

EFFECTS OF IONIZING RADIATION ON AQUATIC ORGANISMS

**Recommendations of the
NATIONAL COUNCIL ON RADIATION
PROTECTION AND MEASUREMENTS**

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Preface

The National Council on Radiation Protection and Measurements (NCRP) was requested by the US Department of Energy (DOE) to review the literature on the effects of radiation on aquatic organisms and develop a document that reviews the present understanding of such effects and provides guidance for a standard for the protection of populations of aquatic organisms. This request derives from concerns expressed to DOE that deleterious effects may be occurring in freshwaters at DOE operating facilities and that the Department has not adopted a standard for protecting aquatic organisms residing in those environments. Although DOE is not aware of any conditions in the United States where concentrations of radioactivity in aquatic environments result in radiation doses sufficient to cause deleterious effects on populations of aquatic organisms DOE believes that a standard needs to be developed that defines a dose below which deleterious population effects are not expected to occur. This report addresses this subject and provides the necessary guidance on the protection of populations of aquatic organisms. The International System of Units (SI) is used in this report in accordance with the procedure set forth in NCRP Report No. 82, *SI Units in Radiation Protection and Measurements* (NCRP, 1985).

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Contents

Preface	iii
1. Introduction	1
2. Effects Due to Acute Exposure	3
2.1 Mortality	3
2.2 Physiological and Pathological Changes	4
2.3 Developmental Effects	5
2.4 Reproductive Effects	6
2.5 Environmental Factors	7
2.6 Summary	8
3. Effects Due to Chronic Exposure	9
3.1 Mortality	10
3.2 Physiological Effects	12
3.3 Reproductive Effects	14
3.4 Effects on Growth and Development	19
3.5 Summary	25
4. Cytogenetic and Genetic Effects	29
4.1 Cytogenetic Studies	29
4.2 Genetics	33
4.3 Summary	34
5. Criteria for the Protection of Populations of Aquatic Organisms	36
6. Environmental Dosimetry	39
6.1 Methods	40
6.2 Summary	48
7. Dose to Aquatic Organisms and Man from Environmental Radioactivity	50
8. Conclusions	61
Appendix A Dose Rate Estimates to Aquatic Biota at Example Sites	63
Glossary	83
References	84
The NCRP	97
NCRP Publications	104
Index	114

1. Introduction

The practice of discharging radioactive effluents or wastes either directly to the aquatic environment, or to surface or sub-surface disposal sites on land, has the potential for contaminating aquatic environments to some degree with resultant long-term, low dose rate exposures of the components of such ecosystems (IAEA, 1976; IAEA, 1979; NCRP, 1984). The control of such discharges is primarily based upon the need to ensure that the subsequent direct or indirect potential exposures of humans, either as individuals or as populations, do not exceed acceptable limits. The potential exposure of populations of aquatic organisms associated with those freshwater environments usually has been of secondary concern; nevertheless, rational resource management requires that their potential exposure should also be assessed and included in any consideration of the overall acceptability of a proposed or expected waste disposal practice (ICRP, 1979).

While any such discharges inevitably result in increased exposure of populations of organisms in the affected environment, it is quite clear that the potential impact of the increased exposure can be assessed only if the magnitude of the incremental radiation dose can be determined with assurance and compared with appropriate standards.

The objectives of this report are:

- to review the available literature on the effects of ionizing radiation on aquatic organisms
- to provide guidance for the establishment of a dose rate below which deleterious effects to aquatic populations are acceptably low
- to provide a series of simple dosimetric models that can be employed to demonstrate compliance with such guidance
- to evaluate the validity of the statement that *"if man is adequately protected, then other living things are also likely to be sufficiently protected"* (ICRP, 1977)
- and to make recommendations for pertinent future research.

The scope of this report is limited to consideration of radiation effects upon truly aquatic organisms (e.g., fish, crustaceans, molluscs and benthic invertebrates); detailed consideration has not been given

to semi-aquatic organisms such as birds and muskrats. The International Atomic Energy Agency (IAEA) is presently reviewing the potential effects of radiation on terrestrial organisms, and its review and recommendations will be published as an IAEA Safety Series document in the near future.

Over the last two decades a number of reviews of the effects of radiation on aquatic organisms have been published (Polikarpov, 1966, Templeton *et al.*, 1971; Chipman, 1972; Ophel *et al.*, 1976; Templeton *et al.*, 1976; Woodhead *et al.*, 1976; Blaylock and Trabalka, 1978; IAEA, 1979; Egami, 1980; NRCC, 1983; Woodhead, 1984; Anderson and Harrison, 1986). Whereas these detailed reviews considered field studies and laboratory experimental data from both the marine and freshwater environments, by far the largest amount of data has been collected on marine species. Where reasonable comparisons can be made, there is a lack of evidence that significant differences in responses to radiation exist between marine and freshwater organisms (IAEA, 1976). These reviews and those papers published subsequently provided the basis for the considerations discussed in this report.

2. Effects Due To Acute Exposure

Historically, most of the early research on the acute effects of ionizing radiations on aquatic species was conducted in a comparative framework. Experimental designs for mammals and aquatic organisms have followed the same protocols even though the differences in organisms were significant. Although chronic exposures are expected to produce the principal biological effects, research on the effects of acute exposures to ionizing radiations of aquatic organisms provides useful information in three areas: (1) relative sensitivities of species, (2) relative sensitivities of different life stages of the same species, and (3) interactions between radiation and other environmental factors that affect radiation response. The concern in the following discussion is to show how studies of the responses of aquatic organisms to acute radiation exposure have provided the necessary background information to design and interpret chronic or low-level irradiation experiments. Acute exposure experiments are particularly useful in demonstrating that radiation response at the organismal, cellular, and biochemical levels can be modified by alterations of environmental conditions. Such experiments may not be practical or cost effective for long-term exposures.

