered bioaccumulation factors.

The unfiltered and filtered bioaccumulation factors in Table 3.1.6 are categorized according to source of isotope: fallout-derived ^{137}Cs, chronic release of ^{137}Cs, and stable cesium. In fishes from Italian lakes (Table 3.1.6), Bortoli et al. (1967) reported that bioaccumulation factors calculated from distributions of fallout-derived ^{137}Cs were on the average four times higher than bioaccumulation factors calculated from stable cesium distributions. Clearly, fallout-derived ^{137}Cs is somehow more available to the food web than stable cesium. A greater proportion of fallout ^{137}Cs, which enters watersheds from the atmosphere as a soluble radionuclide, may be in the soluble phase or on exchangeable sites (adsorbed) of clay particles. The recently introduced ^{137}Cs may not be in isotopic equilibrium with the stable cesium on the clay particles, especially stable cesium at non-exchangeable sites. According to Eyman's study of Wintergreen Lake, Michigan (1972), bioaccumulation factors for ^{137}Cs were generally higher than bioaccumulation factors for stable cesium. In contrast, however, Kolehmainen's study (1972) of White Oak Lake (Table 3.1.6), showed that bioaccumulation factors for ^{137}Cs originating from a chronic release were about the same as bioaccumulation factors derived from stable cesium data. This may result from the time history of ^{137}Cs input into White Oak Lake. During the years preceding Kolehmainen's study, ^{137}Cs input was higher than during the study. We also note that the stable cesium concentration in water was measured on water that had received gross filtration to remove larger particles. All these patterns would suggest that the bioaccumulation factor for ^{137}Cs near a power station is somewhat higher--roughly by a factor of two--than the bioaccumulation factor for stable cesium. The magnitude of the difference may depend on suspended solids concentrations.

We have shown that the bioaccumulation factor for cesium is highly variable from one environment to another and that much of this variation derives from differing proportions of radiocesium relative to potassium in the soluble phase and possibly sediment type. Unfortunately, the data necessary for accurate quantification of bioaccumulation factor relations are not available. Approximate relations, however, can be stated.

To obtain these relations we plotted the bioaccumulation factor for cesium in fishes against the potassium concentration in water on logarithmic graph paper, as shown in Figure 3.1.1 Unfiltered bioaccumulation factors were used so as to better predict radionuclide concentration in the organism. For turbid environments a filtered bioaccumulation factor may be more than 10 times greater than an unfiltered bioaccumulation factor since more than 90 percent of the cesium in water may be in the particulate phase. Fishes were grouped into two categories: piscivorous fishes and non-piscivorous fishes. By inspection we chose the relation

$$BF(Cs)_i = 1.5 \times 10^4/[K]_w \qquad (3.1.1)$$

as an upper-bound relation for piscivorous fishes and

$$BF(Cs)_i = 5 \times 10^3/[K]_w \qquad (3.1.2)$$

as an upper-bound relation for non-piscivorous fishes, where $[K]_w$ has units of ppm. Equations (3.1.1) and (3.1.2) are applicable to environments of low turbidity since they will greatly overpredict bioaccumulation

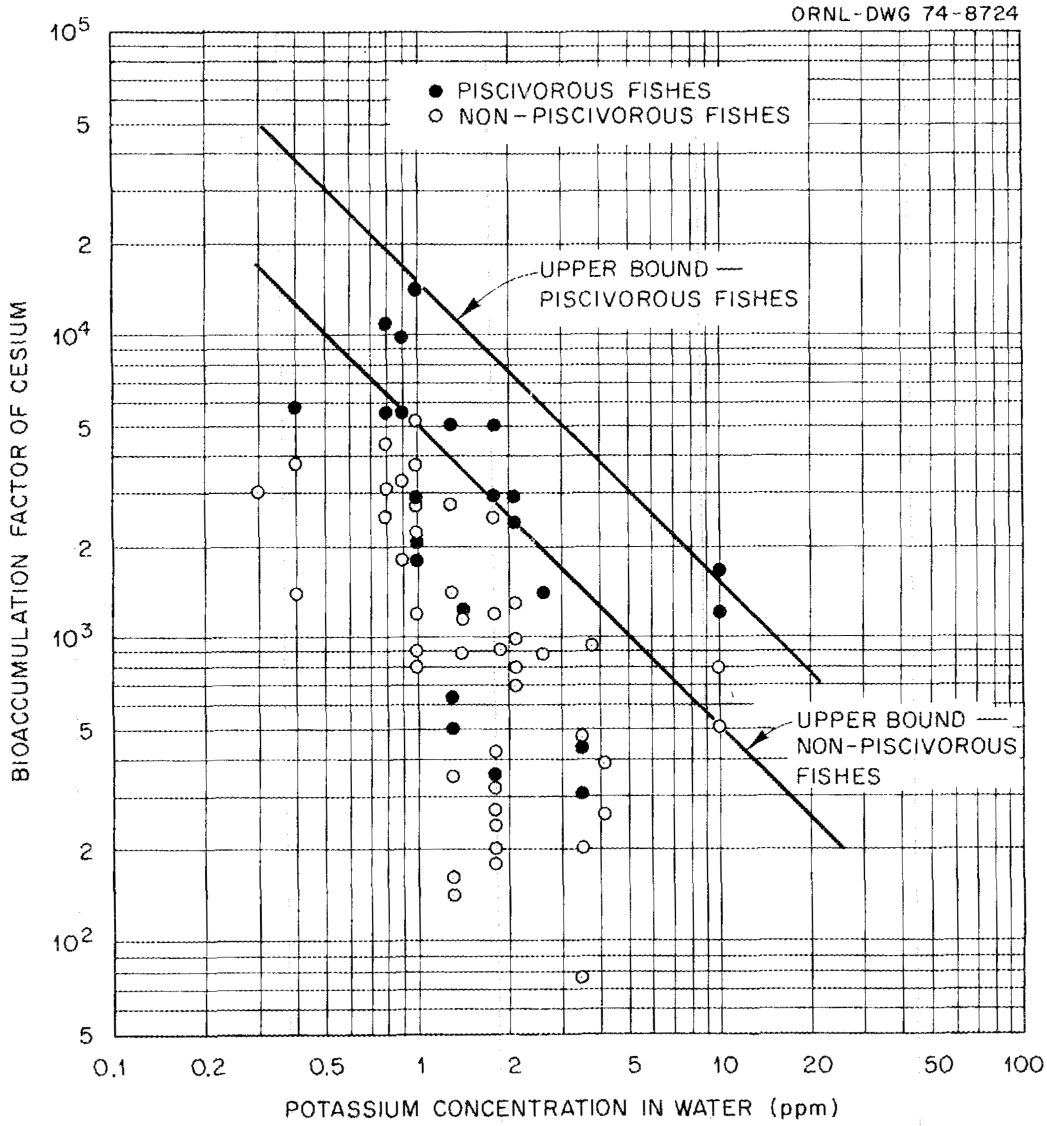

Figure 3.1.1 Bioaccumulation factors for ^{137}Cs and stable cesium in freshwater fishes as a function of potassium concentration in water. (All data in Tables 3.1.4 and 3.1.6 were plotted except (1) stable cesium bioaccumulation factors from studies in which ^{137}Cs bioaccumulation factors were reported and (2) BF*s.)

factors for fishes from turbid environments. Dividing Equations (3.1.1) and (3.1.2) by five gives rough approximations for the respective relations for fishes in turbid waters; that is,

$$BF(Cs)_i = 3 \times 10^3/[K]_w \qquad (3.1.3)$$

for piscivorous fishes from turbid waters and

$$BF(Cs)_i = 1 \times 10^3/[K]_w \qquad (3.1.4)$$

for non-piscivorous fishes from turbid waters. We may consider waters having greater than 50 ppm suspended solids as being turbid based on the data in Table 3.1.3. Alternatively, bioaccumulation factors for particular species may be estimated from the raw data in Tables 3.1.4 and 3.1.6. The bioaccumulation factor selected should preferably come from an environment having the same suspended solids and potassium concentrations as the environment to which it is applied. Even if this is done, a degree of uncertainty remains because of the unspecified effect of sediment type on bioaccumulation factors as well as error associated with measurement and experimental design.

3.1.3.2 Cesium Bioaccumulation Factors for Algae

As can be seen from the bioaccumulation factors for *Cladophora glomerata* and *Pithophora oedegonia* (Table 3.1.7), the potassium concentration in water has little effect on the bioaccumulation factor for cesium in multicellular algae. Recently, Gertz (1973) has shown that the potassium concentration has no effect on the bioaccumulation factor for cesium in *Chlamydomonas reinhardii*, a unicellular green alga. He found that the

Table 3.1.7 Bioaccumulation factors for cesium in freshwater algae[a]

Taxon/functional category	K concentration (ppm)	Bioaccumulation factors: ^{137}Cs BF	^{137}Cs BF*[b]	Stable Cs BF	Stable Cs BF*	Location	Source of ^{137}Cs	Reference
Blue-green algae:								
Plecotonema boryanum	3.6	100[c]				Lab study		1.22
Unidentified	1.4	1,200				Par Pond, South Carolina	Chronic-release	1.19
Unidentified	—	3,400				Connecticut River	Chronic-release	1.1
Diatoms:								
Navicula seminulum	14	130[c]				Lab study		1.22
Mixed species[d]	1.6				2900[e]	Lake Michigan		1.4
Euglenophyta:								
Euglena intermedia	8	700				Lab study		1.51
Green algae, multicellular:								
Chara sp.	—	20				Lab study		1.45
Chara aspera	—	18				Lab study		1.45
Chara braunii	1	60				Experimental channel receiving river water	Chronic-release	1.16
Chara fragilis	—	1,000				Lake Bol shoe Missavo, Russia		1.33
Chara fragilis	—	36				Lab study		1.45
Chara tomentosa	—	80				Lake Bol shoe Missavo, Russia		1.33
Cladophora fracta	—	120				Lab study		1.45
Cladophora glomerata	0.1	170				Lab study		1.48
Cladophora glomerata	30	59				Lab study		1.48
Cladophora glomerata	—	160				Lab study		1.45
Chlorella pyrenoidosa	8	150				Lab study		1.51
Gonium pectorale	10	140				Lab study		1.51
Mougeotia sp.	9.8	330		500		Wintergreen Lake, Michigan	Fallout	1.7

Table 3.1.7 (continued)

Taxon/functional category	K concentration (ppm)	Bioaccumulation factors: ^{137}Cs BF	^{137}Cs BF*[b]	Stable Cs BF	Stable Cs BF*	Location	Source of ^{137}Cs	Reference
Green algae, multicellular:								
Nitella hialina	—	170				Lake Bol shoe Missavo, Russia		1.33
Oedogonium sp.	—	1,200				1.1 hectare pond	Chronic-release	1.39
Oedogonium sp.	0.3	3,000				Valkealampi Lake, Finland	Spike input	1.32
Oedogonium vulgare	1	790				Lab study		1.51
Pithophora oedogonia	0.1	170				Lab study		1.48
Pithophora oedegonia	30	56				Lab study		1.48
Rhizoclonium hieroglyphicum	1	1,500				Lab study		1.51
Scenedesmus acuminatus	—	39				Lab study		1.45
Scenedesmus quadricauda	—	28				Lab study		1.45
Spirogyra communis	13	220				Lab study		1.51
Spirogyra crassa	—	28				Lab study		1.45
Spirogyra ellipsospora	1	340				Lab study		1.51
Spirogyra sp.	—	190				Lab study		1.45
Spirogyra sp.	—	380				Lake Bol shoe Missavo, Russia		1.33
Spirogyra sp	1	150				Experimental channel receiving river water	Chronic-release	1.15
Stigeoclonium lubricum	14	89[c]				Lab study		1.22
Tolipellopsis stelligera	—	220				Lake Bol shoe Missavo, Russia		1.33
Ulothrix sp.	—	1,400				Lab study		1.52
Ulothrix sp.	—	460				Isere River	Fallout	1.52
Vaucheria walzii	1	500				Experimental channel receiving river water	Chronic-release	1.15

Table 3.1.7 (continued)

Taxon/functional category	K concentration (ppm)	Bioaccumulation factors: ^{137}Cs BF	Bioaccumulation factors: ^{137}Cs BF*[b]	Bioaccumulation factors: Stable Cs BF	Bioaccumulation factors: Stable Cs BF*	Location	Source of ^{137}Cs	Reference
Green algae, multicellular:								
Mixed species	—	1,500-4,000				Concrete-lined pond	Spike input	1.39
Mixed species	1.8	230	600[f]			White Oak Lake, Tennessee	Chronic-release	1.27
Green algae, unicellular:								
Chlamydomonas sp.	8	52				Lab study		1.51
Chlamydomonas moewussi	—	130-370				Lab study		1.26
Oocystis elliptica	10	670				Lab study		1.51
Functional categories:								
Phytoplankton	1.6				1900[e]	Lake Michigan		1.4

[a]Dry-weight bioaccumulation factors were divided by 10 to convert them to wet-weight bioaccumulation factors.

[b]A BF* is a bioaccumulation factor calculated from isotope concentration in water after filtration.

[c]Average over all temperatures.

[d]Includes samples where more than 80% of the cells were diatoms.

[e]Geometric mean of bioaccumulation factors calculated from each pair of filtered (0.45 μ pore size) water concentrations and non-zero "corrected" concentrations in phytoplankton given in Appendix II of Copeland and Ayers (1970).

[f]Water ultracentrifuged — equivalent to 0.7 μ pore-size filter.

cesium bioaccumulation factor was decreased by increasing sodium concentration in water. Both Harvey (1969, 1970) and Gertz (1973) reported that the cesium bioaccumulation factor is unaffected by non-lethal temperatures.

Both the nutrient concentration and general health of the algae influence the cesium bioaccumulation factor for algae, since the bioaccumulation factor is higher at higher phosphate concentrations (Gertz, 1973) and the bioaccumulation factor of dead cells is lower than that of live cells (Williams, 1970; Williams, 1960). Moreover, the cesium bioaccumulation factor for multicellular algae is higher in flowing water than in still water (Watts and Harvey, 1963).

Most bioaccumulation factors for algae in Table 3.1.7 are near or less than 10^3. We therefore recommend that for algae in general a bioaccumulation factor of 10^3 be used.

3.1.3.3 Cesium Bioaccumulation Factors for Vascular Aquatic Plants

Some vascular aquatic plants may obtain a significant fraction of their body content of cesium and other elements from root uptake of elements from interstial water of bottom sediments. Diffusion of elements into interstitial water from water overlying sediments is slow. This may explain why the bioaccumulation factor of fallout-derived ^{137}Cs in Nuphar sp. (Table 3.1.8) is much lower than the corresponding bioaccumulation factor of stable cesium. Since most bioaccumulation factors in Table 3.1.8 are less than or about 10^3, we recommend that a bioaccumulation factor of 10^3 be used for aquatic vascular plants.

Table 3.1.8 Bioaccumulation factors for cesium in aquatic vascular plants[a]

Species	K concentration in water (ppm)	BF, fallout ^{137}Cs	BF, spike input of ^{137}Cs	BF, chronic release of ^{137}Cs	BF, stable Cs	BF*, stable Cs[b]	Location	Reference
Azolla fuliculoides	—		250				Concrete-lined pond	1.39
Ceratophyllum demersum	—		400-1,000				Concrete-lined pond	1.39
Ceratophyllum sp.	9.8	370			490		Wintergreen Lake, Michigan	1.7
Elodea canadensis	—		1,000				Concrete-lined pond	1.39
Elodea canadensis		390					Isere River	1.52
Elodea canadensis	2.6				1,400		200-m^2 carp pond	1.53
Lemna minor	—		500				Concrete-lined pond	1.39
Lemna minor	—	500					Isere River	1.52
Nuphar luteum	3.5	130					Finnish lakes	1.30
Nuphar luteum	1.8	410					Finnish lakes	1.30
Nuphar luteum	1.0	1,000					Finnish lakes	1.30
Nuphar luteum	0.8	1,500					Finnish lakes	1.30
Nuphar luteum	0.6	1,100					Finnish lakes	1.30
Nuphar sp.	9.8	770			3,800		Wintergreen Lake, Michigan	1.7
Nymphea lutea	2.1				280	390	Lake Maggiore, Italy	1.3,1.35
Potomogeton pectinatus	—		700				Concrete-lined pond	1.39

[a]Dry-weight bioaccumulation factors were multiplied by 0.15 to convert them to wet-weight bioaccumulation factors.

[b]A BF* is a bioaccumulation factor calculated from isotope concentration in water after filtration.

3.1.3.4 Cesium Bioaccumulation Factors for Emergent Vascular Plants

Most bioaccumulation factors for emergent vascular plants (Table 3.1.9) are less than or near 10^3; we therefore recommend that for this group in general a bioaccumulation factor of 10^3 be applied.

3.1.3.5 Cesium Bioaccumulation Factors for Invertebrates

There appears to be much interspecific variation in bioaccumulation factors for crustaceans (Table 3.1.10). Nevertheless, most bioaccumulation factors in Table 3.1.10 are near 10^3, therefore, a bioaccumulation factor of 10^3 is recommended for invertebrates in general. However, bioaccumulation factor for shells of invertebrates may be much lower.

3.1.3.6 Cesium Bioaccumulation Factors for Amphibians

The few bioaccumulation factors collected for amphibians (Table 3.1.11) suggest that a bioaccumulation factor of 10^4 be used.

3.1.3.7 Cesium Bioaccumulation Factors for Waterfowl

The few bioaccumulation factors collected for waterfowl (Table 3.1.12) suggest that a bioaccumulation factor of $3x10^3$ be used.

Table 3.1.9 Bioaccumulation factors for cesium in emergent vascular plants

Species/part	K concentration in water (ppm)	BF, fallout ^{137}Cs	BF, spike input of ^{137}Cs	BF, chronic release of ^{137}Cs	BF, stable Cs	Location	Reference
Equisetum fluviatile[a]	3.5	480				Finnish lakes	1.30
Equisetum fluviatile[a]	1.8	1,100				Finnish lakes	1.30
Equisetum fluviatile[a]	1.0	1,600				Finnish lakes	1.30
Equisetum fluviatile[a]	0.9	1,200				Finnish lakes	1.30
Equisetum fluviatile[a]	0.8	7,400				Finnish lakes	1.30
Phragmites communis	2.6				1,200	200-m^2 carp pond	1.53
Polygonium lapathifolium:	—					1.1 hectare pond	1.39
seeds				240			
Polygonium persicaria:	—					Concrete-lined pond	1.39
leaves			600				
seeds			400				
Scirpus acutus:	—						1.38,1.39
culms			90			Concrete-lined pond	
seeds			400			Concrete-lined pond	
leaves			100			Concrete-lined pond	
roots			400			Concrete-lined pond	
seeds				70		1.1 hectare pond	
Scirpus americanus:	—					Concrete-lined pond	1.39
culms			50				
seeds			300				
Typha latifolia:	—					Concrete-lined pond	1.38,1.39
leaves			250				
seeds			100				
roots			250				

[a]Dry-weight bioaccumulation factors were multiplied by 0.20 to convert them to wet-weight bioaccumulation factors.