2.1 Mortality

The first determination in risk assessment of the effects of ionizing radiation on aquatic organisms is the measurement of the upper limit of radiation sensitivity. The generally accepted measure of lethal effects is the LD_{50} , i.e., that dose of radiation that is required to kill 50% of a population of organisms in a specified period of time. For mammals, that time period is usually 30 days, but for aquatic organisms the 30-day time period is generally not appropriate because of the variety of life spans (i.e., days to years), variable metabolic rates, and dependency upon the environmental conditions. Reviews on the topic of radiation responses of aquatic organisms have been published by a number of authors (Templeton *et al.*, 1971; Rice and Baptist, 1974; Templeton *et al.*, 1976; Ophel *et al.*, 1976;

Blaylock and Trabalka, 1978; NRCC, 1983; Anderson and Harrison, 1986). A compilation of radiation sensitivities from these reviews is shown in Table 2.1.

In general, aquatic organisms tend to be more resistant to radiation than terrestrial mammals. Since aquatic organisms are poikilothermic and some may be euryhaline, their radiation responses can be modified significantly by changing environmental conditions. This particular topic is discussed in detail later in this section.

2.2 Physiological and Pathological Changes

The physiological and pathological changes that occur in aquatic organisms in response to acute radiation exposure are characterized by changes at both the cellular and biochemical levels. Generally these changes have been studied using the mammalian radiation syndrome as a model. Many of the difficulties encountered in studies of aquatic organisms are due to the poikilothermic nature of the organisms. In addition, aquatic organisms live in a liquid medium that varies in composition and quality. The mammalian model for the effects of acute radiation exposure on blood-forming tissues was used for fish. The time scale of the effects of radiation on red blood cell turnover was different from the mammalian model and did not

TABLE 2.1—Range of sensitivities of aquatic organisms to acute radiation exposure as measured by LD_{50} ^a

ORGANISMS	RADIATION DOSE						
	Gy	0.1	1.0	10.0	100.0	1000.0	10,000.0
Vertebrates							
Freshwater fish				<----->			
Saltwater fish				<----->			
Amphibians			<----->				
Invertebrates							
Crustaceans				<----->			
Molluscs						<----->	
Echinoderms						<----->	
Microorganisms							
Protozoans				<----->			
Algae				<----->			
Bacteria				<----->			

^aRadiation either hard x-ray or gamma. Radiosensitivities are based on LD_{50} s for aquatic species published in the following sources: Templeton et al., 1971; Rice and Baptist, 1974; Ophel et al., 1976; Templeton et al., 1976; Blaylock and Trabalka, 1978; NRCC, 1983; and Anderson and Harrison, 1986.

correlate well, because the half-life of the fish red blood cell was about three times longer (Engel, *et al.*, 1966; Lockner, *et al.*, 1972; Cosgrove, *et al.*, 1975). The results showed that either high doses were required to produce an effect or that the repair of the hematopoietic tissue might occur before the circulating red blood cells could be severely affected.

Anderson and Harrison (1986) discussed and tabulated a number of physiological responses among fish and invertebrates. In addition to the effects on blood-forming tissues of fish, considerable work has been done on the immune response of fish to radiation (Preston, 1959; Shechmeister *et al.*, 1962). Other investigations have been conducted on the effects of radiation on the ability of fish to osmoregulate following acute radiation exposure (Conte, 1965; Angelovic *et al.*, 1969). In comparison few examples are available to characterize the physiological effects of acute irradiation on invertebrates. Engel *et al.* (1973) and Engel and Shelton (1980) demonstrated that radiation caused changes in the osmoregulatory abilities of euryhaline estuarine crustaceans, the blue crab and grass shrimp. In both cases radiation caused disruption of isosmotic intracellular regulation that is mediated by free amino acids. These changes can be verified statistically but are difficult to interpret because failure of the osmoregulatory system results in death. Therefore, the response is usually all or none.

2.3 Developmental Effects

The effects of acute irradiation on the development of fish and invertebrate eggs have been reviewed by Anderson and Harrison (1986). A partial compilation of sensitivities in normal development due to irradiation are shown in Table 2.2.

TABLE 2.2—Range of sensitivities of the early developmental stages of both fish and invertebrates to acute exposures^a

DEVELOPMENTAL STAGE	RADIATION DOSE				
Gy	0.01	0.1	1.0	10.0	100.0
Fish					
Unfertilized Ovum		<-->			
One-cell stage		<----->			
32-cell stage to late embryo			<----->		
Invertebrates					
Dormant eggs				<----->	
Developing embryos				<----->	

^aThe data used to generate these sensitivity ranges can be found in NRCC, 1983 and Anderson and Harrison, 1986.

According to a report by Donaldson and Foster (1957) developing salmon embryos could be disrupted by doses as low as 0.15 Gy of x radiation.¹ The 0.15 Gy value comes from the work of Bonham and Welander (1963) and is also quoted earlier as unpublished data in a review by Donaldson and Foster (1957). This particular determination for an LD₅₀ at 150 days is significantly lower than other published values for fish. Frank (1971) also demonstrated with carp eggs that "radiosensitivity decreased as development increased after organogenesis, but sensitivity of early stages did not decrease in chronological order. Stages in decreasing order of sensitivity were newly-fertilized eggs, early gastrulation, early cleavage, and post organogenesis." Similar results were obtained for the Atlantic silverside *Menidia menidia* and the silver salmon *Oncorhynchus kisutch* where the newly-fertilized ova were more radiosensitive than those that were irradiated later in development (Bonham and Welander, 1963; Engel *et al.*, 1965). With invertebrates the doses of radiation necessary to significantly alter development were generally higher than those for fish by about an order of magnitude (Table 2.2 and Blaylock and Trabalka, 1978). When newly-hatched brine shrimp *Artemia salina* nauplii were irradiated with a single sublethal dose of 5 Gy, the irradiated animals grew faster and larger and reached reproductive age sooner than the unirradiated control animals (White *et al.*, 1967). The literature reviewed on the effects of radiation on developing aquatic animals shows that the early period in the life cycle is the most radiosensitive.