Table 3.1.10 Bioaccumulation factors for cesium in aquatic invertebrates

Taxon	K concentration in water (ppm)	BF, fallout ^{137}Cs	BF*, fallout ^{137}Cs[a]	BF, spike input ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable Cs	BF*, stable Cs	Location	Reference
Crustaceans:										
Gammarus lacustris	1.7		1,000					340[b]	East Twin Lake, Colorado	1.12,1.18
Hyallela azteca	—			11,000					Concrete-lined pond	1.39
Mysis relicta	1.6							60[c]	Lake Michigan	1.6
Zooplankton (primarily copepods)	1.6							600[d]	Lake Michigan	1.4
Zooplankton (*Daphnia* and *Cyclops*)	1.7							340[b]	East Twin Lake, Colorado	1.18
Insect larvae:										
Chaoborus sp.	9.8	830					1,600		Wintergreen Lake, Michigan	1.7
Chironomid larvae	1.8				640	1,600			White Oak Lake, Tennessee	1.27
Erythemis callocata	—			800					Concrete-lined pond	1.39
Ischnura sp.	—			800					Concrete-lined pond	1.39
Snails:										
Limnea stagnalis	2.6						1,300		200-m^2 carp pond	1.53
Radix japonica	—			600					Concrete-lined pond	1.39

Table 3.1.10 (continued)

Taxon	K concentration in water (ppm)	BF, fallout ^{137}Cs	BF*, fallout ^{137}Cs[a]	BF, spike input ^{137}Cs	BF, chronic release of ^{137}Cs	BF*, chronic release of ^{137}Cs	BF, stable Cs	BF*, stable Cs	Location	Reference
Clams:										
Lampsillis radiata	1.4								Small stream, South Carolina	1.20
soft tissues					>220[e]					
shell					>25					

[a]A BF* is a bioaccumulation factor calculated from isotope concentration in water after filtration.

[b]Spike input of stable cesium.

[c]Water concentration taken from Copeland and Ayers (1970).

[d]Bioaccumulation factor equals geometric mean of "uncorrected" concentrations of cesium in zooplankton from Appendix II of Copeland and Ayers (1970) divided by mean cesium concentration in water.

[e]On the basis of biological half-life given by the author, the clams had not fully attained steady-state with the ^{137}Cs concentration of water into which they were placed.

Table 3.1.11 Bioaccumulation factors for ^{137}Cs in amphibians (Pendleton and Hanson, 1958)

Species/Tissue	BF
Bullfrog tadpole (*Rana catesbeiana*):	
entire	2,600
gut	4,500
flesh	1,000
Spadefoot toad tadpole (*Scaphiopus hammondi intermontanus*):	
entire	6,000
Bullfrog adult:	
muscle	8,000

Table ~~3.1.12~~ Bioaccumulation factors for ^{137}Cs in waterfowl[a] ~~(Pendleton and Hanson, 1958)~~

Species/Tissue	BF
American coot	
(*Fulica a. americana*):	
muscle	1,800
liver	2,200
bone	800
Common mallard	
(*Anas platyrhynchos*):	
muscle	2,000
liver	2,500
bone	700
Ruddy duck	
(*Oxyura jamaicensis rubida*):	
muscle	2,200
liver	2,800
bone	900

[a] Adapted from Pendleton and Hanson, 1958

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The numbers in the left-hand margin correspond to the reference numbers used in the tables of section 3.1.

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3.2 STRONTIUM

3.2.1 Strontium Metabolism

As a result of their chemical similarities and the greater abundance of calcium in nature, calcium is a non-isotopic carrier for strontium. Since calcium is homeostatically maintained at constant concentrations in fishes (Agnedal, 1967; Ophel and Judd, 1973; Peterson, 1970; Reed and Nelson, 1973; Suzuki et al., 1972; Templeton and Brown, 1964) and presumably other animals, the bioaccumulation factor for calcium in animals is given by Equation (2.1.7). The bioaccumulation factor for strontium is then given by Equation (2.1.8). These relations do not hold for algae since the calcium concentration in water does not appear to greatly influence the bioaccumulation factor for strontium in algae (Kevern, 1964a; Williams, 1970).

The primary uptake of calcium and strontium in fishes, and probably in many aquatic invertebrates, occurs directly from the water. The gill membranes of fishes are the primary sites of calcium and strontium uptake (Nelson, 1966; Suzuki et al., 1972; Templeton and Brown, 1964). Only about one-tenth of the calcium and strontium taken up by fishes is through the food chain (Agnedal, 1966). Since uptake from water is the primary pathway, the food chain dynamics of calcium and strontium are of little importance in the determination of bioaccumulation factors for strontium in aquatic organisms. Thus, the discrimination coefficient q_i in Equation (2.1.8) is not a function of trophic level in fishes because the primary source of strontium to fishes is water. The discrimination coefficient has also been shown to be independent of the calcium concentration in water (Agnedal, 1967; Ophel and Judd, 1973; Peterson, 1970; Reed and Nelson, 1973; Suzuki et al., 1972; Templeton and Brown, 1964).

3.2.2 Environmental Strontium

Strontium is commonly found in nature with calcium, its closest chemical analogue (Nelson, 1961). Strontium concentrations in freshwater rivers and lakes may range between 0.004 and 0.23 ppm (Nelson, 1967; Ophel et al., 1972; Suzuki et al., 1972; Templeton and Brown, 1964). The more abundant element calcium is found in freshwater rivers and lakes in concentrations ranging from 1.9 to 114 ppm (Austin, 1963; Beninson et al., 1966; Nelson, 1967; Ophel et al., 1972; Suzuki et al., 1972; Templeton and Brown, 1964). Strontium has physicochemical properties similar to calcium and, like calcium, appears mainly in ionic form in water (Templeton and Brown, 1964). Strontium and calcium are not strongly sorbed by suspended particulate materials in the water. The fraction of strontium present in the suspended phase is given for a number of freshwater ecosystems in Table 3.2.1. The fraction of strontium in the suspended phase ranges between 0.9 and 10%. Since q_i is independent of $[Ca]_w$ and because both calcium and strontium have nearly the same physicochemical forms in water, the discrimination coefficient is not expected to vary much among sites.

3.2.3 Review of Strontium Bioaccumulation Factors

Bioaccumulation factors are discussed for the following categories of aquatic biota:

Table 3.2.1 Percentage of radiostrontium in water in the particulate phase in some freshwater ecosystems

Water body	Mean % in particulate phase	Suspended solids (ppm)	Separation method	Reference
White Oak Lake, Tennessee[a]	2	31 (mean)	Ultracentrifugation: Particles >0.7 μ	2.24
Clinch River, Tennessee[a]	2-9	25-185	Ultracentrifugation: Particles >0.7 μ	2.24
Tennessee River, Tennessee[a]	9-10	9-22	Ultracentrifugation: Particles >0.7 μ	2.24
Experimental Pond[b]	0.9	21	0.45-μ filter	2.7

[a]Range of mean of three river stations.

[b]Based on five water values sampled between 24 and 80 days after introduction of radionuclides.

1. Fishes
2. Algae
3. Aquatic and Emergent Macrophytes
4. Molluscs.

The bioaccumulation factors were examined for taxa in each of the above categories. For all taxa except algae, the strontium bioaccumulation factors were derived only from concentrations in natural or semi-natural bodies of water.

3.2.3.1 Strontium Bioaccumulation Factors for Fishes

A regression analysis of $\ln BF(Ca)_i$ versus $\ln [Ca]_w$ for brown trout (Templeton and Brown, 1964) and for three species of fish from Swedish lakes (Agnedal, 1966) yielded a linear plot with a slope of approximately -1 and a high degree of negative correlation (Peterson, 1970). The bioaccumulation factor for calcium can thus be given by Equation (2.1.7). Since calcium is a non-isotopic carrier for strontium, the bioaccumulation factor for strontium is given by Equation (2.1.8). The analyses by Peterson (1970) of data from Templeton and Brown (1964) and Agnedal (1967) clearly showed that the q_i values for freshwater fishes are independent of the concentrations of calcium in the water. This implies that the regression of $\ln BF(Sr)_i$ on $\ln [Ca]_w$ has a slope of about -1.

The strontium bioaccumulation factors, calcium concentrations in fishes and water, and q_i values for strontium in fish bone and flesh are given in Table 3.2.2 for studies that reported calcium concentration in water. Only data based on several measurements of strontium in fishes and water are listed. Some studies have listed values for the strontium discrimination coefficients in bone and no bioaccumulation factors. In these studies, the discrimination coefficients were calculated from strontium and calcium concentrations in ashed bone. The ratio between ash weight and fresh weight in bone is not constant for all species or for all ashing procedures. Since a fresh weight cannot be assumed for a given ashed weight, bioaccumulation factors for strontium could not be calculated from these studies.

Because q_i is not a function of feeding habits, we decided to derive a relation between the strontium bioaccumulation factor and $[Ca]_w$ for all fish data combined. To do this, regressions of ln $BF(Sr)_i$ versus ln $[Ca]_w$ for all fishes combined were made. Results of these regressions for fish flesh and bone are given in Table 3.2.3 and in Figure 3.2.1. The slope of the regression for fish flesh is not significantly different ($P > 0.05$) from the expected slope of -1. The slope of the regression for bone, although significantly different from -1 ($P < 0.05$), is not appreciably different. To predict the bioaccumulation factor of strontium in fishes we recommend use of Figure 3.2.1 or the relation given in Table 3.2.3.

3.2.3.2 Strontium Bioaccumulation Factors for Algae

The bioaccumulation factors for strontium in algae are given in Table 3.2.4 for both field and laboratory studies. In a laboratory study, Kevern (1964) showed that the bioaccumulation factor for strontium in the unicellular green alga Oocytis eremosphaeria was independent of the calcium concentration in the water. In Table 3.2.4, the multicellular algae Pithophora and Cladophora glomerata (Williams, 1970) show only a small decrease in the bioaccumulation factor for strontium accompanying a very large increase in calcium

Table 3.2.2 Bioaccumulation factors and discrimination coefficients for strontium in fishes

Species	Tissue	Ca Concentration Tissue (ppm)	Ca Concentration Water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Perch	Bone Muscle	23100 480	2.0	4,030 92	0.35 0.38	Lake Langsjon, Sweden	2.1
Perch	Bone	31000	40.0	170	0.22	Lake Storacksen, Sweden	2.1
Perch	Bone Muscle	31600 390	46.0	160 3	0.23 0.35	Lake Erken, Sweden	2.1
Perch	Bone Muscle	43600 430	54.0	130 1.3	0.16 0.16	Lake Glisstjarn, Sweden	2.1
Perch	Bone	39400	4.4	1,840	0.21	Lake Ulkesjon, Sweden	2.1
Perch	Bone	40000	12	410	0.12	Lake Viggen, Sweden	2.1
Perch	Bone	34200	18	330	0.17	Lake Orlangen, Sweden	2.1
Perch	Bone	39800	26	380	0.25	Lake Magelungen, Sweden	2.1
Perch	Bone	31700	63	100	0.20	Lake Sovdesjon, Sweden	2.1
Perch	Bone	30200	63	110	0.23	Lake Nittsjon, Sweden	2.1
Pike	Bone Muscle	42500 910	2.0	4,930 125	0.23 0.27	Lake Langsjon, Sweden	2.1

Table 3.2.2 (continued)

Species	Tissue	Ca Concentration Tissue (ppm)	Ca Concentration Water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Pike	Bone	65400	40.0	290	0.18	Lake Storacksen, Sweden	2.1
	Muscle	470		3	0.25		
Pike	Bone	51000	54.0	180	0.19	Lake Glisstjarn, Sweden	2.1
	Muscle	440		1.7	0.21		
Pike	Bone	47000	26	310	0.17	Lake Magelungen, Sweden	2.1
	Muscle	590		6	0.26		
Pike	Bone	35100	2.0	8,810	0.50	Lake Rogen, Sweden	2.1
Pike	Bone	44500	4.4	2,190	0.22	Lake Ulkesjon, Sweden	2.1
Pike	Bone	42000	4.5	1,770	0.19	Lake M. Mollesjon Sweden	2.1
Pike	Bone	44500	12	480	0.13	Lake Viggen, Sweden	2.1
Pike	Bone	51200	18	450	0.16	Lake Orlangen, Sweden	2.1
Pike	Bone	20100	63	50	0.16	Lake Sovdesjon, Sweden	2.1
Pike	Bone	43100	63	120	0.18	Lake Nittsjon, Sweden	2.1
Roach	Bone	38600	2.0	9,170	0.48	Lake Langsjon, Sweden	2.1
	Muscle	880		198	0.45		
Roach	Bone	45300	40.0	390	0.35	Lake Storacksen, Sweden	2.1
	Muscle	930		9	0.39		
Roach	Bone	47000	46.0	310	0.30	Lake Erken, Sweden	2.1
	Muscle	800		7	0.41		
Roach	Bone	37300	54.0	240	0.35	Lake Glisstjarn, Sweden	2.1
	Muscle	550		3	0.30		

Table 3.2.2 (continued)

Species	Tissue	Ca Concentration Tissue (ppm)	Ca Concentration Water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Roach	Bone Muscle	30600 580	26	420 10	0.36 0.45	Lake Magelungen, Sweden	2.1
Roach	Bone	38700	12	730	0.23	Lake Viggen, Sweden	2.1
Roach	Bone	19500	18	350	0.32	Lake Orlangen, Sweden	2.1
Roach	Bone	25800	63	120	0.29	Lake Sovdesjon, Sweden	2.1
Roach	Bone	50200	63	270	0.34	Lake Nittsjon, Sweden	2.1
Salmo trutta	Bone Muscle	47800 203	0.84	34,559 198	0.61 0.82	Loch Glutt, United Kingdom	2.26
Salmo trutta	Bone Muscle	80000 140	4.90	6,260 9.8	0.38 0.34	Windermere, United Kingdom	2.26
Salmo trutta	Bone Muscle	80400 289	4.50	3,194 32	0.18 0.50	River Prysor, United Kingdom	2.26
Pomoxis annularis	Bone Flesh	 135	27	 1.00	0.29 0.20	Clinch River, Tennessee	2.17
Aplodinatus grunniens	Bone Flesh	 122	27	 0.86	0.21 0.19	Clinch River, Tennessee	2.17
Roccus chrysops	Bone Flesh	 106	27	 0.74	0.21 0.19	Clinch River, Tennessee	2.17
Ictalurus punctatus	Bone Flesh	 89	27	 0.73	0.20 0.22	Clinch River, Tennessee	2.17
Lepomis macrochirus	Bone Flesh	 157	27	 0.79	0.20 0.13	Clinch River, Tennessee	2.17

Table 3.2.2 (continued)

Species	Tissue	Ca Concentration Tissue (ppm)	Ca Concentration Water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Perca flavescens	Whole Bone		5.7	472 3,225		Perch Lake, Ontario	2.20
Ictalurus nebulosus	Whole Bone		5.7	981 8,000		Perch Lake, Ontario	2.20
Semotilus margarita	Whole		5.7	456		Perch Lake, Ontario	2.20
Lipomis gibbosus	Whole		5.7	1,250		Perch Lake, Ontario	2.20
Perca fluviatalis	Whole		18.9	130		Lake Maggiore, Italy	2.4
Perca fluviatalis	Whole		9.5	110		Lake Varese, Italy	2.4
Perca fluviatalis	Whole		24.2	120		Lake Comabbio, Italy	2.4
Perca fluviatalis	Whole		31.4	300		Lake Monate, Italy	2.4
Scardinius erythrophtalmus	Whole		18.9	180		Lake Maggiore, Italy	2.4
Scardinius erythrophtalmus	Whole		9.5	210		Lake Varese, Italy	2.4
Scardinius erythrophtalmus	Whole		24.2	180		Lake Comabbio, Italy	2.4
Scardinius erythrophtalmus	Whole		31.4	380		Lake Monate, Italy	2.4

Table 3.2.2 (continued)

Species	Tissue	Ca Concentration		$BF(Sr)_i$	q_i	Location	Reference
		Tissue (ppm)	Water (ppm)				
Lepomis gibbosus	Whole		18.9	110		Lake Maggiore, Italy	2.4
Lepomis gibbosus	Whole		9.5	90		Lake Varese, Italy	2.4
Lepomis gibbosus	Whole		24.2	90		Lake Comabbio, Italy	2.4
Lepomis gibbosus	Whole		31.4	280		Lake Monate, Italy	2.4
Lepomis gibbosus	Bone Flesh		3.6[a]	2,400 <48		Par Pond, South Carolina	2.9
Micropterus salmoides	Bone Flesh		3.6[a]	1.700 <48		Par Pond, South Carolina	2.9
Ictalurus natalis	Bone Flesh		3.6[a]	2,100 <48		Par Pond, South Carolina	2.9
Perca flavescens	Bone		6.3		0.21	Perch Lake, Ontario	2.21
Ictalurus nebulosus	Bone		6.3		0.54	Perch Lake, Ontario	2.21
Perca flavescens	Bone		26.7		0.18	Lake Huron, Ontario	2.21
Catostomos catostomos	Bone		26.7		0.21	Lake Huron, Ontario	2.21
Dorosoma cepedianum	Bone		26.7		0.25	Lake Huron, Ontario	2.21

Table 3.2.2 (continued)

Species	Tissue	Ca Concentration Tissue (ppm)	Ca Concentration Water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Cyprinus carpio	Bone		26.7		0.53	Lake Huron, Ontario	2.21
Carassius auratus	Bone		10.8		0.86	Lake Toyanogata, Japan	2.25
Carassius auratus	Bone		10.2		0.78	Lake Mikata, Japan	2.25
Carassius auratus	Bone		11.8		0.92	Lake Barato, Japan	2.25

[a]Personal communication.

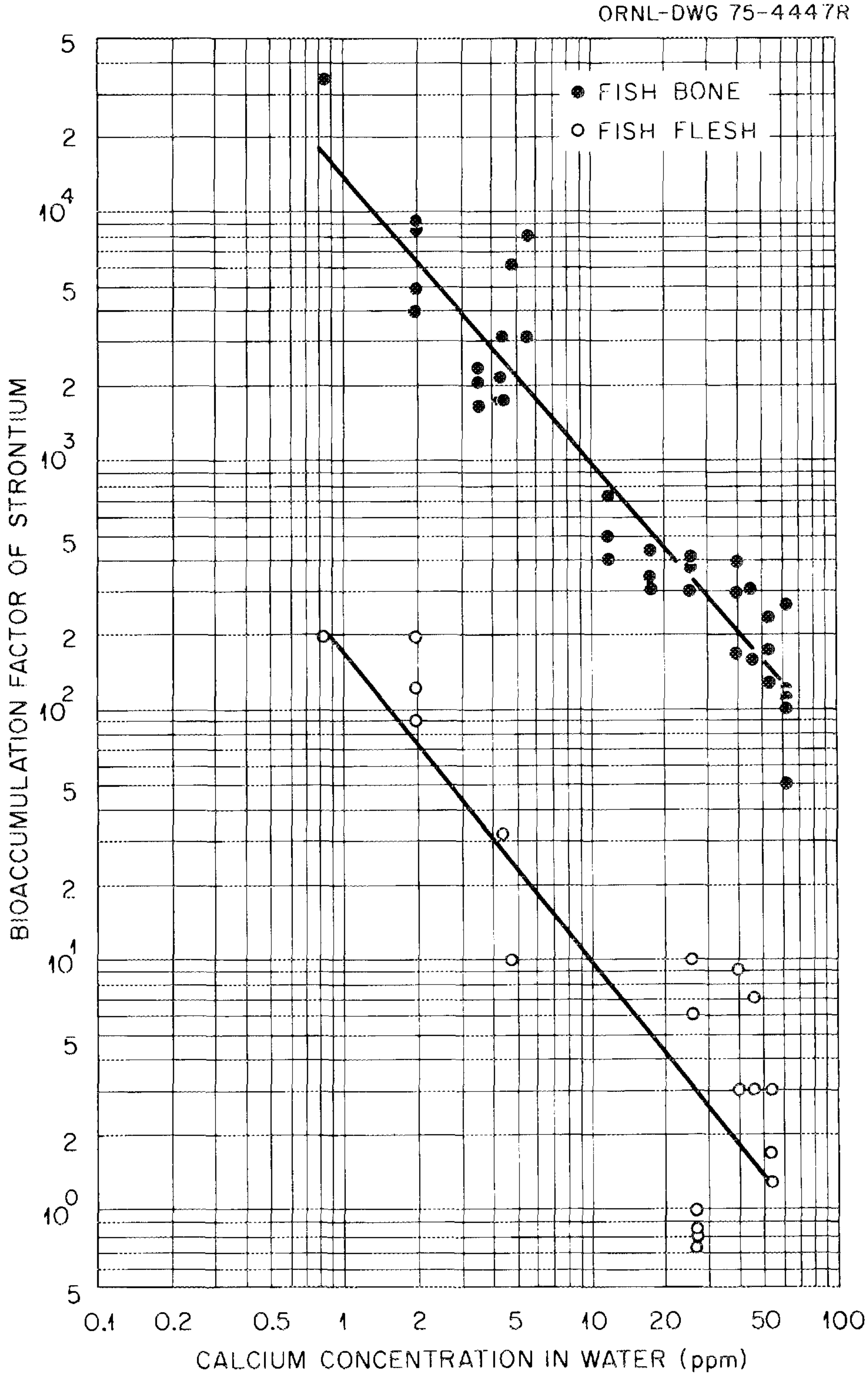

Figure 3.2.1 Bioaccumulation factors for strontium in freshwater fishes as a function of calcium concentration in water.

Table 3.2.3. Parameters from linear regressions between $\ln BF(Sr)_i$ and $\ln [Ca]_w$ for bone and flesh of fishes. [To predict $BF(Sr)_i$ use the relation $BF(Sr)_i = \exp(\text{intercept} + \text{slope} \times \ln [Ca]_w)$.]

Tissue	Intercept ± SE	Slope ± SE	R^2
Bone	9.59 ± 0.18	-1.15 ± 0.06	0.901
Flesh	5.18 ± 1.11	-1.21 ± 0.37	0.736

Table 3.2.4 Bioaccumulation factors and discrimination coefficients for strontium in algae[a]

Species	Ca concentration in water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Mixed blue-green algae	3.6[b]	600		Par Pond, South Carolina	2.9
Cladophora glomerata	0.1	2,200		Laboratory	2.28
Cladophora glomerata	1.0	2,100		Laboratory	2.28
Cladophora glomerata	30	1,000		Laboratory	2.28
Cladophora sp.	23		1.8	Lough Neagh, U.K.	2.26
Navicula seminulum	17	70[c]		Laboratory	2.12
Nitella sp.	23		0.38	Lough Neagh, U.K.	2.26
Pithophora oedogonia	0.1	2,100		Laboratory	2.28
Pithophora oedogonia	1.0	2,300		Laboratory	2.28
Pithophora oedogonia	30	1,280		Laboratory	2.28
Plectonema boryanum	39	240[c]		Laboratory	2.12

Table 3.2.4 (continued)

Species	Ca concentration in water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Spirogyra sp.	5.7	120		Perch Lake, Ontario	2.20
Spirogyra sp.	5.4	900		Experimental Channel	2.8
Stigeoclonium lubricum	10.5	120[c]		Laboratory	2.12
Vaucheria walzii	5.4	1,400		Experimental Channel	2.8

[a]Dry-weight bioaccumulation factors were divided by 10 to convert them to wet-weight bioaccumulation factors.

[b]Personal communication.

[c]Average over all temperatures.

concentration in the water. Thus it may be concluded that the bioaccumulation factor for strontium in algae is relatively independent of the calcium concentration in the water. Until additional data are available, a strontium bioaccumulation factor of 2×10^3 is recommended for algae.

3.2.3.3 Strontium Bioaccumulation Factors for Aquatic and Emergent Macrophytes

The bioaccumulation factors for aquatic and emergent macrophytes are found in Table 3.2.5. Discrimination coefficients are given when available. The data do not permit us to infer whether the strontium bioaccumulation factor is dependent on the calcium concentration in the water. On the basis of these few data, we recommend a bioaccumulation factor of 2×10^2 for aquatic and emergent macrophytes.

3.2.3.4 Strontium Bioaccumulation Factors for Molluscs

Table 3.2.6 gives the experimental data obtained on bioaccumulation of strontium in mollusc shells. The data in Table 3.2.6 and Equation (2.1.8) were used to calculate a bioaccumulation factor relation for strontium in mollusc shells. Mean values of q_i, the discrimination coefficient, and Σ_i^*, the concentration of calcium in mollusc shells, are taken from Table 3.2.6 to derive the relation. For mollusc shells, the strontium bioaccumulation factor relation derived was

$$BF(Sr)_i = \frac{6.8 \times 10^4}{[Ca]_w} ,$$

where $[Ca]_w$ has units of ppm.

Table 3.2.5 Bioaccumulation factors and discrimination coefficients for strontium in aquatic and emergent macrophytes

Species	Ca concentration in water (ppm)	$BF(Sr)_i$	q_i	Location	Reference
Braseria schreberi	6.3		0.55	Perch Lake, Ontario	2.21
Ceratophyllum demersum	5.7	220		Perch Lake, Ontario	2.20
Elodea canadensis	23		1.2	Lough Neagh, U.K.	2.26
Fontinalis sp.	6.0	240	1.2	Perch Lake, Ontario	2.20, 2.21
Myriophyllum spicatum	2.4		1.2	Devoke Water, U.K.	2.26
Myriophyllum spicatum	23		1.5	Lough Neagh, U.K.	2.26
Nuphar variegatum	6.3		1.8	Perch Lake, Ontario	2.21
Nymphea odorata	6.3		0.90	Perch Lake, Ontario	2.21
Pontederia cordata	6.3		1.4	Perch Lake, Ontario	2.21
Potamogeton natans	2.4		1.5	Devoke Water, U.K.	2.26
Potamogeton pectinatus	23		1.7	Lough Neagh, U.K.	2.26
Potamogeton perfoliatus	23		0.84	Lough Neagh, U.K.	2.26
Potamogeton pusillus	6.0	190	2.0	Perch Lake, Ontario	2.20, 2.21
Scirpus fluitans	2.4		0.57	Dover Water, U.K.	2.26
Scirpus subterminalis	5.7	30		Perch Lake, Ontario	2.20
Sparangium fluctuans	5.7	150		Perch Lake, Ontario	2.20
Sparangium sp.	2.4		0.89	Devoke Water, U.K.	2.26
Utricularia vulgaris	5.7	160		Perch Lake, Ontario	2.20

Table 3.2.6 Bioaccumulation factors and discrimination coefficients for strontium in mollusc shells

Species	Ca Concentration		$BF(Sr)_i$	q_i	Location	Reference
	Shell (g/g)	Water (ppm)				
Unioninae						
Elliptio dilatatus	0.40	27	2,988	0.20	Clinch River, Tennessee	2.16
Elliptio crassidens	0.40	27	3,781	0.25	Clinch River, Tennessee	2.16
Pleurobema cordatum	0.40	27	2,913	0.20	Clinch River, Tennessee	2.16
Fusconaia subrotunda	0.40	27	2,671	0.18	Clinch River, Tennessee	2.16
Lampsilinae						
Actinonaias carinata gibba	0.40	27	2,691	0.18	Clinch River, Tennessee	2.16
Ligumia recta latissima	0.40	27	2,640	0.18	Clinch River, Tennessee	2.16
Lampsilis ovata	0.40	27	3,246	0.22	Clinch River, Tennessee	2.16
Unioninae						
Quadrula mantenevra	0.40	19	2,567	0.12	Tennessee River, Tennessee	2.16
Quadrula pustulosa	0.40	19	3,217	0.15	Tennessee River, Tennessee	2.16
Elliptio crassidens	0.40	19	3,117	0.15	Tennessee River, Tennessee	2.16
Pleurobema cordatum	0.40	19	3,767	0.18	Tennessee River, Tennessee	2.16

Table 3.2.6 (continued)

Species	Ca Concentration		BF(Sr)$_i$	q_i	Location	Reference
	Shell (g/g)	Water (ppm)				
Unioninae (cont'd)						
Amblema costata	0.40	19	3,202	0.15	Tennessee River, Tennessee	2.16
Megalonaias gigantea	0.40	19	2,995	0.14	Tennessee River, Tennessee	2.16
Cycloraias tuberculata	0.40	19	3,235	0.15	Tennessee River, Tennessee	2.16
Lamsilinae						
Plagiola lineolata	0.40	19	3,170	0.15	Tennessee River, Tennessee	2.16
Mean Σ_i	0.40		Mean q_i	0.17		

There is not a great deal of experimental data available on bioaccumulation of strontium in the soft tissue of molluscs. A recommended bioaccumulation factor for strontium in the soft tissue of molluscs was made on the basis of the limited experimental data that are available (Brungs, 1965; Harvey, 1969; Merlini et al., 1967; Nelson, unpublished data; Ophel, 1963). The recommended strontium bioaccumulation factor for the soft tissue of molluscs is 3×10^2. It must be noted however that this value is given for bodies of water with calcium concentrations of approximately 20 to 30 ppm. The bioaccumulation factor is of course a function of the calcium and strontium concentration in the particular body of water and bioaccumulatior factors as high as 720 were found for waters of very low calcium content (Ophel, 1963).

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3.3 TRITIUM

Tritium was included in this report because of a concern that the relative kinetics of tritium and protium resulting from tritium's heavier mass ("isotope effect") might lead to a preferential accumulation of tritium over protium. Experiments in aquatic systems indicate that this does not occur and that the bioaccumulation factor for tritium is less than or about equal to the bioaccumulation factor for protium, which has a bioaccumulation factor approximately equal to 1. This section discusses the movement and distribution of protium in aquatic organisms and reviews experiments that compare the behavior of tritium and protium in aquatic systems. In the discussion the term hydrogen is used to refer to both isotopes of the element collectively.

3.3.1 Hydrogen Bioaccumulation

Hydrogen atoms in living organisms can be separated into two major categories or pools. The first pool, tissue-water hydrogen (TWH), is defined as all hydrogen atoms present in water molecules within the organism. The second pool, tissue-bound hydrogen (TBH), is defined as all hydrogen present in organic molecules, such as proteins, fats, and carbohydrates. As an illustration, a fish consists of approximately 75% water and 25% dry tissue. The percentages of water and dry tissue, however, do not accurately reflect the relative sizes of the two hydrogen pools because on a weight basis water contains 11% hydrogen, while dry fish muscle contains 8%. As a result the tissue-water hydrogen constitutes 80% of the total fish hydrogen and the tissue-bound hydrogen constitutes 20%. Because of this lower hydrogen concentration in dry tissue, the bioaccumulation factor for protium in fish is approximately 0.93. Subsequent discussions focus on tritium and the degree to which its heavier mass may cause it to have a different bioaccumulation factor from that of protium.

3.3.2 Pathways of Hydrogen Entry into Aquatic Organisms

Tritium enters aquatic systems in the form of tritiated water, that is, HTO, where T represents tritium; H, protium; and O, oxygen. Tritiated water behaves like HOH and mixes rapidly with the tissue water of aquatic organisms. From the tissue water pool tritium can enter hydrogen sites in organic tissue. These hydrogens comprise the tissue-bound hydrogen pool which may be further subdivided into two components termed exchangeable (ET) and nonexchangeable (NET), which have different rates of turnover and different modes of uptake and loss (Figure 3.3.1). The exchangeable component consists of those hydrogens bonded to OH, COOH, NH, SH, and ortho and para positions of phenol groups. These hydrogens undergo a relatively rapid chemical exchange reaction with hydrogens of water molecules in the tissue water. This exchange is not dependent on enzymatic reactions and occurs in metabolically inactive tissues, such as wood, as well as metabolically active tissue. The nonexchangeable component consists predominantly of hydrogens bonded to carbon atoms of organic tissue. Movement of hydrogen into and out of this component is dependent on various enzymatic reductions and oxidations of tissue organics. Various metabolic reactions reduce organic molecules by incorporating hydrogen from tissue water into stable carbon-hydrogen bonds in

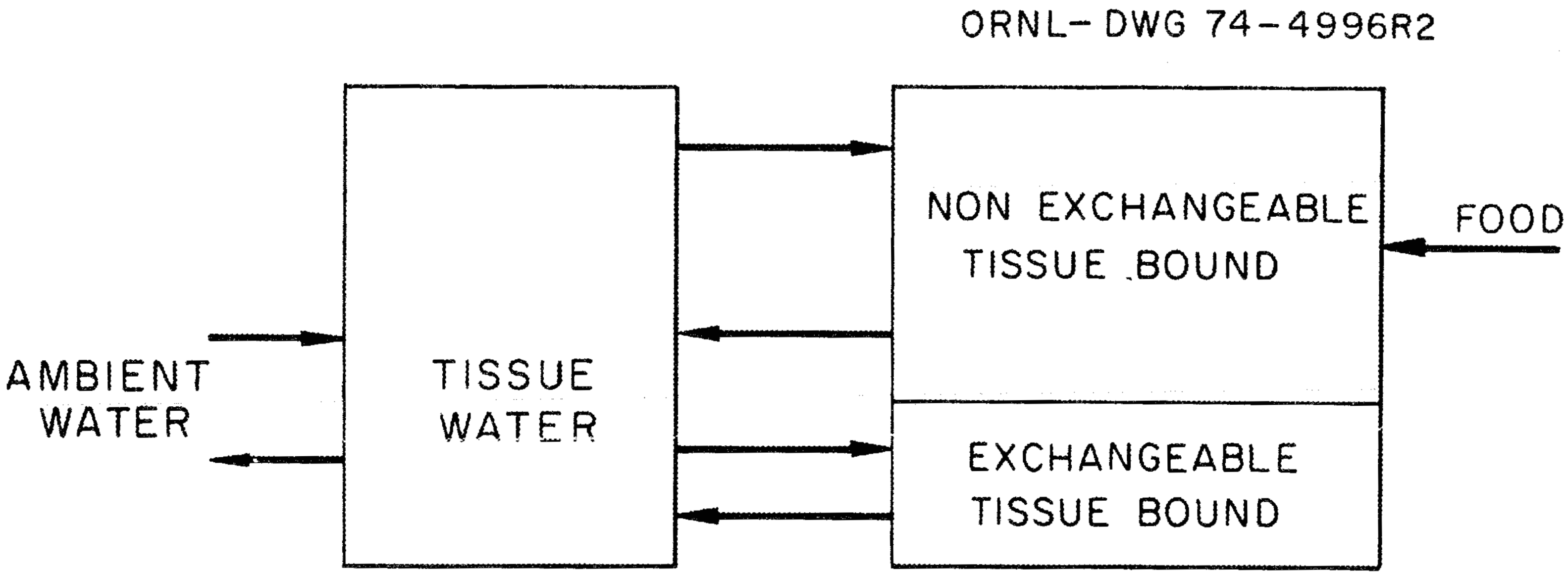

Figure 3.3.1 Pathways of hydrogen entry into the hydrogen components of an aquatic animal.

intermediary products used to synthesize lipids and proteins. Hydrogen is liberated from this nonexchangeable pool by oxidation reactions. These oxidations enzymatically remove nonexchangeable hydrogen from tissue organics and ultimately combine the hydrogens with oxygen to form tissue water. The turnover of this pool is much slower than the exchangeable pool. For a discussion of reactions that incorporate hydrogen from tissue water into the nonexchangeable pool, see Smith and Taylor (1969).

A second pathway for hydrogen entry into the nonexchangeable component, which applies to animals, consists of the ingestion and incorporation of intact food molecules containing nonexchangeable hydrogen. The absence of this pathway in plant uptake and its importance in animals and food chain transfers is discussed in Section 3.3.4.

3.3.3 Compartment Size and Turnover Rates

Tissue-water components constitute from 80 to 93% of aquatic plant weight and averages 75% in aquatic vertebrates. Tritium has been shown to move rapidly from ambient water into tissue water, with bioaccumulation factors approximating unity and biological half-times measured in minutes for small unicellular algae to hours for large aquatic macrophytes and aquatic vertebrates (Elwood, 1973; Harrison and Koranda, 1973; Stewart et al., 1973). Tritium in emergent portions of rooted macrophytes, such as cattails, may fail to reach steady-state with ambient water possibly because of a mixing of water in leaves with less contaminated moisture in the air (Raney and Vaadia, 1965).

Relative sizes of exchangeable and nonexchangeable components have not been determined for aquatic organisms. Data from small mammal experiments, indicating that exchangeable hydrogen constitutes 30% and nonexchangeable hydrogen 70% of the tissue bound component, were used as tissue-bound compartment sizes in constructing Table 3.3.1 and Figure 3.3.1 (Pinson and Langham, 1957; Siri and Evers, 1961).

Data on turnover rates of tritium in the bound components of aquatic organisms are sparse. Elwood (1973) and Patzer (1973), working on goldfish and mosquito fish, respectively, demonstrated 8-day half-times for tritium elimination from the combined tissue-bound component. For aquatic snails, a 62-hr half-time for 70% of the tissue-bound component has been demonstrated (Stewart et al., 1973). A typical study on rats demonstrated half-times of 22 days for 50% of the tissue-bound tritium and 130 days for the other 50% (Thompson and Ballou, 1956). Half-times in humans characteristically show a short tissue-bound component of 30 days and a longer component of 300 days (Pinson and Langham, 1957). In one case Pinson and Langham observed a 2020 day half-time in a chronically exposed human. Although aquatic organisms have demonstrated longer half-times for tissue-bound tritium than for tissue-water tritium, no measurements of elimination rates for exchangeable and nonexchangeable tritium have been reported for aquatic organisms exposed to tritium in their food and water under controlled conditions. Results of experiments cited above indicate some of the tissue-bound components may require a significant fraction of the life of the organism to equilibrate

Table 3.3.1 Component size and source contribution for a hypothetical fish

Components	% Source contribution		% Body hydrogen in each component
	Water	Organic food	
Tissue water			80
Tissue bound			
Exchangeable	100	0	6[b]
Nonexchangeable	60[a]	40[a]	14[b]

[a]Data of Patzer (1973).