2.4 Reproductive Effects

The effects of radiation on reproductive potential have always been one of the primary concerns with environmental radiation exposures to populations of aquatic organisms. In determining the effects of radiation on aquatic organisms, it is important to know the stages of the germ cells that are being irradiated. However, in most invertebrates the gonadal division rates and maturation are not synchronous. This will inevitably increase the variability in gamete viability data. Acute doses >10 Gy cause irreversable changes in the gonads and permanent sterility in fish (Egami and Ijiri, 1979). Hyodo-Taguchi (1980) demonstrated that a dose of 5 Gy to male medaka *Oryzias latipes* caused a decrease in mating success and an increase

¹For the conversion of R units of exposure in the published literature to SI units of absorbed dose for this report, 1 Gy = 100 rad \approx 105 R.

in infertile/unfertilized eggs, and some sterility. Welander et al. (1948) and Konno (1980) using chinook salmon *Oncorhynchus tshawytscha* and rainbow trout *Salmo gairdneri*, showed that 2.4 Gy significantly reduced the number of female germ cells in irradiated salmon and that 6 Gy caused sterility in rainbow trout.

The invertebrate germ cells appear to be less sensitive to radiation than those of vertebrates. Hoppenheit (1973) showed that a radiation dose of 2.1 Gy reduced egg production rates in adult gammarid amphipods. Anderson and Harrison (1986) showed that 1 and 5 Gy reduced brood size in adult polychaete worms *Neanthes arenaceodentata* but that the juveniles required a dose of 5 to 10 Gy to produce the same response, suggesting that repopulation of the gonads by undamaged cells may occur. In the freshwater snail *Physa acuta* a dose of 20 Gy reduced fecundity and fertility, while 1,000 Gy eliminated their reproductive capacity (Ravera, 1966).

2.5 Environmental Factors

In considering the potential impacts of ionizing radiations on aquatic organisms, the physical and chemical characteristics of the environment in which the organism lives must be considered. Environmental factors that classically have been considered are salinity and temperature, but in most aquatic systems, the presence of anthropogenic contaminants may also have substantial effects on survival and species diversity. Interactions among radiation, salinity, and temperature have been documented (Rice and Baptist, 1974; NRCC, 1983; Anderson and Harrison, 1986). Temperature alone can modify radiation responses in aquatic organisms. Hyodo (1965a, 1965b) and Buhringer (1970) showed that after acute lethal exposures to radiation, low temperature (4–6° C) prolonged survival of freshwater carp and goldfish, but if the temperature was increased to ambient, the radiation response developed at the normal rate and produced the same endpoint.

Salinity also can modify radiation response among fish and invertebrates, particularly those that are from either estuarine or marine environments (Ophel *et al.*, 1976; Engel, 1979; NRCC, 1983). The salinity effects that have been demonstrated in both fish and invertebrates have been concerned with radiation damage to the osmoregulatory systems. Although these effects can be measured, as noted earlier, interpretation is difficult because osmoregulation is a critical physiological function, and when disrupted will result in death.

In freshwater systems, it is necessary to consider water chemistry and quality as important parameters. In freshwaters where trace

element concentrations are low, the accumulation of radionuclides by aquatic organisms may be affected by the competition between chemically similar stable and radioactive isotopes. For example, the uptake of ^{137}Cs by fish can be influenced significantly by the potassium concentration in the water (NCRP, 1984). This same type of relationship was observed in the euryhaline clam *Rangia cuneata* in a coastal river (Wolfe, 1967, 1971). The ^{137}Cs concentration decreases in the clams along the salinity gradient from freshwater to brackish as the salinity and the potassium concentration in the water increased. Two other elements that should be considered are calcium and strontium (Templeton and Brown, 1964). The uptake of ^{90}Sr appears to be influenced by water hardness (i.e., calcium concentration), and the same type of gradient can be expected in a transition from freshwater to seawater. Therefore, not only is the water chemistry important in the interpretation of radiation exposure and effects data, but it is also of prime importance in understanding the partitioning and accumulation of radionuclides by aquatic organisms.

2.6 Summary

Experiments have demonstrated that adult fish have radiation sensitivities similar to terrestrial mammals if the response is followed for sufficient time, while invertebrates tend to be more resistant. The most sensitive periods in the life cycle of aquatic organisms are the early developmental stages, and generally radiation sensitivity decreases with increasing time of development.

Radiation responses of aquatic organisms can be modified by alterations of their environment before, during, and after irradiation (i.e., salinity, temperature, water chemistry, and contaminant load). Also, any factor that affects metabolic rate will alter the rate and intensity of the radiation response of the organism.

Acute radiation exposure studies are not particularly relevant to the prediction of population effects in normal environmental situations because the high doses required to produce an effect greatly exceed those that would result from controlled discharge of low-level radioactive wastes during routine operations. They do, however, provide some information on relative radiation sensitivities and potential interactions between radiation and environmental factors which cannot be studied effectively under experimental conditions of chronic exposure.

3. Effects Due To Chronic Exposure

Radionuclides released from nuclear fuel cycles can become incorporated in the biogeochemical cycles of freshwater systems, having entered these systems in direct liquid discharges and/or through secondary processes—erosion, runoff, and groundwater infiltration from landscapes. Many radionuclides concentrate in both biotic and abiotic compartments (e.g., mineralized tissues and bottom sediments) enhancing exposures of aquatic organisms to chronic irradiation from both external and internal sources. The latter include contaminated food organisms and sediment in transit through the digestive tract, as well as radionuclides incorporated into organs and tissues.

Higher total doses of radiation are required to produce mortality or other injuries when organisms are chronically exposed rather than acutely exposed. The existence of repair mechanisms makes it possible to define a chronic exposure rate at which no significant effects should result. In the ideal, studies of natural populations that have been exposed to chronic low-level radiation for many generations should provide the information necessary to set standards for adequate protection of aquatic organisms.

Effects at low levels of irradiation are difficult to detect, however, especially in natural populations where environmental factors and changes in population dynamics may overshadow the subtle effects of radiation. Further, less than 10 percent of the published studies that deal with aquatic organisms were based on chronic or continuous radiation exposure; only a small fraction of these involved natural populations. A major concern is the accurate determination of radiation doses to organs, tissues, or the whole organism in natural populations exposed to multiple sources of irradiation, the relation of these doses to potential effects, and, then, the interpretation of the impact of these effects on natural populations. Thus, it is necessary to review a wide variety of studies on chronic irradiation of aquatic organisms in order to address standard setting for environmental exposures.

3.1 Mortality

Effects on mortality of fish and invertebrates from chronic radiation exposure are presented in Tables 3.1 and 3.2, respectively. Deleterious effects on survival have not been reported at dose rates <100 mGy d⁻¹ in carefully designed experiments conducted under controlled conditions (also see discussion on fish embryo studies in Section 3.4).

Fish

Donaldson and Bonham (1964) did not observe significant mortality in chinook salmon *Oncorhynchus tshawytscha*, irradiated with

TABLE 3.1—Mortality in fish from chronic exposure to radiation under laboratory conditions^a

Organism and Life stage exposed	Exposure regime	Observations	Reference
Fish			
<i>Oncorhynchus tshawytscha</i> embryos and alevins (chinook salmon)	5.1 mGy d ⁻¹ for 61–69 d (0.31–0.35 Gy) ^b	No excess mortality up to release of smolts, and greater return of females to spawn	Donaldson and Bonham, 1964, 1970; Woodhead, 1984
<i>Oncorhynchus tshawytscha</i> embryos and alevins (chinook salmon)	5.0–475 mGy d ⁻¹ for 71–86 d (0.38–38 Gy) ^b	Lower return of spawning adults (released as smolts) at dose rates ≥ 95 mGy d	Hershberger <i>et al.</i> 1978; Woodhead, 1984
<i>Poecilia reticulata</i> 0.3-d neonates to adult (guppy)	40.8–305 mGy d ⁻¹ up to 988 d ^c	No effect on survival to maturity of offspring	Woodhead, 1977
<i>Poecilia reticulata</i> embryos (guppy)	Tritiated H ₂ O: 0.93–18.5 GBq L ⁻¹ ; 50–1,000 mGy d ⁻¹ for 17 d	No excess mortality	Erickson, 1973
1-week-old juveniles	1.85–37 GBq L ⁻¹ ; 100–2,100 mGy d ⁻¹ for 21–30 d	No excess mortality	Erickson, 1973

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad \approx 105R; 1 Bq = 2.7×10^{-11} Ci.

^b⁶⁰Co gamma radiation.

^c¹³⁷Cs gamma radiation.

TABLE 3.2—*Mortality in invertebrates from chronic exposure to ^{60}Co gamma radiation under laboratory conditions^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
Freshwater Invertebrates			
<i>Daphnia pulex</i> all life stages (cladoceran)	217–721 mGy h ⁻¹ for 19 h d ⁻¹ ; 4,119–13,700 mGy d ⁻¹ for 20–35 d	Increased mortality rate (population) at exposure rates ≥11,500 mGy d ⁻¹	Marshall, 1962
<i>Physa heterostropha</i> adults (pond snail)	240–6,000 mGy d ⁻¹ for 168 d	Decreased survival at dose rates ≥2,400 mGy d ⁻¹	Cooley and Miller, 1971
Marine Invertebrates			
<i>Callinectes sapidus</i> Juveniles (blue crab)	770–6,960 mGy d ⁻¹ for 70 d	Lowered survival at highest dose rate	Engel, 1967
<i>Argopecten irradians</i> juveniles (scallop)	1.4–8,900 mGy d ⁻¹ for 84 d	No deleterious effects	Baptist <i>et al.</i> , 1976
<i>Mercenaria mercenaria</i> juveniles (clam)	1.4–8,900 mGy d ⁻¹ for 426 d	Lowered survival at dose rates of 3,800–8,900 mGy d ⁻¹ only	Baptist <i>et al.</i> , 1976

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad ≈ 105R.

5.1 mGy d⁻¹ as embryos and alevins, up to the time of release of smolts to the environment from the University of Washington rearing facility. Irradiated females returned to this site to spawn in greater numbers than controls, producing a larger number of viable eggs (Donaldson and Bonham, 1970). In later studies (Hershberger *et al.*, 1978; Woodhead, 1984), lower returns of spawning adults were observed after exposures to ≥95 mGy d⁻¹. No effect on survival of offspring of the guppy *Poecilia reticulata* occurred when fish were subjected to lifetime exposures of 40.8–305 mGy d⁻¹ (Woodhead 1977). No increased mortality was observed after 90 d when embryos and 1-week-old juveniles of the guppy *Poecilia reticulata* were exposed to tritiated water concentrations of 0.93–18.5 GBq L⁻¹, (50–1,000 mGy d⁻¹ for 17 d) and 1.85–37.0 GBq L⁻¹, (100–2,100 mGy d⁻¹ for 21–30d), respectively (Erickson, 1973).

Invertebrates

Marshall (1962) observed a significantly increased population mortality rate in *Daphnia pulex* only at dose rates ≥11,600 mGy d⁻¹.

Survivorship of snails *Physa heterostroph*a exposed to dose rates 240–6,000 mGy d⁻¹ for 168 d was reduced at $\geq 2,400$ mGy d⁻¹ (Cooley and Miller 1971). Juvenile blue crabs *Callinectes sapidus* exposed for 70 d exhibited decreased survival only at dose rates $\geq 6,960$ mGy d⁻¹ (Engel 1967). Juveniles of the clam *Mercenaria mercenaria* and the scallop *Argopecten irradians* were subjected to dose rates ranging from 1.4–8,900 mGy d⁻¹ (Baptist et al., 1976). After a 14-month exposure, clams showed reduced growth and survival at the highest dose rate only (3,800–8,900 mGy d⁻¹); deleterious effects on scallops, exposed for a shorter period of time (3 months), were not observed.