[b]Data of Siri and Evers (1961).

with environmental tritium levels. As a result, this nonexchangeable tritium component does not reflect current exposure levels unless long-term steady-state condition has been maintained under constant environmental levels.

3.3.4 Tritium Concentration in Aquatic Food Chains

3.3.4.1 Tissue-Water Tritium

Bioaccumulation factors reported for tissue-water tritium in aquatic organisms range from 0.58 to 1.1 with a mean value less than 1 (Bruner, 1973; Elwood, 1973; Harrison and Koranda, 1973; Stewart et al., 1973; Weinberger, 1953; Weinberger and Porter, 1953). Thus, on the basis of these empirical studies, unity represents a conservative approximation of the tissue-water tritium bioaccumulation factor for aquatic organisms.

3.3.4.2 Tissue-Bound Tritium

Both laboratory and field measurements indicate that the bioaccumulation factor for the total tissue-bound component is less than 1. The tissue-bound tritium concentrations in plants and invertebrates have been summarized from sixteen references by Bruner (1973). In only two of these cases were tissue-bound tritium concentrations higher than ambient water (Cohen and Kneip, 1973; Koranda, 1965). These two measurements were made in areas of pulsed tritium inputs in the vicinity of nuclear power facilities or test sites and probably do not represent steady-state situations. Other field and laboratory measurements on tissue-bound tritium in plants and vertebrates showed tritium bioaccumulation factors below unity (Elwood, 1973; Harrison and Koranda, 1973; Kanazawa et al., 1972; Porter, 1954; Rosenthal and Stewart, 1973; Stewart et al., 1973; Strand et al., 1973; Weinberger, 1953; Weinberger and Porter, 1953). Most studies did not include measurements of exchangeable tritium and nonexchangeable tritium as separate components. Thus, on the basis of empirical data alone, it is not possible to determine a separate bioaccumulation factor for exchangeable tritium and nonexchangeable tritium, or to determine the degree to which the specific activities of certain organic molecules might vary around the mean value for the total tissue-bound component. For this reason subsequent discussions focus on physical and biochemical aspects of the two tissue-bound components and their effects on the bioaccumulation factor for exchangeable tritium and nonexchangeable tritium, as well as individual molecules.

3.3.4.2.1 Exchangeable Tissue-Bound Tritium

Movement of tritium into and out of the exchangeable component involves chemical exchange reactions as discussed in Section 3.3.2. A variety of organic groups common in living tissue have been studied to determine the steady-state concentration obtained via exchange with tritiated water (Weston, 1973). The results, which are summarized in Table 3.3.2, indicate that tritium is not likely to pool in exchangeable

Table 3.3.2 Steady-state concentration in organic groups common in living tissue

Organic group	α^{-1}[a]	Reference
OH Groups		
CH_3COOH	1.0	3.4
CH_3OH	0.98	3.9
Cellulose	1.43	3.14
NH Groups		
NH_3	0.94	3.8
$HCONH_2$	0.97	3.1
Ribonuclease	1.0	3.6
DL-polyalanine	1.2	3.7
SH Groups		
H_2S	0.30	3.16
CH_3SH	0.30	3.11
CH Groups		
$HC\text{-}CCH_3$	0.78	3.15
CH_3SH	0.64	3.30

[a] α^{-1} = (Specific Activity organic group)/(Specific Activity water). Since α^{-1} is based on specific activities it is independent of variations in hydrogen content, and must be multiplied by the percent hydrogen in the tissue and divided by the hydrogen content of water (11%) to produce a "dry weight" bioaccumulation factor for the organic group.

components because in most cases the specific activity* of the molecules is less than that of the water.

A food chain is diagrammed in Figure 3.3.2 with organisms compartmentalized as in Figure 3.3.1 and each consumer utilizing the organism below him as the food source. In both Figure 3.3.1 and 3.3.2 there is no pathway to accommodate the food-chain transfer of exchangeable tritium even though this obviously occurs. The effect of food-chain transfer of exchangeable tritium is unimportant because 1) the exchangeable tritium of each organism has ready access to the same ambient tritium concentrations via exchange with tissue water, and 2) most aquatic organisms eat only a small fraction of their body weight per day thus the turnover of exchangeable tritium appears rapid in comparison.

3.3.4.2.2 Nonexchangeable Tissue-Bound Tritium

At the base of the food chain in Figure 3.3.2, photosynthesis, along with other reduction reactions, incorporates tritium from tissue water into the nonexchangeable component of plants. Data on the total tissue-bound component in plants indicate that there is a 2 to 55% discrimination against incorporation of tritium relative to protium (Bruner, 1973; Harrison and Koranda, 1973; Rosenthal and Stewart, 1973; Weinberger, 1953; Weinberger and Porter, 1953). These determinations, however, did not differentiate between the two tissue-bound components.

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*Specific activity is used here to mean the ratio of radioactive to stable atoms of an element in a sample.

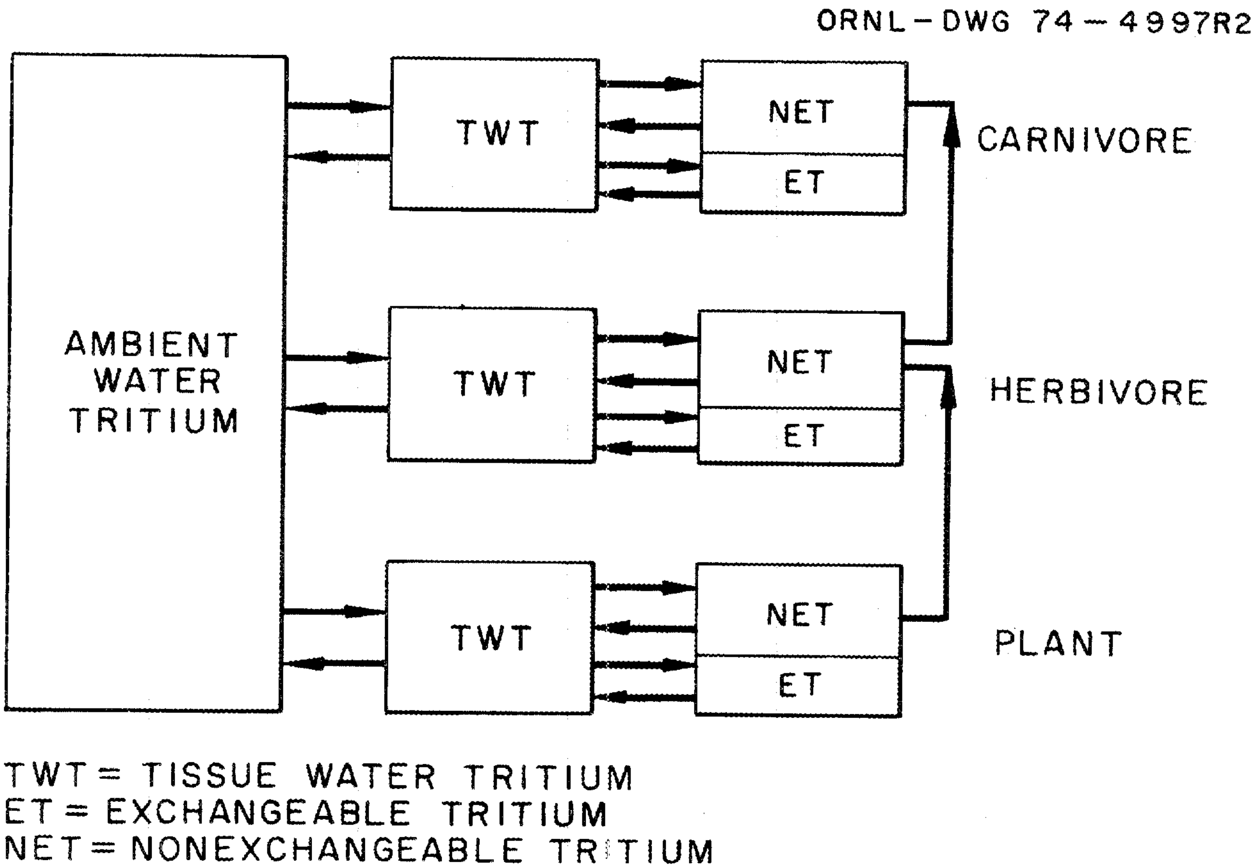

Figure 3.3.2 Three-link food chain diagram with organisms compartmentalized.

Rambeck and Bassham (1973) and Kanazawa et al. (1972) investigated the effects of oxidation and reduction reactions on specific activities of individual organic molecules produced during the Krebs Cycle in algae. After five generations in a closed system the average specific activity of the individual molecules investigated was 30% less than the water to which they were exposed. Only two out of the 14 molecules investigated showed specific activity ratios greater than 1, citrate 1.8 and fumarate 1.2. These results, although limited to 2 species of algae and the Krebs Cycle, indicate that most organic molecules discriminate against tritium incorporation into nonexchangeable sites and that significant increases in the specific activity of certain molecules due to the kinetics of nonexchangeable tritium are not likely. Further studies are needed to verify these results for various series of metabolic reactions in other organisms.

Consumer organisms obtain nonexchangeable tritium by assimilation of intact tritiated food molecules as well as by reduction of tissue organics with tissue-water hydrogens. It can be calculated from Patzer's (1973) data on herbivorous fish that the nonexchangeable component receives approximately 60% of its hydrogen from tissue-water hydrogen and 40% from food. Data are not available on the exact degree to which the specific activity of the nonexchangeable component is affected by food-chain transfers of nonexchangeable tritium in food molecules, or by various oxidations and reductions.

Although empirical data on bioaccumulation factors for nonexchangeable tritium in organisms are wanting, it is possible to establish a maximum bioaccumulation factor for the nonexchangeable component by using the tissue-bound compartment sizes from Table 3.3.1, an average specific activity ratio for the exchangeable component from Table 3.3.2, and bioaccumulation factors determined empirically for the total tissue-bound component in numerous freshwater organisms. Based on Table 3.3.2, 0.7 represents the best estimate of an average bioaccumulation factor for the exchangeable component. Thus, the empirically-derived bioaccumulation factor of less than 1 for the total tissue-bound component, consisting of 30% exchangeable tritium and 70% nonexchangeable tritium, requires that the bioaccumulation factor of the nonexchangeable component be no greater than 1.1.

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3.4 IODINE

3.4.1 Environmental Iodine

The stable iodine content of freshwater ranges from 0.4 to 10.0 ppb depending upon the water source and the environment through which the water travels. Streams originating from glaciers or runoff are low in iodine, less than 1 ppb, while those originating from springs and ground water sources are higher, up to 5 ppb. Livingston (1963) calculated a median river water concentration of 2 ppb. Table 3.4.1 lists some representative stable iodine concentrations from various freshwater, marine, and geologic sources. Iodine in oceans is approximately 50 ppb and apparently originates from the erosion of land masses (Bowen, 1966). The dominant physiocochemical form of iodine and its relative biological availability in freshwater is uncertain (Winchester, 1970).

3.4.2 Iodine Metabolism

The thyroid removes inorganic iodine from the blood and uses it in the production of thyroxine, a hormone necessary for maintaining a variety of metabolic and growth functions. The thyroid releases thyroxine to the blood stream where it combines with large proteins to form protein-bound iodine. Thyroxine is split from the large protein and enters the body tissues where it is degraded. Inorganic iodine produced by the degradation of thyroxine re-enters the blood and is eliminated by urinary excretion. Some evidence indicates that an unspecified portion of this may be reabsorbed by the thyroid (Chavin and Bouwman, 1965).

Table 3.4.1 Iodine concentrations in surface waters, the ocean, and geologic sources

Water Sample	Average Concentration (ppb)	Range (ppb)	Reference
German lake	3.5	2.5 - 4.9	4.18
Canadian lake	1.54	0.49 - 2.9	4.21
Lake Michigan	1.0	----	4.10
Lake Michigan	5.0	----	4.39
Michigan streams	1 or less	----	4.39
Missouri spring	4.7 or less	----	4.12
New Jersey spring	5.2 or less	----	4.12
Swiss rivers	0.56	0.4 - 1.3	4.15
Panama rivers	1 or less	1.0 - 0.6	4.14
River Average	2.0	< 1.0 - 10	4.28
Oceans	50	----	4.47
Soils	----	600 - 6000	4.47
Rock			
Igneous	500	----	4.5
Limestone	1200	----	4.5
Shale	2200	----	4.5
Coal	6000	----	4.5

Although there are numerous studies on the physiological role of iodine, few of these investigations provide data on elimination rates and turnover times for iodine in aquatic organisms. The few studies that were found are summarized in Table 3.4.2. The rapid component (C-1) is taken to represent plasma clearance through urinary excretion, while the slower component (C-2) is attributed to the thyroid.

3.4.3 Review of Iodine Bioaccumulation Factors

3.4.3.1 Stable Iodine Bioaccumulation Factors for Fish Tissue

Only the thyroid tissue and ovaries (eggs) show great iodine bioaccumulation. No studies were found that provided data allowing direct calculation of bioaccumulation factors for the thyroid tissue because thyroid stable iodine measurements for freshwater fish were not expressed as concentrations. Data were available on thyroid iodine concentrations in some marine species and these marine data are cited to corroborate our indirectly estimated freshwater thyroid bioaccumulation factors. Muscle bioaccumulation factors of freshwater and marine fish are very similar despite the much higher iodine content of oceans (Tables 3.4.3 and 3.4.4). Organic and inorganic iodine show the same percent distribution in the serum of estuarine flounder and freshwater whitefish (Hickman, 1962). The similar iodine bioaccumulation factors for muscle and the similar iodine metabolism in fish from freshwater and marine habitats provide some justification for comparing bioaccumulation factors of marine and freshwater species.

References on stable iodine in fish ovaries did not report comparable iodine water concentrations, thus, ovary bioaccumulation factors

Table 3.4.2 Elimination rates for iodine-131 in aquatic vertebrates

Species	Biological Half-time for Iodine (days)	Administered Activity (%)	Elimination Coefficient (day^{-1})	Reference
Salmo gairdneri				
C-1	2.0	----	.347	4.13
C-2	8.6	----	.086	4.13
Salmo gairdneri				
C-1	1.7	----	.408	4.22,4.50
C-2	---	----	----	-----
Cyprinus carpio				
C-1	---	----	----	-----
C-2	14.0	----	.050	4.36
Carassiuss auratus				
C-1	0.8	----	.866	4.36
C-2	14.0	----	.050	4.36
Perch (sp.)				
C-1	1.5	----	.460	4.42
C-2	---	----	----	----
Micropogon undulatus				
C-1	0.25	59	2.770	4.2
C-2	2.25	40	.310	4.2
C-3	24.0	1	.028	4.2
Rana pipiens				
C-1	---	----	----	----
C-2	56.0	----	.012	4.50
Hyla versicolor				
C-1	5.0	----	.140	4.25
C-2	---	----	----	----
Taricha granulosa				
C-1	2.0	74	.350	4.50
C-2	210.0	26	.003	4.50

Table 3.4.3 Stable iodine in freshwater fish muscle

Species	Muscle Concentration (μg/kg)	Water Concentration (μg/liter)	Stable Iodine BF[a]	Location	Reference
Salmo fario	36	----	36	Switzerland	4.16[b]
Salmo fario	48	----	48	Germany	4.4[b]
Salmo fario	24	----	24	New Zealand	4.20[b]
Salmo fario	50	----	50	New Zealand	4.20[b]
Salvelinus namaycush	10	----	10	Lake Erie	4.45[b]
Luciopimelodus pati	30	----	30		4.32[b]
Pimelodus albidus	40	----	40		4.32[b]
Lepomis incisor	40	----	40	Mississippi River	4.45[b]
Microperus salmoides	50	----	50	Potomac River	4.45[b]
Pomoxis annularis	10	----	10	Mississippi River	4.45[b]
Perca flavescens	20	----	20	Potomac River	4.45[b]
Perca fluviatilis	20	----	20	Switzerland	4.16[b]
Freshwater species in general	40	----	40		4.33[b]
Carp	17	----	17		4.6
Micropterus salmoides	30	----	30		4.6
Salvelinus namaycush	31	----	31		4.6
River perch	40	----	40		4.6
Salmo gairdneri	15	1	15	Black River, Michigan	4.39
Salmo gairdneri	40	----	40	Pacific Coast Stream	4.26
Oncorhynchus kisutch	120	1	120	Lake Michigan	4.11[c]
Perca flavescens	120	1	120	Lake Michigan	4.11[c]
Salvelinus namaycush	170	1	170	Lake Michigan	4.11[c]
Salmo trutta	110	1	110	Lake Michigan	4.11[c]
Coregonus clupeaformis	140	1	140	Lake Michigan	4.11[c]
Mean	50 ± 45				

[a]Tissue concentration reported without comparable water values was divided by 1 μg/liter to approximate a conservative bioaccumulation factor.

[b]Cited in Vinogradov (1953).

[c]Tissue concentrations represent averages calculated from Copeland's raw data that clearly specified that the tissue analyzed was muscle.

Table 3.4.4 Stable iodine in anadromous and marine fish muscle

Species	Muscle Concentration (μg/100 g)	Water Concentration (μg/liter)	Stable Iodine BF	Remarks	Reference
Sea trout	320	-----		Anadromous	4.6
Salmo salar	341	-----		Anadromous	4.6
Salmo gairdneri (sea run)	380	-----		Anadromous	4.26
Oncorhynchus	296	-----	53	Anadromous	4.26
Mackerel	371	-----	74	Marine	4.6
Herring	520	-----	104	Marine	4.6
Roccus americanus	742	-----	148	Marine	4.6
Cod	1463	-----	293	Marine	4.6
Hippoglossus	520	-----	104	Marine	4.6
Melanogrammus aeglefinus	318	-----	64	Marine	4.6
Clupea harengus	600	-----	120	Marine	4.31[a]
Melanogrammus aeglefinus	513	-----	103	Marine	4.34[a]
		Mean BF	118 ± 72		

[a]Reference cited in Vinogradov (1953).

could not be directly calculated from the studies. However, we were able to indirectly estimate an average ovary bioaccumulation factor for stable iodine from studies which reported both ovary and muscle iodine contents in the fish collected. From these data an ovary:muscle ratio was calculated and multiplied times the average muscle bioaccumulation factor to estimate an average ovary bioaccumulation factor for stable iodine.

Data on stable iodine in fish muscle were adequate to determine a reliable bioaccumulation factor. Due to the limited data on iodine in fish thyroid and ovaries, and the manner in which these data were reported, our estimates for thyroid and ovary bioaccumulation factors represent approximations.