3.2 Physiological Effects

Research characterizing physiological effects of chronic irradiation on fish and invertebrates is presented in Table 3.3. Some effects on the immune system and spermatogenesis appear to have been detected at dose rates on the order of 10 mGy d⁻¹.

Antibody synthesis against the disease bacterium *Chondrococcus columnaris* was reduced, relative to controls, in juvenile and yearling rainbow trout *Salmo gairdnerii* exposed for 20 days as embryos to tritiated water concentrations of 37 and 370 MBq L⁻¹ (2 and 20 mGy d⁻¹), suggesting a depressed immune response (Strand et al., 1973a). Mild atrophy of hemopoietic activity in kidney and spleen was recorded in some mosquitofish *Gambusia affinis* irradiated at 360–720 mGy d⁻¹ for 128 d (Cosgrove et al. 1975). No damage was demonstrable when *Gambusia* were exposed to dose rates of 120–1300 mGy d⁻¹ for a shorter period (37 d).

Testicular atrophy was the only histopathological effect observed when *Gambusia* were irradiated at 343–1300 mGy d⁻¹ for 47 d (Cosgrove and Blaylock, 1973). In guppies *Poecilia reticulata*, irradiated from birth at 40.8–305 mGy d⁻¹, deleterious effects on oogenesis were observed at all dose rates, appearing progressively earlier with increasing dose rates, but effects on male germ cells were observed only at the highest dose rate (Woodhead, 1977). However, the total effect on fecundity at 40.8–96.0 mGy d⁻¹ was more severe than that indicated by results from histological studies of the gonads, suggesting that effects on the pituitary function may have been involved. Marked depletion in the numbers of primary spermatogonia (1b) resulted when male medaka *Oryzias latipes* received dose rates ≥ 100 mGy d⁻¹ (Hyodo-Taguchi and Egami, 1977; Hyodo-Taguchi, 1980). Temporary, early reductions in primary spermatogonia occurred at dose rates as low as 10 mGy d⁻¹ in fish exposed to

TABLE 3.3—*Physiological change in fish and invertebrates from chronic exposure to radiation under laboratory conditions^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
Fish			
<i>Salmo gairdnerii</i> embryos (rainbow trout)	Tritiated H ₂ O: 37 and 370 MBq L ⁻¹ for 20 d; 2 and 20 mGy d ⁻¹	Lowered antibodies to <i>Chondrococcus columaris</i> disease in juveniles/ yearlings, indicating immune response depression	Strand <i>et al.</i> , 1973a
<i>Gambusia affinis</i> adults (mosquitofish)	120–130 mGy d ⁻¹ up to 128 d ^b	No hemopoietic damage after 37 d; mild hemopoietic atrophy in kidney and spleen in some fish after 128 d at 360 or 720 mGy d ⁻¹	Cosgrove <i>et al.</i> , 1975
<i>Gambusia affinis</i> adults (mosquitofish)	312–1300 mGy d ⁻¹ 47 d ^b	Testis atrophy at all dose rates; damage to other tissues/organs not observed	Cosgrove and Blaylock, 1973
<i>Poecilia reticulata</i> 0–3 d neonates to adult (guppy)	40.8–305 mGy d ⁻¹ , up to 974 d ^c	Oogenesis affected at all dose rates; damage appears earlier as dose rate increases	Woodhead, 1977
<i>Oryzias latipes</i> adult males (medaka)	Tritiated H ₂ O: 0.19–3.7 GBq L ⁻¹ , (10–210 mGy d ⁻¹) for 30 d	Severe depletion of spermatogonia 1b at ≥100 mGy d ⁻¹ ; temporary, early (5-d) depletion at lower dose rates	Hyodo-Taguchi and Egami, 1977
<i>Oryzias latipes</i> adult males (medaka)	12–800 mGy d ⁻¹ for 120 d ^c	Clear depletion of spermatogonia 1b at ≥148 mGy d ⁻¹ ; temporary, early (10-d) depletion at 28 and 65 mGy d ⁻¹	Hyodo-Taguchi, 1980
<i>Ameioba splendens</i> young adults	185 mGy d ⁻¹ up to 244 d ^c	Spermatogenesis more radiosensitive than oogenesis	Rackham and Woodhead, 1984

TABLE 3.3—*Continued*

Organism and Life stage exposed	Exposure regime	Observations	Reference
Invertebrates			
<i>Physa heterostropha</i> adults (pond snail)	240–1,200 mGy d ⁻¹ for 98 d ^b	Partial atrophy of gonad in some snails at 1,200 mGy d ⁻¹ , but not at 240–480 mGy d ⁻¹	Cooley, 1973a

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad \approx 105R; 1 Bq = 2.7×10^{-11} Ci.

^b⁶⁰Co gamma radiation.

^c¹³⁷Cs gamma radiation.

tritiated water and 28 mGy d⁻¹ in fish exposed to gamma radiation. Spermatogenesis was more sensitive than oogenesis in adult *Ameca splendens*, irradiated at 185 mGy d⁻¹, but both sexes eventually became sterile (Rackham and Woodhead, 1984). Gonadal atrophy was observed in some snails *Physa heterostropha* after 98-d exposures to ⁶⁰Co gamma radiation at 1,200 mGy d⁻¹, but not at lower dose rates of 240–480 mGy d⁻¹ (Cooley, 1973a).

3.3 Reproductive Effects

Effects of chronic radiation exposure on fertility of aquatic invertebrates and fish have been observed at dose rates <100 mGy d⁻¹ (Table 3.4—natural populations and Tables 3.5 and 3.6—experimental conditions: fish and invertebrates, respectively).