3.4.3.1.1 Stable Iodine Bioaccumulation Factors for Fish Muscle

Stable iodine bioaccumulation factors for fish muscle from various species of freshwater fish are listed in Table 3.4.3. Bioaccumulation factors from a variety of investigators varied from 10 to 50 while determinations by Copeland et al. (1973) ranged from 120 to 170. By using a 5 ppb Lake Michigan iodine concentration, as reported by Robertson and Chaney (1953), with Copeland's tissue iodine concentrations, a range of bioaccumulation factors can be calculated for Copeland's samples that agrees well with ranges reported by other investigators. However, Winchester (1970), reported 1 ppb or less as the iodine concentration in Lake Michigan, thus, indicating that Copeland's water concentrations are correct and that higher tissue iodine concentration and higher bioaccumulation factors for Lake Michigan fish cannot be explained by an underestimate of iodine concentration in water. At this time it is the opinion of the authors that although Copeland's data are higher than the others, we must accept their bioaccumulation factors of 120 to 170 for stable iodine in freshwater fish muscle until other data clarify the issue. From data in Table 3.4.3 we calculate an average bioaccumulation factor of 52 for stable iodine in fish muscle with a range from 10 to 170.

Based on clear decreases in iodine tissue concentrations from marine to anadromous to freshwater species, it does not appear that homeostatic control is a factor in controlling muscle iodine levels in fish (Table 3.4.4). Anadromous fish sampled during their spawning run show stable iodine tissue concentrations higher than strictly freshwater species (Table 3.4.4). This is evidently due to their recent exposure to higher iodine concentrations in oceans. Fontain et al. (1948), found 1280 μg/g of iodine in the serum of Atlantic salmon at the beginning of their migration spawn. At the end of migration, serum iodine had dropped to 280 μg/g. It was not determined if iodine in other tissues showed a similar decline. The drop in serum iodine could be due to lower freshwater iodine concentrations and a lack of homeostatic control, or to a combination of physiological factors.

3.4.3.1.2 Stable Iodine Bioaccumulation Factors for Fish Thyroids and Ovaries

In rainbow trout and most teleosts the thyroid is not assembled into a single gland but, instead, is scattered throughout the lower jaw. In some cases thyroid tissue may occur in the eyes and kidneys as well (Baker et al., 1955). The removal of diffuse thyroidal tissue without portions of nonthyroidal tissue is difficult. As a result, iodine concentrations are either not expressed on a unit weight basis or qualified to the extent that the tissue may contain nonthyroid material (Short et al., 1969).

Robertson and Chaney (1953) analyzed sections of carefully dissected Lake Michigan rainbow trout tissue excised from the floor of the mouth from the first to the fourth gill arch. The iodine content was expressed as total micrograms of iodine in the total "thyroid mass" (Table 3.4.5.). The weights of these samples were not reported. No determinations were made of the stable thyroidal iodine content, expressed per unit of exclusively thyroid tissue of freshwater fish. Using 0.003 g/100 g of fish as an approximation of the mass of the rainbow trout thyroid (Spector, 1956), we calculated an approximate thyroid iodine concentration of 269 μg/g by using the average weights of the trout used in Robertson and Chaney's (1953) study. Dividing this thyroid concentration by Copeland's Lake Michigan iodine concentration of 1 ppb gives an estimate of 270,000 for the trout thyroid bioaccumulation factor. To provide some verification of this thyroid bioaccumulation factor, an average thyroid bioaccumulation factor for marine fish of 770,000 was calculated using data compiled from Vinogradov (1953) in Table 3.4.6. Considering the approximate nature of our estimates, these two values are fairly close and tend to support the freshwater thyroid bioaccumulation factor.

Hickman (1962) measured inorganic iodine in the serum and thyroids of estuarine flounder and found that the serum levels of iodine increased on a seasonal cycle as the salinity of the water increased. However, in the case of thyroidal iodine, only inorganic iodine showed a clear

Table 3.4.5 Total iodine present in the thyroid and eggs of five rainbow and California steelhead trout (in micrograms)[a]

Rainbow Trout		Steelhead Trout[b]	
Thyroid	Eggs	Thyroid	Eggs
3.5	38.6	367	1170
0.5	18.6	755	2323
12.3	41.3	377	1814
11.9	70.0	---	1749
16.0	55.0	---	1065
10.0[c]	44.0[c]	500[c]	1624[c]

[a]Robertson and Chaney (1953).

[b]Steelhead trout are anadromous rainbow trout (Salmo gardneri). They were secured from the mouths of California rivers up to 150 miles upstream.

[c]Mean values.

Table 3.4.6 Thyroid stable iodine bioaccumulation factors for marine fish

Species	Thyroid Iodine (ppm)	Thyroid BF[a]	Reference[b]
Gadus aeglefinus	2,800	56,000	4.9,4.30
Gadus morrhua	11,600	232,000	4.9,4.31
Salmo eriox	173,600	3,500,000	4.30,4.31
Salmo salar	9,600	192,000	4.9,4.31
Scomber scombrus	32,200	644,000	4.29,4.30
Sebastes marinus	46,200	924,000	4.29,4.31
Pleuronectes platessa	7,800	156,000	4.9,4.31
Molva vulgaris	23,800	476,000	4.9
	Mean	772,500	

[a]Calculated using 50 ppb as marine water iodine concentration (Table 3.4.1).

[b]Cited in Vinogradov (1953).

response to environmental iodine fluctuations. The protein-bound iodine, which constitutes 95% of thyroidal iodine, varied much less and with no apparent pattern. Protein-bound iodine was, in fact, highest in September when environmental iodine was at a low point. Thyroid iodine concentration appears to fluctuate little in response to environmental iodine. Whether this is due to homeostatic control, slower turnover rates, or other factors is uncertain.

From Table 3.4.5 it is apparent that the ovaries (eggs) contain most of the iodine present in female rainbow trout. Due to the manner in which the data were reported and to uncertainties concerning analytical procedures, we calculated an ovary:muscle concentration ratio for stable iodine (Table 3.4.7) and then multiplied this ratio by the average muscle bioaccumulation factor from Table 3.4.3 to estimate a stable iodine bioaccumulation factor of 800 for ovaries.

3.4.3.2 Iodine-131 Bioaccumulation Factors for Fish Tissue

3.4.3.2.1 Iodine-131 Bioaccumulation Factors for Fish Muscle

The few experiments reporting iodine-131 levels in fish muscle are listed in Table 3.4.8. Experiments by Short et al. (1969) indicate that the food web is the principal mode of uptake. One study by Hunn and Reineke (1964) indicates that iodine can be accumulated directly from the water. In Short's study, an aquarium and a lake were both tagged with iodine-131. After 29 days fish in the lake showed a muscle bioaccumulation factor of 100, while those in the aquaria, without their natural food sources, showed a bioaccumulation factor of 1.7. It is possible, although experimental procedures were

Table 3.4.7 Stable iodine bioaccumulation factor for fish ovaries (eggs) derived from relative concentrations of stable iodine in muscle and ovaries

Species	Ovary/muscle Ratio	Ovary[a] BF	Reference
Salmo gardneri	15	750	4.39
Salmo gardneri	14	700	4.39
Salmo fario	29	1450	4.16
Perca fluviatalis	6	300	4.16
Mean Ovary BF		800	

[a] Ovary bioaccumulation factor = Ovary/muscle ratio times average muscle bioaccumulation factor from Table 3.4.3.

Table 3.4.8 Iodine-131 bioaccumulation factors for muscle in freshwater animals

Species	Iodine-131 BF	Location	Reference
Carp _Cyprinus carassius_ (whole fish)	25	Lake	4.27
Prochilodus	0.5	Aquaria	4.3
Pimelodus	0.8	Aquaria	4.3
Cichlasoma	0.5	Aquaria	4.3
Miscellaneous aquatic 17 species leech gastropods crustaceans insect larvae carp tadpoles	20 (mean)	Aquaria	4.43
Carp _Cyprinus carpio_	12	Aquaria	4.44
Salmo gairdneri	100	Lake	4.40
Mean Fish Muscle BF 39[a]			

[a]See Sec. 3.4.3.2.1.

not described, that the low bioaccumulation factors reported by Beninson (1966) for fish in aquaria studies were due to the absence of a contaminated food source. The low bioaccumulation factors do not appear reliable in view of Short's study and the much higher stable iodine bioaccumulation factors reported in Table 3.4.8. An average iodine-131 bioaccumulation factor of 39 for fish muscle is calculated for fish muscle from Table 3.4.8 (Beninson's data omitted).

Because of the 8-day radioactive half-life of iodine-131, the turnover rate of iodine in a specific tissue and the length of the primary contamination pathway control the degree to which bioaccumulation factors for iodine-131 are less than bioaccumulation factors for stable iodine. Iodine-131 absorbed directly from the water will have less time to decay than iodine-131 obtained from food. Section 2.4 and Equation (2.4.9) describe a method for calculating bioaccumulation factors for radioactive isotopes based on their stable element bioaccumulation factors. Using Equation (2.4.9), the average stable element bioaccumulation factor for fish muscle (Table 3.4.3), and an average fast component elimination coefficient of 0.43 (Table 3.4.2), we calculate an iodine-131 bioaccumulation factor of 43. Based on the similarity between this calculated bioaccumulation factor and the average bioaccumulation factor determined from empirical studies, we recommend an average bioaccumulation factor of 40 for iodine-131 in fish muscle.

3.4.3.2.2 Iodine-131 Bioaccumulation Factors for Fish Thyroids

Short et al. (1969) removed the lower terminus of the gill arch, including the dorsal aorta and surrounding muscle, from lake trout removed from an iodine-131 tagged lake after 29 days of exposure. An iodine-131 bioaccumulation factor of 12,000 was calculated for these "thyroid" samples which contained unspecified amounts of other tissue. This value appears low when compared to the thyroid bioaccumulation factor of 270,000 for stable iodine (Sec. 3.4.3.1.2). Since no freshwater iodine-131 bioaccumulation factors have been determined for thyroids free from attached nonthyroidal tissue, we recommend an iodine-131 thyroid bioaccumulation factor of 110,000. An 11-day elimination coefficient for the long component in trout (Table 3.4.2) and an average stable iodine bioaccumulation factor for thyroid tissue of 270,000 were used in calculating the recommended thyroid bioaccumulation factor. Further research is needed to clarify this issue.

3.4.3.3 Stable Iodine Bioaccumulation Factors for Plants

The only stable iodine determination found for freshwater plants was an average bioaccumulation factor of 800, calculated by Copeland and Ayers (1970) for samples of phytoplankton (green and bluegreen algae) taken from Lake Michigan.

3.4.3.4 Iodine-131 Bioaccumulation Factors for Plants

Bioaccumulation factors for rooted and floating macrophytes are combined in Table 3.4.9 since they are within the same range. Iodine-131 bioaccumulation factors for freshwater algae from one study averaged 255 (Short et al., 1969).

3.4.3.5 Stable Iodine Bioaccumulation Factors for Invertebrates

Table 3.4.10 lists stable iodine bioaccumulation factors for a small number of aquatic invertebrates. In insects, and probably

Table 3.4.9 Iodine-131 bioaccumulation factors for freshwater plants

Species	Iodine-131 BF	Remarks	Reference
Macrophytes			
Bacopa	130	Tank	4.3
Cabomba	178	Tank	4.3
Elodea	145	Tank	4.3
Ceratophylum	178	Tank	4.3
Echinodorus	162	Tank	4.3
Ceratophyllum	95	Tank	4.44
Myriophyllum	100	Tank	4.44
Utricularia	209	Tank	4.44
Lemna minor	71	Tank	4.44
L. trisulca	154	Tank	4.44
Potamogeton	26	Tank	4.44
Elodea canadensis	134	Tank	4.44
Stratiotes	60	Tank	4.44
Hydrocharis	165	Tank	4.44
Carex	101	Tank	4.44
Sphagnum	90	Lake	4.27
Nuphar	60	Lake	4.27
Mean of 32 Plant Species	93	Tank	4.44
Mean Macrophyte BF	119		
Algae			
Nitella	380	Lake	4.40
Spirogyra and *Oedogonium*	130	Lake	4.40
Mean Algae BF	255		

Table 3.4.10 Stable iodine bioaccumulation factors for invertebrates

Organism	Concentration in Tissue (ppb)	Concentration in Water (ppb)	Stable Iodine BF	Tissue	Reference
Crustacea					
Shrimp (freshwater)	60[a]	0.6[b]	10	----	4.14
Snail	150	0.6[b]	250	----	4.14
Shrimp (*Mysis*)	565[c]	1	565	----	4.10
Amphipod (*Pontoporeia*)	390[c]	1	390	----	4.10
Benthos (mostly crustacean)	500	1	500	----	4.10
Mollusca					
Clam (*Dreissensia*)	----	----	1000	Shell	4.49
	Mean Crustacean BF		343		

[a]A factor of 10 was used to convert dry weight values to wet weight values reported here.

[b]Water concentrations represent drinking water measurements of a local Panama city.

[c]Calculated from Copeland's (1970) raw data using samples of benthos said to consist of 100% *Mysis* or *Pontoporeia*.

crustacea as well, most of the iodine appears to be concentrated in the integuement. Iodine-131 studies indicate that little iodine is lost from insects except when the integuement is discarded (Odum and Golley, 1963; Van Hook and Crossley, 1969). The average bioaccumulation factor for crustacea, the only group of invertebrates for which stable iodine values of soft tissues or whole animals were found, is 343.

3.4.3.6 Iodine-131 Bioaccumulation Factors for Invertebrates

Table 3.4.11 lists iodine-131 bioaccumulation factors for various invertebrates. An average bioaccumulation factor value is recorded for aquatic insects and molluscs. The difference in the lake bioaccumulation factor of 140 and the aquaria bioaccumulation factor of 600 reported for the amphipod is attributed to incomplete mixing of lake waters (Short et al., 1969).

Table 3.4.11 Iodine-131 bioaccumulation factors for invertebrates

Organism	Iodine-131 BF	Tissue	Remarks	Reference
Crustacea:				
Crayfish (*Pacifastacus*)	10	Muscle	Lake	4.40
	62	Soft parts	Lake	4.40
	240	Carapace	Lake	4.40
Amphipod	600	Whole	Tank	4.40
	140	Whole	Lake	4.40
Aegla			Tank	4.3
Littoral plankton	530		Lake	4.40
Limnetic plankton	500		Lake	4.40
Aquatic Insects:				
Dragon fly larvae (*Leucorrhinia*)	151	?	Tank	4.48
May fly larvae (*Cloeon*)	690	?	Tank	4.48
Caddis fly larvae (*Glyphotaelius*)	163	?	Tank	4.48
Drone fly larvae (*Eristalis*)	600	?	Tank	4.48
Mean BF for Aquatic Insects	400			
Worm:				
Herpobdella	10	?		4.38,4.44
Sponge:				
Spongilla	200	?	Lake	4.27
Molluscs:				
Diplodon	11	?	Tank	4.3
Ampullaria	23	?	Tank	4.3
Planorbis	134	?	Tank	4.3
Margaritifera	10	Muscle	Tank	4.40
	53	Soft parts	Lake	4.40
Limnaea	23	?	Tank	4.38,4.44
Radix	14	?	Tank	4.38,4.44
Anisus	53	?	Tank	4.38,4.44
Planorbis	70	?	Tank	4.38,4.44
Dreissensia	140	Gills	Tank	4.19
	70	Mantle	Tank	4.19
	40	Viscera	Tank	4.19
	20	Byssus	Tank	4.19
	400	Shell	Tank	4.19
Pond snail	20	?	Tank	4.43
Mean BF for Molluscs	50			

References for Section 3.4

The numbers in the left-hand margin correspond to the reference numbers used in the tables of section 3.4.

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3.5 MANGANESE

3.5.1 Environmental Manganese

Manganese is found in six major forms in natural waters (Gibbs, 1973; Wangersky, 1963). In the suspended phase, manganese may be (1) a part of the crystalline structure of suspended material, (2) precipitated onto the suspended material probably as MnO_2, (3) adsorbed onto solids by ion exchange, and (4) incorporated into solid biological materials. In the dissolved phase manganese may occur as one or more (5) dissolved ionic species, e.g., Mn^{++} or inorganic associations, and as (6) complexes with organic molecules in solution.

Gibbs (1973) found that 10% of the manganese in the Yukon River and 17% of the manganese in the Amazon River were in the dissolved phase; less than 1% of the manganese in either river was on ion exchange sites. In the Yukon River 46% of the manganese was precipitated as metallic coatings, 37% was in the crystalline structure and 7% was in biological material. In the Amazon River 50% of the manganese was on the precipitated metallic coatings, 27% was in the crystalline structure and 5% was in biological material.

In surface waters of five Alaskan lakes and Par Pond (South Carolina), 51 to 94% of the manganese was in the particulate phase (Marshall and LeRoy, 1973). In lakes, most of the manganese in the particulate phase is probably MnO_2 precipitates (Marshall and LeRoy, 1973). Dissolved manganese may be largely bound to organics since Mn^{++} will most likely oxidize to MnO_2 on surfaces under aerobic conditions (Gibbs, 1973; Wangersky, 1963). Physicochemical form of manganese affects biological availability (Gibbs, 1973): (1) manganese in the crystalline structure of sediments is not available for uptake; (2) manganese in organic materials and manganese precipitates may be somewhat available; and (3) manganese in solution or at in exchange sites may be very available. Since the proportions of these different physicochemical forms may vary considerably from site to site, the availability of manganese to organisms would be expected to vary from site to site. This picture is further complicated by strong seasonal cycles of manganese concentration in these forms (Marshall and LeRoy, 1973).

3.5.2 Manganese Metabolism

It was noted in Section 2.3 that the availability of a homeostatically-controlled element in different physicochemical forms had no effect on the concentration of that element in the organism. The bioaccumulation factor of a homeostatically-controlled stable element and the bioaccumulation factor of the radionuclide are given by Equation (2.1.4). Fortunately, the considerable site to site variability in manganese availability is not an issue for many organisms since manganese is homeostatically controlled in vertebrates and in some invertebrates. In a literature review Schroeder, Balassa, and Tipton (1966) concluded that: "An efficient homeostatic mechanism for manganese appears to operate in all vertebrate and most invertebrate animals." Bryan and Ward (1965) suggested that the marine lobster *Homerus* has a regulatory mechanism for manganese and further suggested that a similar mechanism should exist in molluscs and fishes. Cavallero and Merlini (1967) found only small site to site variations in manganese concentrations among aquatic vertebrates. Bortoli et al. (1969) found no differences in manganese concentrations among fishes from four different Italian lakes and Lentsch et al. (1973) detected little or no variation in manganese concentrations in fishes from the Hudson River, despite great changes in manganese concentration in the water.

3.5.3 Review of Manganese Bioaccumulation Factors

3.5.3.1 Manganese Bioaccumulation Factors for Fishes

As indicated in Section 3.5.2 the evidence for homeostatic control of manganese concentration in fishes is strong. Manganese concentrations in fishes and water and the bioaccumulation factors for manganese in fishes are tabulated in Table 3.5.1. Table 3.5.2 gives the means and coefficients of variation for the data on manganese concentrations in water and fishes given in Table 3.5.1. The small coefficients of variation for fishes in Table 3.5.2 and the large range of bioaccumulation factors in Table 3.5.1 support the hypothesis of homeostatic control of manganese. It is also interesting to note that manganese concentrations in estuarine and marine fishes are about the same as those in freshwater fishes. The average muscle concentration of manganese in Pomatomus saltatrix (bluefish), a marine epipelagic carnivore, is 0.22 ppm wet weight and the average concentration of manganese in Antimora rostrata, a bathy-demersal fish, is 0.21 ppm (Cross et al., 1973). The concentration of manganese in the ocean is about 0.002 ppm. Manganese concentrations in five species of estuarine fishes was 4.7 ppm on a whole fish basis (Cross and Brooks, 1973). These values for marine and estuarine fishes fall within one standard deviation of the mean for freshwater fishes (Table 3.5.2).