Studies of Natural Populations

Both a significantly higher fecundity and frequency of nonviable embryos have been observed in natural populations of mosquitofish *Gambusia affinis* from radionuclide-contaminated White Oak Lake at the Oak Ridge National Laboratory, in 1966, 1967, 1973, and 1978 (Blaylock, 1969; Trabalka and Allen, 1977; Blaylock and Frank, 1980). By 1973, this population had been exposed to environmental radiation for approximately 60 generations and had received a cumulative dose estimated to be in excess of 20 Gy (Trabalka and Allen, 1977). The dose rate to these *Gambusia* was estimated to be ≥ 10 mGy d⁻¹ in the early 1960's, falling to 3.5, 1.8, and 0.6

TABLE 3.4—*Reproductive effects in fish and invertebrates from natural populations exposed to chronic irradiation^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
Fish			
<i>Gambusia affinis</i> all life stages (mosquitofish)	White Oak Lake, Oak Ridge, TN, contaminated since 1943; 1960 dose rate ≥ 10 mGy d ⁻¹ falling to 3.5, 1.8, and 0.6 mGy d ⁻¹ in 1965, 1971, & 1975, respectively	Greater brood size and embryo mortality in fish collected directly from White Oak Lake in 1965, 1967, 1973, & 1978	Blaylock, 1969; Trabalka and Allen, 1977; Blaylock and Frank, 1980
		Higher fecundity absent in laboratory-reared F ₁ fish, but higher embryo mortality persists; indicates that effects (1) on fecundity may be an artifact and (2) on embryo mortality a product of radiation-induced genetic load	Trabalka and Allen, 1977
<i>Rutilus rutilus</i> all life stages (roach)	Ural Lake (USSR) contaminated in the 1950s; early 1970s dose rate 7–15 mGy d ⁻¹ , ⁹⁰ Sr and ¹³⁷ Cs in lake water at 5.9 and 0.14 kBq L ⁻¹ , respectively	Lowered fecundity and delay in spawning; questionable controls and aberrant chemistry of system confound data interpretation	Voronina <i>et al.</i> , 1974; Peshkov <i>et al.</i> , 1978
Invertebrates			
<i>Physa heterostropha</i> all stages (pond snail)	White Oak Lake; 6.5 mGy d ⁻¹	Egg-capsule production reduced, but with more eggs/capsule, resulting in egg production rate comparable to controls	Cooley, 1973b

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad \approx 105R; 1 Bq = 2.7×10^{-11} Ci.

TABLE 3.5—*Reproductive effects in fish from chronic exposure to radiation under laboratory conditions^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
Fish			
<i>Oncorhynchus tshawytscha</i> embryos (chinook salmon)	5.0–475 mGy d ⁻¹ for 80 d ^b	Gonadal development of smolts was retarded by ≥ 95 mGy d ⁻¹ delivered to embryos	Bonham and Donaldson, 1972
<i>Poecilia reticulata</i> embryos (guppy)	Tritiated H ₂ O: 1.9–17.4 GBq L ⁻¹ ; 100–410 mGy d ⁻¹ for 17 d	Male courting activity reduced after exposure to 4.0 mGy d ⁻¹ as embryos	Erickson, 1973
<i>Poecilia reticulata</i> 0–3-d neonates to adult (guppy)	40.8–305 mGy d ⁻¹ , for varied periods ^c	½ and ⅓ pairs, respectively, became infertile at 40.8 and 96 mGy d ⁻¹ , but fecundity was unaffected in pairs remaining fertile; total sterility at a dose rate of 305 mGy d ⁻¹	Purdom and Woodhead, 1973
<i>Poecilia reticulata</i> 0–3-d neonates to adult (guppy)	40.8–305 mGy d ⁻¹ , up to 988 d ^c	Fecundity 57, 52, and 3.5 percent of controls at 40.8, 96.0, and 305 mGy d ⁻¹ , respectively; lower brood size and increased sterility at all dose rates	Woodhead, 1977
<i>Oryzias latipes</i> adult males (medaka)	12–800 mGy d ⁻¹ for 60–120 d ^c	Dose rates ≥ 65 mGy d ⁻¹ for 60 d increased sterility and number of unfertilized eggs (from matings with unirradiated females)	Hyodo-Taguchi, 1980
<i>Ameioba splendens</i> young adults	185 mGy d ⁻¹ , up to 244 d ^c	Complete sterility after 190 d; offspring, born at 75 d, were sterile at maturity	Rackham and Woodhead, 1984

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad \approx 105R; 1 Bq = 2.7×10^{-11} Ci.

^b⁶⁰Co gamma radiation.

^c¹³⁷Cs gamma radiation.

TABLE 3.6—*Reproductive effects in invertebrates from chronic exposure to ^{60}Co gamma radiation under laboratory conditions^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
<i>Daphnia pulex</i> all life stages (cladoceran)	217–721 mGy h ⁻¹ for 19 h d ⁻¹ ; 4110–13,700 mGy d ⁻¹ for 20–35 d	Decreased population birth rate at dose rates ≥4,610 mGy d ⁻¹	Marshall, 1962
<i>Physa heterostropha</i> adults (pond snail)	240–6,000 mGy d ⁻¹ for 168 d	Egg and egg- capsule production reduced progressively over the range 480–6,000 mGy d ⁻¹ and stopped at 6,000 mGy d ⁻¹ ; 25°C temperature	Cooley and Miller, 1971
<i>Physa heterostropha</i> adults (pond snail)	240–1,200 mGy d ⁻¹ for 98 d	Egg production reduced progressively over range 240–1,200 mGy d ⁻¹ ; egg- capsule production reduced at ≥240 mGy d ⁻¹ at 25°C, but only at 1,200 mGy d ⁻¹ at 15°C	Cooley, 1973a

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad ≈ 95R.

mGy d⁻¹ by 1965, 1971, and 1975, respectively. Offspring of *Gambusia*, collected in 1973 and reared to maturity under laboratory conditions along with four control populations, exhibited the higher embryo mortality, but not the higher fecundity, characteristic of the field population. This indicated that the increased fecundity of *Gambusia* inhabiting White Oak Lake was probably associated with the high productivity of this eutrophic ecosystem, and not with effects of chronic radiation exposure. The decreased embryo viability may well be attributable to a genetic load of radiation-induced, recessive lethal mutations, but the overall fecundity and fitness of the *Gambusia* have not been consequently reduced (Trabalka and Allen, 1977).