Table 3.5.1 Bioaccumulation factors for manganese in freshwater fishes

Species	Tissue	Feeding habits	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Alburnus alborella (bleak)	whole	plankton feeder	20.6±0.7(6)	9.0-100[a]	2300-210	Five irrigated sites[b]	5.4
Lepomis gibbosus	whole		18.1±0.9(5)	9.0-100[a]	2000-180	Five irrigated sites[b]	5.4
Alburnus alborella	whole	plankton feeder	12.7±0.8(11)	9.0-100[a]	1400-130	Five irrigated sites[c]	5.4
Lepomis gibbosus	whole		8.1±0.3(21)	9.0-100[a]	900-80	Five irrigated sites[c]	5.4
Cobitis taenia	whole	bottom feeder	5.2±0.4(4)	9.0-100[a]	580-50	Five irrigated sites[c]	5.4
Cyprinus carpio (carp)	whole	bottom feeder	4.8±0.7(4)	9.0-100[a]	530-50	Five irrigated sites[c]	5.4
Tinca tinca (tench)	whole	scavenger	3.9±0.3(6)	9.0-100[a]	430-40	Five irrigated sites[c]	5.4
Average of *Perca fluviatilis* (perch), *Scardinius erythrophthalmus* (rudd), and *Lepomis gibbosus*[c]	whole		7.6	42[a]	180	Lake Varese, Italy	5.2
	whole		8.6	23[a]	374	Lake Camabbio	5.2
	whole		3.6	17[a]	211	Lake Maggiore	5.2
	whole		5.2	8[a]	650	Lake Monate	5.2
Alosa pseudoharengus (alewife)	whole	zooplankton feeder	4.6±0.3(19)	0.92±0.08(53)[d]	5000	Lake Michigan	5.6
	flesh		0.88±0.11(6)	0.92±0.08(53)[d]	960	Lake Michigan	5.6
Coregonus macrophthalmus (whitefish, *Bondella*)	flesh	plankton and benthic feeder	0.25±0.03	17±2 (4)[a]	15	Lake Maggiore	5.17
	bone		25	17±2 (4)[a]	1500	Lake Maggiore	5.17
Alburnus alborella (bleak)	flesh	plankton feeder	0.36±0.04	17±2 (4)[a]	21	Lake Maggiore	5.17
	bone		41	17±2 (4)[a]	2400	Lake Maggiore	5.17

Table 3.5.1 (continued)

Species	Tissue	Feeding habits	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Scardinius erythrophthalmus (rudd)	flesh	omnivorous	0.35±0.4	17±2 (4)[a]	21	Lake Maggiore	5.17
	bone		39	17±2 (4)[a]	2300	Lake Maggiore	5.17
Notropis hudsonius (spottail shiner)	whole	unknown	3.3±0.3(15)	0.92±0.08(53)[d]	3600	Lake Michigan	5.6
	flesh	benthic	0.51±0.06(3)	0.92±0.08(53)[d]	550	Lake Michigan	5.6
Coregonus sp. (chub)	flesh	benthic	0.34±0.04(5)	0.92±0.08(53)[d]	370	Lake Michigan	5.6
Coregonus artedii (cisco)	flesh	zooplankton and benthic feeder	0.29±0.05(6)	0.92±0.08(53)[d]	315	Lake Michigan	5.6
Lepomis sp. (sunfish)	flesh		0.5±0.1(3)	0.92±0.08(53)[d]	500	Lake Michigan	5.6
Prosobium cylindraceum (rand whitefish)	flesh	benthic	0.25±0.03(13)	0.92±0.08(53)[d]	270	Lake Michigan	5.6
Osmerus mordax (smelt)	whole	plankton and	3.9±0.4(16)	0.92±0.08(53)[d]	4200	Lake Michigan	5.6
	flesh	benthic feeder	0.27±0.05(9)	0.92±0.08(53)[d]	290	Lake Michigan	5.6
Perca flavescens (yellow perch)	flesh	partially pisciverous	0.37±0.03(24)	0.92±0.08(53)[d]	400	Lake Michigan	5.6
Oncorhynchus kisutch (coho salmon)	flesh	pisciverous	0.23±0.03(20)	0.92±0.08(53)[d]	250	Lake Michigan	5.6
Salvelinus namaycush (lake trout)	flesh	pisciverous	0.21±0.02(24)	0.92±0.08(53)[d]	230	Lake Michigan	5.6
Salmo trutta (brown trout)	flesh	pisciverous	0.23±0.02(11)	0.92±0.08(53)[d]	250	Lake Michigan	5.6

Table 3.5.1 (continued)

Species	Tissue	Feeding habits	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Salmo gairdneri (steelhead)	flesh	piscivorous	0.23±0.02(6)	0.92±0.08(53)	250	Lake Michigan	5.6
Micropterus dolomieui (smallmouth bass)	flesh	piscivorous	0.25±0.02(3)	0.92±0.08(53)	270	Lake Michigan	5.6

[a]Unfiltered value.

[b]Measurements on fish made in July - August.

[c]Measurements on fish made in September - November.

[d]Filtered value.

Table 3.5.2 Concentration and coefficients of variation of manganese concentration (ppm) in water and freshwater fishes (fresh weight).

	Water	Flesh	Whole Fishes
Arithmetic Mean	0.028 ± 0.034	$0.34 \pm .17$	7.9 ± 5.5
Coefficient of Variation	119%	50%	70%
N	7	16	14
Geometric Mean	$0.014 \times\!\!/\!\!\div 4.43$	$0.32 \times\!\!/\!\!\div 1.45$	$6.7 \times\!\!/\!\!\div 1.84$
Coefficient of Variation	343%	45%	84%
N	7	16	14

Using the geometric means of stable manganese concentration in freshwater fishes in Table 3.5.2 and Equation (2.1.3),

$$BF(Mn)_i = 6.7/[Mn]_w \qquad (3.5.1)$$

for whole fish and

$$BF(Mn)_i = 0.32/[Mn]_w \qquad (3.5.2)$$

for fish flesh, where $[Mn]_w$ has units of ppm. These relations and data from Table 3.5.1 are plotted in Figure 3.5.1. Note that it is inconsequential whether filtered or unfiltered values are plotted on the graphs. However, for predictive purposes, an unfiltered value for the manganese concentration in water should be used (Section 2.5.3) unless the user is willing to accept the gross overestimates obtained by use of filtered concentrations.

We hypothesize that the differences in manganese concentration among species of fishes are owing to physiological demands rather than differences in concentration of manganese in the fishes' food. However, bottom feeders have lowest concentrations, plankton feeders have the highest, and piscivores have intermediate concentrations. These differences merit further research.

3.5.3.2 Manganese Bioaccumulation Factors for Plants

Table 3.5.3 summarizes the few data available on manganese bioaccumulation factors for freshwater plants. Harvey (1969, 1970), using a continuous flow, laboratory system to control temperature, showed that nonlethal temperatures have no effect on the manganese bioaccumulation factors for algae. On the basis of data in Table 3.5.2, we recommend

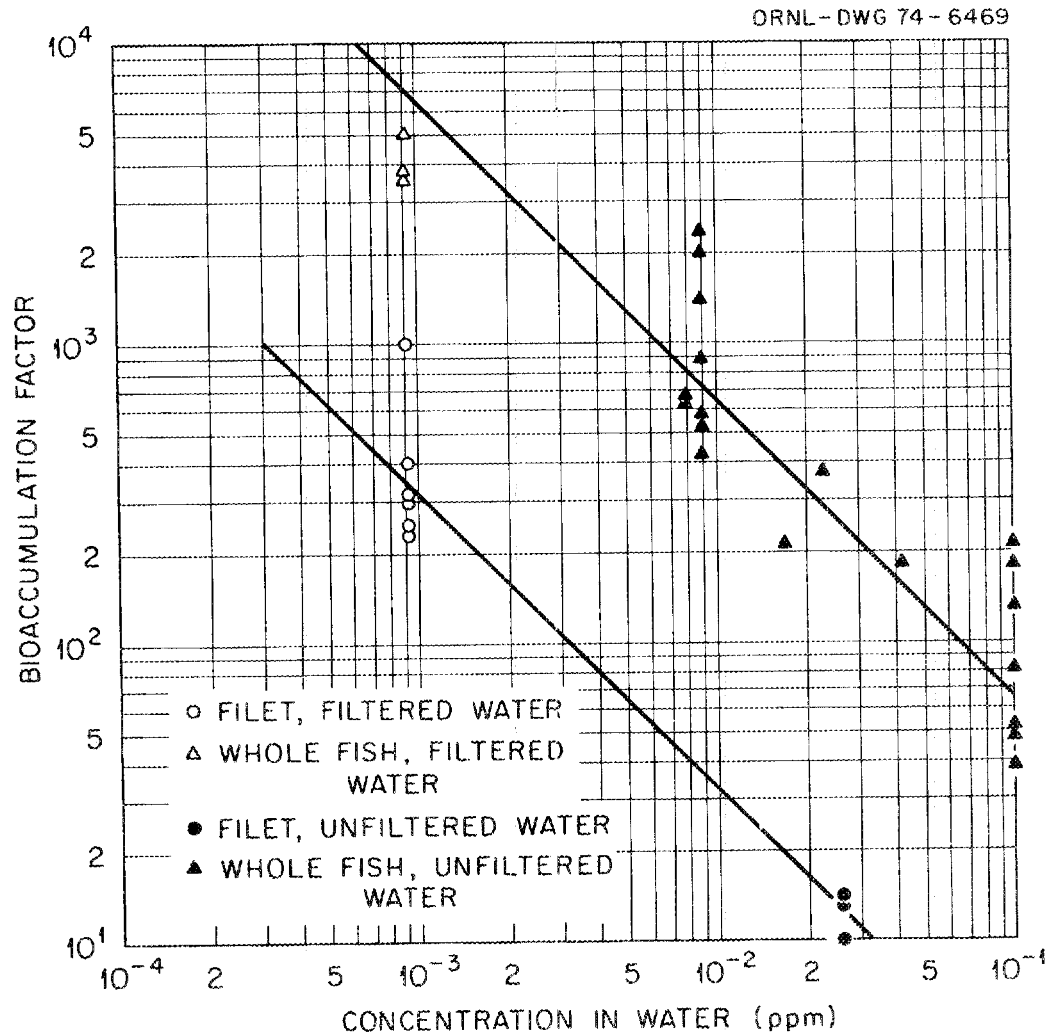

Figure 3.5.1 Bioaccumulation factors for manganese in freshwater fishes as a function of manganese concentration in water.

Table 3.5.3 Bioaccumulation factors for manganese in aquatic plants

Species	Description	Stable element concentration in plant (ppm)	Stable element concentration in water (ppm) Filtered	Unfiltered	^{54}Mn concentration in plant (pCi/g)	^{54}Mn concentration in water (pCi/l)	BF[a]	Location	Reference
Chlorophyta (green algae) and Bacillariophyceae (diatoms)	phytoplankton	11±4 (14)[b]	0.00092± 0.00008 (53)				12,000	Lake Michigan	5.5
Mix of 1. Lyngbya, Oscillatoria, Phormidium; 2. Cosmarium; and 3. Nitzchia, Melosira	1. filamentous blue-green algae; 2. green algae; and 3. diatoms				11.0	0.48[c]	23,000	Reactor discharge canal on Connecticut River	5.1
Stigeoclonium lubricum	filamentous green algae				2.3×10^5	4.5×10^4	5,000	Laboratory, continuous flow exp. (23°-32°C)	5.12
Navicula seminulum	unicellular diatom				0.70×10^5	4.5×10^4	1,600	Laboratory, continuous flow exp. (23°-32°C)	5.12
Plectonema boryanum	filamentous blue-green algae				1.4×10^5	4.5×10^4	3,200	Laboratory, continuous flow exp. (23°-32°C)	5.12
Potamogeton perfoliatus	submerged macro-phyte	250-2500	0.01-0.1				25,000[d]	Hudson River	5.14
Ceratophyllum sp.	rootless, sub-merged thin stemmed macro-phyte	81	0.003	0.01			8,100	Lake Maggiore, Italy, mesotrophic	5.19
Lagarosiphon sp.		6	0.003	0.01			600	Lake Maggiore, Italy, mesotrophic	5.19
Elodea sp.	submerged aquatic plant	105		0.026			4,000	Lake Maggiore, Italy, mesotrophic	5.17
Potamogeton sp. (pondweed)	submerged floating levels	46		0.026			18,000	Lake Maggiore, Italy, mesotrophic	5.17

Table 3.5.3 (continued)

Species	Description	Stable element concentration in plant (ppm)	Stable element concentration in water (ppm) Filtered	Unfiltered	^{54}Mn concentration in plant (pCi/g)	^{54}Mn concentration in water (pCi/l)	BF[a]	Location	Reference
Myriophyllum sp. (millfoil)	mostly submerged, top of plant out of water	32		0.026			1,200	Lake Maggiore, Italy, mesotrophic	5.17
Phragmites sp. (reed grass)	emergent	28		0.026			1,100	Lake Maggiore, Italy, mesotrophic	5.17
Nymphea lutea (water lily)	large floating leaves	9	0.003	0.01			900	Lake Maggiore, Italy, mesotrophic	5.19
Najas sp. (pondweed)	submerged	14		0.026			540	Lake Maggiore, Italy, mesotrophic	5.17
Nuphar sp. (water lily)	large floating leaves	5		0.026			190	Lake Maggiore, Italy, mesotrophic	5.17

[a]Bioaccumulation factors are all based on unfiltered concentrations in water unless only filtered values were reported.

[b]Mean ± S.E. (no. of samples) of all non-zero "corrected" (Copeland and Ayers 1970) concentrations.

[c]Estimate from release rates.

[d]Estimate from nonlinear regression analysis.

that a value of 10^4 be used as the manganese bioaccumulation factor in algae. For submerged macrophytes we recommend a bioaccumulation factor of 10^4. On the basis of the few data from Lake Maggiore, we recommend that a value of 10^3 be used for macrophytes with floating leaves and emergent vegetation.

3.5.3.3 Manganese Bioaccumulation Factors for Invertebrates

3.5.3.3.1 Manganese Bioaccumulation Factors for Molluscs

In bivalves, manganese concentrations in both the shell and soft tissues increase with size (Merlini, 1966; Merlini et al., 1965; Merlini et al., 1967). This fact explains part of the standard error associated with the mean concentrations of manganese given for bivalve tissues in Table 3.5.4. The data in Table 3.5.4 were averaged over all size classes. Manganese concentrations in the mantle and visceral sac may vary considerably with nature of the substrate for the same lake. This factor is in large part responsible for the large standard errors for Unio in Lake Maggiore since Unio was taken from sites having different substrates.

In snails, manganese concentrations do not vary greatly with size (Merlini, 1966; Merlini et al, 1965; Merlini et al, 1967). Moreover, only small differences in manganese concentration were found in the snail Viviparus with change in habitat within the same lake (Merlini et al., 1965). The data on bioaccumulation factors for manganese in molluscs (Table 3.5.4) suggest that $BF(Mn)_i$ is constant or that manganese concentrations are only partially homeostatically controlled. The bioaccumulation factor for Unio tissues in Lake Maggiore is about

Table 3.5.4 Bioaccumulation factors for manganese in molluscs

Taxon/Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppm)[a]	Bioaccumulation Factor	Location	Reference
Bivalves:					
Anodonta cygnea				Lake Maggiore	5.18
Shell	419±50(2)[b]	0.016	26,000		
Gills	2450±150(2)[b]		150,000		
Mantle	720±190(2)[b]		45,000		
Visceral sac	155±35(2)[b]		9,700		
Muscle	30.5±12(2)[b]		1,900		
Anodonta cygnea					
Whole	341±36(4)	~0.1	3,400	Irrigation canal, Crescentino, Italy	5.4
Whole	206±7(4)	~0.1	2,100	Irrigation canal, Vercelli, Italy	5.4
Lamellidens marginalis		0.011[c]		Lake Masunda, India	5.21
Labial palps	9872		900,000		
Gills	3210		300,000		
Mantle	1228		110,000		
Visceral sac	1220		110,000		
Muscle	834		80,000		
Foot	354		30,000		
Total soft	1247		110,000		
Shell	942		90,000		
Quadrula pustulosa		0.004[c]			5.20
Shell	486		120,000		
Unio mancus elongatus		0.016		Lake Maggiore	5.16, 5.18, 5.19
Shell	382±43(12)[d]		24,000		
Gills	2100±260(7)[d]		130,000		
Mantle	1400±330(7)[d]		88,000		
Muscle	260±53 (7)[d]		16,000		
Visceral sac	370±96 (7)[d]		23,000		

Table 3.5.4 (continued)

Taxon/Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppm)[a]	Bioaccumulation Factor	Location	Reference
Unio mancus elongatus		0.033		Varese	5.16
Shell	870±57(3)[b]		26,000		
Mantle	2400±300(3)[b]		71,000		
Visceral sac	600±58(3)[b]		18,000		
Snails:					
Lymnea ovata		0.016		Lake Maggiore	5.18, 5.19
Shell	35±1(4)		2,200		
Soft tissues	28±1(4)		1,800		
Lymnea peregra					
Whole	126±41(3)	∿0.1	1,300	Irrigation canal, Casalino, Italy	5.4
Whole	85 (1)	∿0.1	850	Irrigation canal, Vercelli, Italy	5.4
Lymnea stagnalis					
Whole	107±25(6)	∿0.1	1,100	Irrigation canal, Casalino, Italy	5.4
Whole	70±6(5)	∿0.1	700	Irrigation canal, Crescentino, Italy	5.4
Whole	110±14(15)	∿0.1	1,100	Irrigation canal, Vercelli, Italy	5.4
Physa acuta					
Whole	25±2(16)	0.009	2,800	Irrigation canal, Cameri, Italy	5.4
Whole	85±13(2)	∿0.1	850	Irrigation canal, Casalino, Italy	5.4
Planorbis sp.					
Whole	599±221(3)	∿0.1	6,000	Irrigation canal, Casalino, Italy	5.4
Whole	750±56(2)	∿0.1	7,500	Irrigation canal, Vercelli, Italy	5.4

Table 3.5.4 (continued)

Taxon/Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppm)[a]	Bioaccumulation Factor	Location	Reference
Viviparus ater		0.016		Lake Maggiore	5.4
Shell	88 ± 3.7(19)[d]		5,500		
Soft tissues	12 ± 0.46(19)[d]		750		
Viviparus contectus					
Whole	202 ± 53(4)	$\sim$0.1	2,000	Irrigation canal, Crescentino, Italy	5.4
Whole	92 ± 8(6)	$\sim$0.1	920	Irrigation canal, Vercelli, Italy	5.4

[a]Unfiltered concentrations unless indicated otherwise.

[b]Average of means for size classes.