The fecundity of Siberian roach *Rutilus rutilus lacustris*, collected from a lake contaminated with radioactivity for approximately 15 years (Trabalka *et al.*, 1980a), was lower than that of controls (one local population from Cheliabinsk Province in the Urals and literature sources for four other geographic zones of the Soviet Union) (Voronina *et al.* 1974; Peshkov *et al.* 1978). The time of spawning was also delayed relative to the local control population, and occurred

at water temperatures 3–4°C higher. The dose rate to the roach at the time of the study was estimated at 7–15 mGy d⁻¹, and the lake water contained ⁹⁰Sr and ¹³⁷Cs at concentrations of 5.9 kBq L⁻¹ and 0.14 kBq L⁻¹, respectively (Peshkov *et al.*, 1978). Since ¹³⁷Cs sorbed on bottom sediments contributed 33–67 percent of the dose rate and its concentration in lake water was 26 percent of the Soviet drinking water limit for humans, it was concluded that ecological considerations should play a role in standard setting—a view not shared by others (Pitkayanen *et al.*, 1978; Pechkurenkov and Pokrovskaya, 1978). It is questionable whether suitable controls were available for this study because the water chemistry of the contaminated lake is clearly aberrant, e.g., calcium concentration 226 mg L⁻¹ (Balabanova, 1957; Trabalka, 1981). The association of this site with nuclear facilities also means that other confounding factors may have been present, i.e., thermal and/or hazardous chemical effluents. “*There never has been an ecological study of [this lake], and consequently it is hardly justifiable to attribute everything to one factor*” (Pechkurenkov and Pokrovskaya, 1978).

A natural population of the snail *Physa heterostrophica* from White Oak Lake, exposed to an estimated 6.5 mGy d⁻¹, exhibited both a lower frequency of egg capsule production and an increased number of eggs per capsule (Cooley and Nelson, 1970; Cooley, 1973b). As a result, the overall fecundity of the population was similar to the control.

Laboratory Studies

Exposure rates ≥ 95 mGy d⁻¹ during the 80 d incubation period of chinook salmon *Oncorhynchus tshawytscha* markedly retarded gonadal development in the smolts (Bonham and Donaldson, 1972; Woodhead, 1984). Male courting activity in the guppy *Poecilia reticulata* was reduced after embryonic exposure to 410 mGy d⁻¹ from tritiated water (Erickson, 1973). Fecundity of the guppy, exposed to dose rates of 40.8, 96.0 and 305 mGy d⁻¹ from the 0–3 d neonatal stage to adulthood, was reduced at all dose rates because of decreased mean brood size and increased frequency of sterile adults (Woodhead, 1977). In an earlier study conducted using the same dose regimes (Purdum and Woodhead, 1973), an increased frequency of sterility was evident at the median rate, becoming complete at the highest dose rate, but no reduction in fecundity was observed in the adults which remained fertile in the low and median dose-rate regimes. Significant increases in the proportion of sterile individuals and in the percentage of unfertilized eggs (from matings with unirradiated

females) were observed after adult male medaka *Oryzias latipes* had been irradiated at dose rates ≥ 65 mGy d⁻¹ for 60 d (Hyodo-Taguchi, 1980).

The birth rate of *Daphnia pulex* populations fell sharply in the exposure range 4,280- 9,500 mGy d⁻¹ from ⁶⁰Co gamma radiation, whereas the death rate was barely affected (Marshall, 1962; Table 3.2). Significant reductions in the population birth rate occurred at exposure dose rates $\geq 4,610$ mGy d⁻¹. Observed reductions in the intrinsic rate of increase in the *Daphnia* populations were thus due almost entirely to changes in the birth rate. Snails *Physa heterostropha* irradiated in the laboratory under a constant-temperature regime of 25°C throughout their life-span showed significant reproductive effects at dose rates $\geq 2,400$ mGy d⁻¹, but not at 240 mGy d⁻¹ (Cooley and Miller, 1971). In a later study, significant effects occurred at dose rates ≥ 240 mGy d⁻¹ when snails were irradiated at 25°C (Cooley, 1973a). However, egg-capsule production was unaffected at 240-480 mGy d⁻¹ in *Physa* irradiated at 15°C, even though total egg production was markedly reduced at dose rates ≥ 240 mGy d⁻¹.

3.4 Effects on Growth and Development

Laboratory Studies

Effects of Radionuclides on Fish Embryos

Studies reporting developmental effects on fish embryos from radionuclides introduced into the incubation medium are quite prevalent in the literature but have contributed little to the understanding of chronic radiation effects (Woodhead, 1984). These studies have provoked considerable debate, leading to intense scrutiny in reviews of radiation effects on aquatic organisms (IAEA, 1976; Blaylock and Trabalka, 1978; NRCC, 1983; Woodhead, 1984; Anderson and Harrison, 1986). The reader is referred to the comprehensive treatment by Woodhead (1984) for specific details. For our purposes, here it is sufficient to note that these studies are replete with methodological flaws (exceptions noted by Woodhead, 1984), including failure to:

- (1) utilize a sufficiently wide range of radionuclide concentrations to construct a *complete* dose-effect curve (0-100 percent response),
- (2) provide sufficient replicates to permit valid statistical comparisons,

- (3) randomize treatment groups of embryos to correct for differential viability of gametes between spawning individuals and as a function of the time of release from individuals (e.g., Trabalka and Burch, 1978), and
- (4) estimate the absorbed dose to the embryos as a function of time, space, and radionuclide concentration (some ^3H studies excepted).