[c]Filtered concentration.

[d]Average over all size classes and study sites.

the same as that in Lake Varese despite a two-fold difference in manganese concentration in the water. The whole-body bioaccumulation factor for *Physa* is about 3.4 times higher for the environment having a manganese concentration in water an order of magnitude lower than that of the other. However, we must not put too much emphasis on these results because the manganese concentrations in water, which are based on only a few measurements, can only be considered rough indicators of the concentration that the organisms would be exposed to over long periods of time since the manganese concentration in water has strong seasonal variability (Section 3.5.1). Further, as pointed out in Section 2.5.5, sediment contamination may affect whole-body concentrations of manganese in invertebrates.

For studies in which bioaccumulation factors are based on unfiltered values for the manganese concentration in water, the bioaccumulation factors for all soft tissues in bivalves except gills are less than 10^5. Thus, we recommend that a bioaccumulation factor of 10^5 be used for soft tissues of bivalves (Table 3.5.4). Note that the value of this recommended bioaccumulation factor compares favorably with the ^{54}Mn bioaccumulation factor of 4.5×10^4 determined from the fallout data of Gaglione and Ravera (1964) for all soft tissues of *Unio* from Lake Maggiore. On the basis of data in Table 3.5.4, we recommend that a bioaccumulation factor of 3×10^4 be used for bivalve shells.

Based on the few data for snails we recommend that a bioaccumulation factor of 10^4 be used for snail shells and whole snails. For soft tissues of snails, we recommend a bioaccumulation factor of 2×10^3.

3.5.3.3.2 Manganese Bioaccumulation Factors for Invertebrates other than Molluscs

On the basis of data in Table 3.5.5, we recommend that a bioaccumulation factor of 10^4 be used for crustaceans. It is apparent from Table 3.5.5 that there is considerable variation in manganese bioaccumulation factors among insect species. Owing to the lack of comprehensive data, we must recommend that a bioaccumulation factor of 10^3 be applied to insects in general.

Table 3.5.5 Bioaccumulation factors for manganese in invertebrates other than molluscs

Species	Description	Size class or life stage	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)[a]	BF	Location	Reference
Calanoid and cyclopoid copepods	zooplankton		whole	3.6±0.8(33)[b]	0.92±0.08(53)[c]	3,900	Lake Michigan	5.5
			whole	4.2±0.7(40)[d]		4,600	Lake Michigan	5.5
Amphipoda (mostly *Pontoporeia* sp.)	small shrimp		whole	7.4±2.1(17)[b]	0.92±0.08(53)[c]	8,000	Lake Michigan	5.5
Ranatra linearis	carnivorous aquatic hemipteran, feeding on tadpoles, insects and larvae	adult	whole	51.8 (1)	9	6,000	Cameri, Italy[e]	5.4
		adult	whole	333±86(2)	~100	3,000	Casalino, Italy	5.4
		adult	whole	354±50(2)	~100	4,000	Crescentino, Italy	5.4
Notonecta glauca	aquatic insect	adult	whole	6.9±0.8(2)	~100	70	Casalino, Italy	5.4
			whole	5.8 (1)	~100	60	Oldenico, Italy	5.4
Hydrophilus piceus	carnivorous aquatic insect	adult	whole	9.4±0.7(2)	9	1,000	Cameri, Italy	5.4
			whole	34.4±7.6(4)	~100	300	Casalino, Italy	5.4
			whole	76.9±1.9(3)	~100	800	Crescentino, Italy	5.4
			whole	44.7±6.6(4)	~100	400	Oldenico, Italy	5.4
			whole	17.3±0.2(4)	~100	200	Vercelli, Italy	5.4
Dysticus marginalis	aquatic insect	adult	whole	6.6 (1)	9	700	Cameri, Italy	5.4
			whole	14.7±7.9(2)	~100	100	Casalino, Italy	5.4
			whole	40.1 (1)	~100	400	Crescentino, Italy	5.4
Haemopis sp. (bloodworm)	parasitic leech found on fish and insects		whole	9.5	~100	100	Oldenico, Italy	5.4

[a]Unfiltered values unless indicated otherwise.

[b]Mean ± S.E. (no. of samples) of all non-zero "corrected" (Copeland and Ayers 1970) concentrations.

[c]Filtered value.

[d]Uncorrected value.

[e]Irrigation canal at this and following locations.

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3.6 COBALT

3.6.1 Cobalt Metabolism

Cobalt is essential for some bacteria, some fungi, several species of blue green algae and several species of mammals (Bowen, 1966). Cobalt II activates enzymes. A principle use of cobalt III is in cyanocobalamin (vitamin B_{12}). Cobalamin is not synthesized by animals but is synthesized by bacteria and actinomycetes (Sherman, 1957). Cobalamin is required by some green algae and diatoms, many dinoflagellates and yellow-green algae, higher plants, insects, fish, birds, and mammals.

Absorption efficiency of cobalt from food is about 50% in snails (Nelson and Malone, 1969) and about 5% in fishes. These low absorption efficiencies are in part responsible for the decrease in the cobalt bioaccumulation factor with increasing trophic level that is sometimes observed (Kevern and Griffith, 1966; Lowman, 1963; Morton, 1965; Ophel and Fraser, 1973). This is because the ratio of intake rate to elimination rate governs isotopic concentration of cobalt in the organism [Equations (2.4.3) and (2.4.4)].

Elimination rates for cobalt are given in Table 3.6.1. Good estimates of turnover rates come from studies in which the organisms have come into steady-state with the source of radionuclide. In such studies, long time components contain most of the radionuclide. Because of its relatively short radioactive half-life (72 days), ^{58}Co will have a lower bioaccumulation factor than the bioaccumulation factors for ^{60}Co (5.27 year half-life) and stable cobalt. However, as can be seen from the biological half-times given in Table 3.6.1,

Table 3.6.1 Biological elimination rates for ^{60}Co in various aquatic organisms

Species	Description	Biological Half-Time (days)	Elimination Rate (10^{-2} $days^{-1}$)	Comment	Percent of Total Radioactivity	Reference
Lemna minor (duckweed)	macrophyte	19	3.6	from elimination after five month uptake	100	6.27
Vallisneria americana (wild celery)	macrophyte	51	1.4	from elimination after five month uptake	100	6.27
Helisoma sp.	snail	51	1.4	from elimination after five month uptake	90	6.30
Goniobasis clavaeformis	snail	4.5	15.	fast compartment	25	6.28
		320	0.22	slow compartment snails tagged by short term feeding	25	6.25
Lampsilis radiata	clam	4.8	14.4	fast compartment	51	6.14
		277	0.25	slow compartment	49	6.14
Cambarus longulus	crayfish	70	0.99	5 g animals, high level dose	90	6.35
		37	1.9	0.9 g animals, high level dose	90	6.35
Diemictylus viridescens	newt	158	0.44	five month uptake	90	6.30
Ictalurus melas (black bullhead)	fish	1.5	46.	single feeding	97.9	6.32
		40.5	1.7	single feeding	1.8	6.32
		long, undetermined	undetermined	single feeding	0.3	6.32
		3.5	20.	4 days of water uptake	50	6.32
		3.5	2.0	4 days of water uptake	35	6.32
		long, undetermined	undetermined	4 days of water uptake	15	6.32
Hydropsyche sp.	Insect Trichoptera larva	105.3	0.66	uptake from water tag	89.3	6.13
		38.4	1.8	uptake from food ingestion	90	6.13

the ^{58}Co bioaccumulation factors are not expected to be greatly lower than the bioaccumulation factors for ^{60}Co or stable cobalt in many organisms. Considering the uncertainty in both biological half-times and in the bioaccumulation factors for ^{60}Co and stable cobalt, we will not distinguish between the bioaccumulation factors of ^{58}Co and the longer-lived isotopes.

3.6.2 Environmental Cobalt and Availability

The proportion of cobalt in the particulate phase varies greatly among bodies of water. In Par Pond, an oligotrophic reservoir of low turbidity, 22% of the stable cobalt in the water is in the particulate phase (Marshall and LeRoy, 1974). Greater than 98% of the stable cobalt in the Amazon and Yukon Rivers is particulate. The cobalt appears as adsorbed on organic solids and in the crystal structure of sediments. Fukai and Murray (1973) contrast the fraction of radiocobalt appearing in the particulate phase in the Columbia River to that in the Clinch River, Tennessee; the Columbia had from 80 to 95% (at Vancouver, Washington) and the eutrophic Clinch, from 2 to 30%, despite the much higher suspended solids concentration in the latter river. In Perch Lake, a small dystrophic-eutrophic lake, essentially all the radiocobalt in the water is in the dissolved phase (Ophel and Fraser, 1973) in contrast to mesotrophic Lake Maggiore in which roughly 80% of stable cobalt is in the particulate phase (Merlini et al., 1967). These observations would suggest that the proportion of cobalt in the particulate phase increases with increase in suspended solids concentration and decreases with increase in eutrophy. This latter correlation may be related to the tendency of cobalt to form associations with dissolved organic matter. Thus, increased dissolved organic matter concentrations in eutrophic waters may serve to keep radiocobalt in solution.

As indicated in Section 2.2, the soluble form of a radionuclide is generally more available to algae and subsequent trophic levels than the particulate form of the radionuclide. Thus, it would seem that the greater the proportion of soluble cobalt, the greater the availability. However, as indicated in Section 2.2, chelated radionuclides are less available to food webs than radionuclides appearing as free ions. For this reason, soluble cobalt in eutrophic environments is probably less available than soluble cobalt in oligotrophic or mesotrophic environments due to the tendency of cobalt to chelate or form other associations with dissolved organic matter. This expected pattern is seen in the review of the literature on cobalt bioaccumulation factors which follows.

3.6.3 Review of Cobalt Bioaccumulation Factors

Because most investigators have based their bioaccumulation factors on filtered concentrations of cobalt isotopes in water, we generally report filtered bioaccumulation factors. As indicated in Section 2.5.3 use of filtered bioaccumulation factors should generally lead to overprediction of radionuclide concentration in organisms. However, we noted that in nonturbid waters and in eutrophic waters, most of the radiocobalt in the water would be in the soluble phase. In these waters, use of filtered bioaccumulation factors would not overpredict radiocobalt concentrations in organisms.

3.6.3.1 Cobalt Bioaccumulation Factors for Fishes

Researchers have reported cobalt bioaccumulation factors for flesh and whole bodies of fishes. The relative concentrations of ^{60}Co in black bullheads from White Oak Lake may be used to convert these bioaccumulation factors to bioaccumulation factors for other tissues. The relative steady-state concentrations are given in Table 3.6.2. The organ with the highest cobalt concentration is the kidney, which has a concentration 23 times that of the whole body. Note that the flesh to whole body ratio is 0.3, which compares well with the ratio of 0.37 determined for brown bullheads from Perch Lake, Ontario (Ophel and Fraser, 1973).

All bioaccumulation factors for cobalt in fishes were derived from steady-state concentrations of ^{60}Co or stable cobalt in natural bodies of water. Because cobalt bioaccumulation factors are expected to decrease with increasing eutrophy, the trophic states of the bodies of water were recorded. The bodies of water fell into two categories, eutrophic and mesotrophic; bioaccumulation factors for the former are given in Table 3.6.3, and bioaccumulation factors for the latter, in Table 3.6.4. Mean bioaccumulation factors for whole fishes and fish flesh are given in Table 3.6.5. As expected, lower cobalt bioaccumulation factors are seen for eutrophic environments. We recommend use of the bioaccumulation factors given

Table 3.6.2 Relative steady-state concentrations of ^{60}Co in tissues of Ictalurus melas (black bullheads) from White Oak Lake. Calculated from data of Reed (1971)

Tissue	Tissue Concentration / Whole Body Concentration
Blood	4.3
Skin	0.92
Flesh	0.30
Liver	3.1
Stomach	0.29
Gut	4.0
Kidney	23
Bone	0.52
Gills	2.3

Table 3.6.3 Bioaccumulation factors for cobalt in fishes from eutrophic environments (based on filtered water concentrations)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Perca flavescens (yellow perch)	Adult (8.7-13)	piscivorous	whole			18	Perch Lake, Canada	6.29
	Adult		flesh			9		6.29
	young (2.4-3.9)	zooplankton and insect larvae	whole			130		6.29
Ictalurus nebulosus (brown bullhead)	Adult (6.7-11.8)	omnivorous, plant material	whole			52		6.29
	Adult		flesh			19		6.29
	young (2.0-2.8)		whole			63		6.29
Lepomis gibbosus (pumpkinseed sunfish)	Adult (4.7-7.9)	omnivorous, plant	whole			80		6.29
	young (2.0-3.2)	eating insect larvae	whole			94		6.29
Semotilus margarita (pearl dace)	Adult (2.8-4.7)	unknown	whole			18		6.29
Notropis heterolepis (blacknose shiner)	Adult (2.0-3.9)	unknown	whole			20		6.29
Hybopsis plumbea (lake chub)	Adult (2.0-3.9)	unknown	whole			38		6.29
Dorosoma cepedianum (gizzard shad)		piscivorous	whole fish less viscera			32	White Oak Lake, Tennessee	6.26
Lepomis macrochirus (bluegill)		chironomid larvae, terrestrial insects, fish, carnivorous	whole fish less viscera			26		6.26

Table 3.6.3 (continued)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Carassius auratus (goldfish)		sediment, algae, bryozoans, chironomids, plant remains, omnivorous	whole fish less viscera			49		6.26
Cyprinus carpio (carp)	large adults	low trophic level (algae, benthic detritovore)	whole			6.4	White Oak Lake	6.17
Ictaluras melas (black bullhead)	large adults	benthic omnivore	whole			70		6.17
Dorosoma cepedianum (gizzard shad)	large adults	partially piscivorous	whole			13		6.17
Lepomis macrochirus (bluegill)	large adults	carnivore	whole			5.0		6.17
Micropterus salmoides (largemouth bass)	large adults	piscivorous	whole			5.5		6.17
Cyprinus carpio (carp)		bottom feeder	flesh			14	Clinch River, Tennessee	6.5, 6.25
			total[a]			10		6.5, 6.25
Carpiodes carpio (carpsucker)		bottom feeder	flesh			25		6.5, 6.25
			total			7		6.5, 6.25
Ictiobus bubalus (smallmouth buffalo)		bottom feeder	flesh			17		6.5, 6.25
			total			7		6.5, 6.25

Table 3.6.3 (continued)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Assorted fish[b]		sight feeders	flesh			5		6.5, 6.25
Esox lucius (northern pike)		piscivorous	flesh	0.07(0.03-0.11)[c]	3 (1-6)[c]	23	Illinois River	6.22
Micropterus salmoides (largemouth bass)		piscivorous	flesh	0.09(0.06-0.18)		30		6.22
Morone chrysops (white bass)		piscivorous	flesh	0.07(0.05-0.10)		23		6.22
Lepisosteus platostomus (shortnose gar)		piscivorous	flesh	0.15(0.11-0.18)		50		6.22
Micropterus dolomieui (smallmouth bass)		piscivorous	flesh	0.15(0.14-0.16)		50		6.22
Ictiobus cyprinellus (bigmouth buffalo)		omnivorous insect larvae, mollusks, algae, aquatic plants	flesh	0.081(0.06-0.12)		27		6.22
Dorosoma cepedianum (gizzard shad)		omnivorous insect larvae, mollusks, algae, aquatic plants	flesh	0.16(0.1-0.25)		53		6.22
Moxostoma macrolipidotum (northern redhorse)		omnivorous insect larvae, mollusks, algae, aquatic plants	flesh	0.083(0.05-0.11)		28		6.22

Table 3.6.3 (continued)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Carpiodes cyprinus (quillback)		omnivorous insect larvae, mollusks, algae, aquatic plants	flesh	0.087(0.04-0.12)		29		6.22
Cyprinus carpio (carp)		omnivorous insect larvae, mollusks, algae, aquatic plants	flesh	0.068(0.03-0.1)		23		6.22

[a]Total fish consists of flesh and bone.

[b]Assorted fish are all sight feeders; *Pomoxis annularis* (white crappie), *Lepomis macrochirus* (bluegill), *Roccus chrysops* (white bass), *Micropterus salmoides* (largemouth bass), *Stizostedian v. vitreum* (sauger), *Aplodinotus grunniens* (drum), *Iactalurus punctalis* (catfish).

[c]Range of concentrations given in parentheses.

Table 3.6.4 Bioaccumulation factors for cobalt in fishes from mesotrophic environments (based on filtered water concentrations)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)(filtered)	BF	Location	Reference
Alosa pseudoharengus (alewife)	0.6-0.8	zooplankton feeder	whole	0.029 ± 0.014	0.19 ± 0.02	190	Lake Michigan	6.6, 6.20
Alosa pseudoharengus (alewife)	4.5-10	zooplankton feeder	whole	0.065 ± 0.005	0.19 ± 0.02	420	Lake Michigan	6.6, 6.7
		benthic feeder						
Osmerus mordax (smelt)	5	zooplankton feeder	whole	0.19 ± 0.05	0.19 ± 0.02	1000	Lake Michigan	6.6, 6.7
Alburnus alborella (bleak)	4.3-5.1	plankton feeder	flesh	0.012 ± 0.002	0.02	600	Lake Maggiore	6.23,6.24
Coregonus macrophthalmus (whitefish, Bondella)	9.8	plankton and benthic feeder	flesh	0.0077 ± 0.0016	0.02	385	Lake Maggiore	6.23,6.24
Osmerus mordax (smelt)	>5	plankton and benthic feeder	flesh	0.043 ± 0.004	0.19 ± 0.02	230	Lake Michigan	6.6, 6.7
			whole	0.091 ± 0.011	0.19 ± 0.02	480	Lake Michigan	6.6, 6.7
Percopsis omiscomaycus (trout-perch)	2.4-3.7	benthic	whole	0.024 ± 0.002	0.19 ± 0.02	130	Lake Michigan	6.6, 6.20
Notropis hudsonius (spottail shiner)	2.2-2.6	benthic and zoo-plankton	whole	0.042 ± 0.015	0.19 ± 0.0	220	Lake Michigan	6.6, 6.20
Notropis hudsonius (spottail shiner)	4.5-5.5	benthic	whole	0.12 ± 0.03	0.19 ± 0.02	630	Lake Michigan	6.6, 6.7
Notropis hudsonius (spottail shiner)	7.5-8	unknown	flesh	0.041 ± 0.003	0.19 ± 0.02	220	Lake Michigan	6.6, 6.7
Scardinius erythrophthalmus (rudd)	1	omnivorous	flesh	0.0058 ± 0.0011	0.02	290	Lake Maggiore	6.23,6.24
Perca flavescens (yellow perch)	8-13	partially piscivorous	flesh	0.053 ± 0.0089	0.19 ± 0.02	280	Lake Michigan	6.6, 6.7

Table 3.6.4 (continued)

Species	Size class (in.)	Description	Tissue	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)(filtered)	BF	Location	Reference
Salvelinus namaycush (lake trout)	>15	piscivorous	flesh	0.045 ± 0.0061	0.19 ± 0.02	236	Lake Michigan	6.6, 6.7
Salmo trutta (brown trout)	>12	piscivorous	flesh	0.044 ± 0.0064	0.19 ± 0.02	230	Lake Michigan	6.6, 6.7
Oncorhynchus kisutch (coho salmon)	6-27	piscivorous	flesh	0.044 ± 0.0036	0.19 ± 0.02	230	Lake Michigan	6.6, 6.7
Prosobium cylindraceum (round whitefish)	>13	benthic (>10 inches)	flesh	0.040 ± 0.0038	0.19 ± 0.02	210	Lake Michigan	6.6, 6.7
Ictalurus melas (black bullhead)		benthic omnivore	flesh	0.030	0.046[a]	650	East Tennessee spring	6.32

[a] D. J. Nelson, private communication.

Table 3.6.5 Mean bioaccumulation factors for cobalt in fishes from mesotrophic and eutrophic waters

Nature of body of water	Number of bodies of water	Bioaccumulation factors	
		Flesh	Whole
Mesotrophic	3	323 ± 47(11)	439 ± 115(7)
Eutrophic	4	26.6 ± 3.5(16)	43.8 ± 10.4(14)

in Table 3.6.5 for mesotrophic and eutrophic environments. In the absence of data for oligotrophic waters, we suggest using the mean value from the mesotrophic waters.

Lucas and Edgington (1970) have suggested that cobalt concentration may be under partial homeostatic control. The inverse correlation between the cobalt bioaccumulation factor and eutrophy could be due to partial homeostatic control if cobalt concentration in the water increased with eutrophy. In Figure 3.6.1 the cobalt concentrations in fish flesh are plotted against cobalt concentrations in water for three mesotrophic waters and one eutrophic water, the Illinois River. If the slope (exponent) of the logarithmic regression were 1, the cobalt bioaccumulation factor would be independent of the cobalt concentration in water. Since the slope is less than 1, either partial homeostatic control or eutrophy coincident with increasing cobalt concentration in water may be operating to diminish the slope. Lack of more extensive data on bioaccumulation factors from environments exhibiting a wide range of cobalt concentrations within each of a few distinct eutrophy categories prevent us from firmly deciding between the hypotheses of partial homeostatic control and eutrophy. In view of the known tendency of cobalt to form organic associations, the latter explanation seems more plausible.

Some workers have observed that the cobalt bioaccumulation factor in fishes decreases with increase in trophic level (Morton, 1965; Nelson et al., 1971; Ophel and Fraser, 1973). Other workers have not found a clear-cut correlation (Kevern and Griffith, 1966;

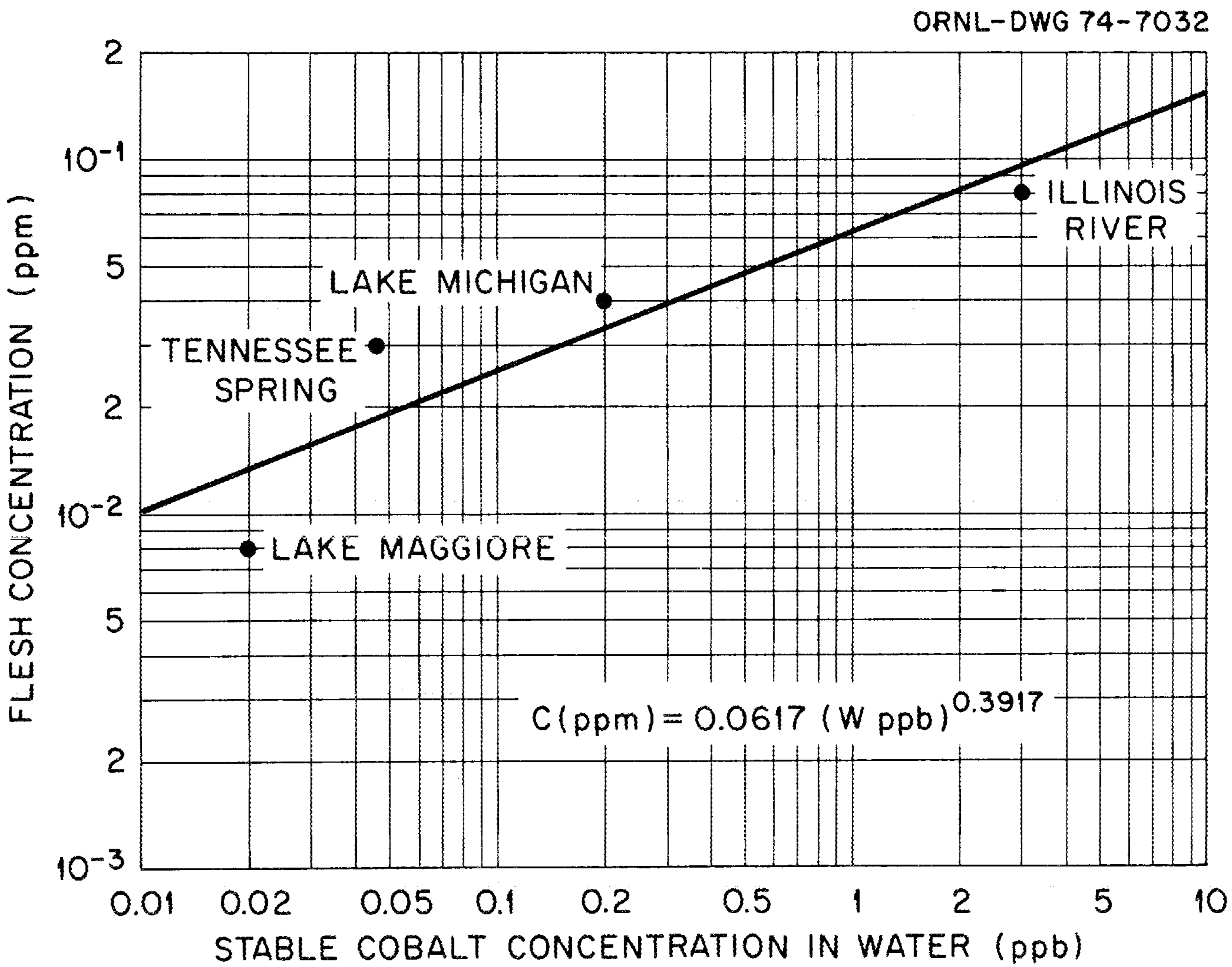

Figure 3.6.1 Concentration of stable cobalt in fish flesh in various environments.

Mathis and Cummings, 1973). For this reason we have specified the feeding habits of the fishes in Tables 3.6.3 and 3.6.4. Based on the limited and conflicting data at hand, however, we cannot specify a relation between the cobalt bioaccumulation factor in fishes and their feeding habits.

3.6.3.2 Cobalt Bioaccumulation Factors for Plants

In Table 3.6.6 are listed cobalt bioaccumulation factors for algae and vascular plants. Compared to fishes the cobalt bioaccumulation factors of plants are high. On the basis of the few data on algae in Table 3.6.6, we recommend a cobalt bioaccumulation factor of 10^4 for algae. The two species of emergent vascular plants exhibit widely differing cobalt bioaccumulation factors. The low bioaccumulation factor was calculated from ^{60}Co concentrations, and the high bioaccumulation factor was calculated from stable element concentrations. Possibly, because diffusion of elements in interstitial water of sediment is slow, the ^{60}Co concentration in interstial water from which Pontederia obtained its ^{60}Co had not reached steady state with the ^{60}Co in the water column above. Based on the stable cobalt bioaccumulation factors for Phragmites, we recommend a bioaccumulation factor of 10^3 for emergent vascular plants.

Physical form of aquatic vascular plants may affect the cobalt bioaccumulation factor. Merlini et al. (1971) reported that floating leaf species have lower bioaccumulation factors than submerged species. Ophel and Fraser (1973) noted that the finely divided submerged species had the highest bioaccumulation factors, that other submerged

Table 3.6.6 Bioaccumulation factors for cobalt in aquatic plants

Taxon	BF (filtered)	BF (unfiltered)	Location	Reference
Algae:				
blue-green algae	30,000		Connecticut River	6.2
diatoms	1,500		Lake Michigan	6.6
Navicula seminulum (diatom)	1,100-2,000		Laboratory	6.16
Nitella sp. (green alga)	500		Perch Lake	6.29
Plectonema boryanum (filamentous blue-green)	250-620		Laboratory	6.16
Stigeoclonium lubricum (filamentous green)	2,800		Laboratory	6.16
Emergent vascular plants:				
Phragmites sp.	2,000	400	Lake Maggiore	6.23
Pontederia cordata	20		Perch Lake	6.29
Vascular plants, floating leaves:				
Nuphar sp.	800	160	Lake Maggiore	6.23
Nuphar variegatum	200		Perch Lake	6.29
Nymphea lutea	900	200	Lake Maggiore	6.24
Nymphea odorata	200		Perch Lake	6.29
Potamogeton amplifolius	600		Perch Lake	6.29
Potamogeton natans	500		Perch Lake	6.29
Sparganium fluctuans	300		Perch Lake	6.29

Table 3.6.5 (continued)

Taxon	BF (filtered)	BF (unfiltered)	Location	Reference
Vascular plants, submerged leaves:				
Ceratophyllum demersum	1,000		Perch Lake	6.29
Ceratophyllum sp.	5,000	1,000	Lake Maggiore	6.24
Elodea sp.	15,000	3,000	Lake Maggiore	6.23
Myriophyllum sp.	12,000	2,400	Lake Maggiore	6.23
Nojas sp.	2,400	500	Lake Maggiore	6.23
Potamogeton amplipolius	2,000		Perch Lake	6.29
Potamogeton pusillus	2,800		Perch Lake	6.29
Potamogeton sp.	10,000	2,000	Lake Maggiore	6.23
Utricula vulgans	600		Perch Lake	6.29
Mixed species (Potamogeton, Myriophyllum, Vallisneria)	5,000[a]		Hudson River	6.18

[a]Geometric mean of five years of data.

species had intermediate bioaccumulation factors, and that floating plants had the lowest. In Table 3.6.7 we give mean cobalt bioaccumulation factors for submerged and floating vascular plants from Lake Maggiore and Perch Lake. In Lake Maggiore the cobalt bioaccumulation factors for submerged plants are significantly greater than those for floating plants.

In Table 3.6.7 the mean cobalt bioaccumulation factor for submerged vascular plants from mesotrophic Lake Maggiore is significantly greater than the mean bioaccumulation factor for dystrophic-eutrophic Perch Lake. This conforms to the expected result of decreased cobalt bioaccumulation factors with increased eutrophy. We will assume the mean values given in Table 3.6.7 for Lake Maggiore and Perch Lake to be representative of mesotrophic and eutrophic waters, respectively. Thus, the cobalt bioaccumulation factors for submerged and floating vascular plants in eutrophic waters are 2×10^3 and 4×10^2, respectively. For mesotrophic and oligotrophic waters we recommend cobalt bioaccumulation factors of 10^4 for submerged vascular plants and 10^3 for floating vascular plants.

3.6.3.3 Cobalt Bioaccumulation Factors for Invertebrates

The few cobalt bioaccumulation factors available for invertebrates are listed in Table 3.6.8. The cobalt bioaccumulation factor for the bivalve Unio from mesotrophic Lake Maggiore is about 25 times larger than the bioaccumulation factors for other bivalves from the eutrophic Illinois River and the dystrophic-eutrophic Perch Lake. We attribute this difference to differences in degree of eutrophy. For bivalves in eutrophic environments we recommend

Table 3.6.7 Mean cobalt bioaccumulation factors for submerged and floating-leaf vascular plants from Perch Lake and Lake Maggiore (based on filtered water concentrations)

Location	Bioaccumulation factors		Significance of difference between plant types
	Submerged	Floating	
Lake Maggiore	8,900 ± 2,300(5)	850 ± 50(2)	$P < 0.05$
Perch Lake	1,600 ± 500(4)	360 ± 81(5)	$P \approx 0.10$
Significance of differences between lakes	$P < 0.05$	$P \approx 0.06$	

Table 3.6.8 Bioaccumulation factors for cobalt in invertebrates (based on filtered water concentrations)

Taxon	Tissue	Feeding habits	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Bivalves:							
Amblema plicata	soft	filter feeder	0.7	3	200	Illinois River	6.2
Elliptio sp.	soft	filter feeder			330	Perch Lake	6.9
Fusconaia flava	soft	filter feeder	1.2	3	400	Illinois River	6.22
Quadrula quadrula	soft	filter feeder	0.8	3	300	Illinois River	6.22
Unio mancus	soft	filter feeder	0.18-0.20	0.02	9,000-10,000	Lake Maggiore	6.24
Crustaceans:							
Cambarus sp. (crayfish)	whole				1,600	Perch Lake	6.29
Copepods	whole	filter feeders	0.134	0.19	700	Lake Michigan	6.6
Mysis relicta	whole	detritus	0.034	0.19	200	Lake Michigan	6,6, 6.10
Insect larvae:							
Argyractis sp.	whole				23,000[a]	Columbia River	6.8
Glossoma sp.	whole				11,000[a]	Columbia River	6.8
Hydrobaeninae	whole	periphyton			5,000[a]	Columbia River	6.8
Hydropsyche cockerelli	whole	phytoplankton			15,000[a]	Columbia River	6.8
Tendipedinae	whole	detritus			3,000[a]	Columbia River	6.8
Snails:							
Amnicola sp.	whole				4,400	Perch Lake	6.29
Stagnicola nuttalliana	soft				9,000[a]	Columbia River	6.9

Table 3.6.8 (continued)

Taxon	Tissue	Feeding habits	Stable element concentration in tissue (ppm)	Stable element concentration in water (ppb)	BF	Location	Reference
Snails (continued):							
Viviparus ater	soft	detritus	0.24-0.51	0.02	12,000-26,000	Lake Maggiore	6.24
Tubificids:							
Limnodrilus hoffmeisteri and *Tubifex tubifex*	whole	detritus	1.6	3	500	Illinois River	6.22

[a]Based on approximate ^{60}Co concentration in water of 2 pCi/l (Watson, personal communication).

a cobalt bioaccumulation factor of 4×10^2. For bivalves in mesotrophic or oligotrophic waters we recommend a cobalt bioaccumulation factor of 10^4. The few data available for insect larvae and snails suggest a cobalt bioaccumulation factor of 10^4 for both of these groups. The cobalt bioaccumulation factor for tubificids in the eutrophic Illinois River is 500, a value that we will assume is characteristic of eutrophic environments. The few data for crustaceans suggest a cobalt bioaccumulation factor of 10^3.

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APPENDIX

EFFECT OF CARRIER CONCENTRATION ON THE DISCRIMINATION COEFFICIENT

The discrimination coefficient, q_i, is usually treated as a constant. The purpose of this appendix is to show that the coefficient can vary with concentration of the carrier in the water and in other sources. Treating i as a single compartment, the time rate of change in amount, R_i, of radionuclide R in organism or tissue i is:

$$\frac{dR_i}{dt} = I_T - R_i k \qquad \text{(I-1)}$$

and the steady-state amount, C_i^*, of carrier element C^* is given by

$$C_i^* = \frac{I_T^*}{k^*} \qquad \text{(I-2)}$$

where

I_T = rate of input of radionuclide from all sources (μCi/day)

I_T^* = rate of input of carrier element from all sources (μg/day),

k = excretion rate coefficient for the radionuclide in i (day^{-1}),

k^* = excretion rate coefficient for the carrier element in i at steady-state concentration (day^{-1}).

At steady state, Eqs. (I-1) and (I-2) imply that

$$(R/C^*)_i \equiv (R_i/C_i) = \frac{I_T k^*}{I_T^* k} \qquad \text{(I-3)}$$

For the case where uptake is from a single prey j and from water:

$$I_T = I_w + I_j, \text{ and} \qquad \text{(I-4)}$$

$$I_T^* = I_w^* + I_j^* , \qquad \text{(I-5)}$$

where

I_w and I_j are uptake rates (μCi/day) of a radionuclide from water and j, respectively; and I_w^* and I_j^* are uptake rates (μg/day) of carrier element from water and j, respectively.

For uptake from water:

$$I_w = a[R]_w \text{ and,} \qquad \text{(I-6)}$$

$$I_w^* = a^*[C^*]_w \qquad \text{(I-7)}$$

where

a = uptake rate coefficient for radionuclide from water (g/day), and

a* = uptake rate coefficient for carrier element from water (g/day).

For uptake from food:

$$I_j = b[R]_j Q \text{ and,} \qquad \text{(I-8)}$$

$$I_j^* = b^*[C^*]_j Q \qquad \text{(I-9)}$$

where

b = absorbtion efficiency of radionuclide uptake from food (unitless),

b^* = absorbtion efficiency of carrier element uptake from food (unitless), and

Q = feeding rate (g/day).

Thus,

$$(R/C^*)_i{}' = \frac{k^*}{k}\left[\frac{a[R]_w + b[R]_jQ}{a^*[C^*]_w + b^*[C^*]_jQ}\right] \tag{I-10}$$

$$= \frac{k^*}{k}\left[\frac{a[C^*]_w\left(\frac{R}{C^*}\right)_w + b[C^*]_j\left(\frac{R}{C^*}\right)_jQ}{a^*[C^*]_w + b^*[C^*]_jQ}\right] \tag{I-11}$$

Dividing Eq. (I-11) through by $(R/C^*)_w$ and incorporating the elimination coefficient of the prey, q_j, gives

$$q_i \equiv \frac{(R/C^*)_i}{(R/C^*)_w} = \frac{k^*}{k}\left[\frac{a[C^*]_w + b[C^*]_j\ q_jQ}{a^*[C^*]_w + b^*[C^*]_jQ}\right]\ , \tag{I-12}$$

Equation (I-12) may be rewritten as

$$q_i = \frac{k^*}{k}\left[\frac{a^*\left(\frac{a}{a^*}\right)[C^*]_w + b^*\left(\frac{b}{b^*}\right)[C^*]_j\ q_jQ}{I_T^*}\right] \tag{I-13}$$

$$= \frac{k^*}{k}\left[\frac{I_w{}^*}{I_T{}^*}\left(\frac{a}{a^*}\right) + \frac{I_j{}^*}{I_T{}^*}\left(\frac{b}{b^*}\right)\ q_j\right] \tag{I-14}$$

Because of homeostatic control, $[C^*]_j$ is constant. Thus, as can be seen from Equation (I-14), q_i should vary with $[C^*]_w$ since uptake from water is assumed proportional to $[C^*]_w$.

Equation (I-14) may be generalized for any set of sources:

$$q_i = \frac{k^*}{k} \left[\sum_{j \neq i} \frac{I_{ij}^*}{I_T^*} \left(\frac{a_{jj}}{a_{ij}^*} \right) q_j \right] \qquad \text{(I-15)}$$

where

I_{ij}^* = intake rate of carrier element by i from j (μg/day), and

a_{ij} = absorption efficiency of uptake of radionuclide from source j at surfaces of i (unitless), and

a_{ij}^* = absorption efficiency of uptake of carrier element from source j at surfaces of i (unitless).

If water is a source, where j = 1 for water, a_{i1} is absorption of the radionuclide brought to the body surfaces and $q_1 \equiv 1$.

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