Woodhead (1984) concluded that:

*"If these [inconsistent] data [on embryo hatching/survival] are excluded then the remaining doses (besides the no-observed-effect level of 86 Gy reported by Till [1976, 1978]) at which an effect has been observed are [21.6 Gy] (Ichikawa and Suyama, 1974) and [7.5 Gy] (Fedorova, 1972). Since Federova (1972) only gives data for a single concentration of ^{14}C , it is not possible to conclude that [7.5 Gy] represents the lower limit for a significant effect on the hatching success of *Coregonus peled*. From the data given the dose to the embryo could be as much as [11.3 Gy]. Federova's data also provide confirmation that a significant part of the total mortality is caused by a small fraction of the total dose delivered in the early part of embryogenesis, and also that the embryo becomes less sensitive during the course of its development.*

There are only 2 sets of [potentially usable] data on the incidence of embryo abnormality (Strand et al. 1973b; Till 1976, 1978). The data of Strand et al. (1973b) can only be accepted with reservations due to the clear lack of consistency within and between the 2 experiments reported; the embryos receiving the highest absorbed dose yielded a lower proportion of anomalous individuals than any other treatment, including the 2 control populations. Till (1976, 1978) observed that the lowest dose at which a significantly increased proportion of abnormal embryos was produced was [34 Gy]."

Effects of Exposure to External Radiation or Tritium

Other results on growth and developmental effects of chronic irradiation in aquatic organisms, obtained under laboratory conditions, are presented in Tables 3.7 (fish) and 3.8 (invertebrates). Because of concerns about methodological problems of the type described in the preceding Section, particularly dosimetry (Anderson and Harrison, 1986), data from studies on invertebrates exposed to radionuclides in water have not been included (partial review by Blaylock and Trabalka, 1978); the few studies with tritiated water (Nelson 1973;

TABLE 3.7—*Developmental effects in fish from chronic exposure to radiation under laboratory conditions^a*

Organism and Life stage exposed	Exposure regime	Observations	Reference
<i>Oncorhynchus kisutch</i> embryos and alevins (coho salmon)	4.2 mGy d ⁻¹ for 91 d (0.38 Gy) ^b	Increased number of opercular defects	Donaldson and Bonham, 1964
<i>Oncorhynchus tshawytscha</i> embryos (chinook salmon)	5.1 mGy d ⁻¹ for 61–69 d (0.31–0.35 Gy) ^b	Increased weight of smolts; inconclusive results on F ₁ generations from returned fish (released as smolts)	Donaldson and Bonham, 1964, 1970;
<i>Oncorhynchus tshawytscha</i> embryos (chinook salmon)	5.0–475 mGy d ⁻¹ for 71–86 d (0.38–38 Gy) ^b	Lower rate of growth (smolts) at dose rates ≥95 mGy d ⁻¹	Hershberger <i>et al.</i> , 1978;
<i>Poecilia reticulata</i> embryos (guppy)	Tritiated H ₂ O: 1.85–18.5 GBq L ⁻¹ ; 100–1,000 mGy d ⁻¹ for 17 d	No consistent pattern of effects on growth and development, but males exposed to 18.5 GBq L ⁻¹ were twice the weight of controls at 21 weeks of age	Erickson, 1973
1-week-old juveniles	18.5–37 GBq L ⁻¹ ; 1,000–2,100 mGy d ⁻¹ for 21–30 d	No consistent pattern of effects on growth and development	Erickson, 1973
<i>Salmo gairdneri</i> embryos (rainbow trout)	Tritiated H ₂ O: 0.37 GBq L ⁻¹ ; 20 mGy d ⁻¹ for 20 d (0.4 Gy)	No effect on growth of larvae by end of 149-d observation period	Strand <i>et al.</i> , 1973b
<i>Gasterosteus aculeatus</i> embryos (stickieback)	Tritiated H ₂ O: 18.5–74 GBq L ⁻¹ ; 1,000–4,100 mGy d ⁻¹ for 7d	Significant reduction in mean eye diameter at 37 and 74 GBq L ⁻¹	Walden, 1973

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad ≈ 105R; 1 Bq = 2.7 × 10⁻¹¹ Ci.

^b⁶⁰Co gamma radiation.

TABLE 3.8—Effects on growth and development of invertebrates from chronic exposure to ^{60}Co gamma radiation under laboratory conditions^a

Organism and Life stage exposed	Exposure regime	Observations	Reference
Freshwater Invertebrates			
<i>Daphnia pulex</i> all life stages (cladoceran)	217–721 mGy h ⁻¹ for 19 h d ⁻¹ ; 4,110–13,700 mGy d ⁻¹ for 20–35 d	Lowered population increase rate only at dose rates $\geq 4,610$ mGy d ⁻¹ in absence of competition for food; population extinction at 12,600 mGy d ⁻¹	Marshall, 1962
<i>Daphnia pulex</i> all life stages (cladoceran)	44–265 mGy h ⁻¹ for 18.5 h d ⁻¹ ; 810–4,900 mGy d ⁻¹ for 385 d	Extinction at 3,940 mGy d ⁻¹ in food-limited populations; net production unaffected at lower dose rates	Marshall, 1966
<i>Daphnia pulex</i> all life stages (cladoceran)	2.0–360 mGy h ⁻¹ for 18 h d ⁻¹ ; 35–6,500 mGy d ⁻¹ for 392 d	Cropping ameliorates radiation effects; dose rate tolerated increases to 5,990 mGy d ⁻¹ in food-limited populations	Marshall, 1967
<i>Physa heterostroph</i> adults (pond snail)	240–6,000 mGy d ⁻¹ for 168 d	Reduced egg hatching at 2,400 mGy d ⁻¹ and no hatching at 6,000 mGy d ⁻¹ ; lowered adult growth rate at $\geq 2,400$ mGy d ⁻¹	Cooley and Miller, 1971
Marine Invertebrates			
<i>Callinectes sapidus</i> juveniles (blue crab)	770–6,960 mGy d ⁻¹ for 70 d	Lowered growth rate at highest dose rate only; increase growth rate in 770 mGy d ⁻¹ exposure group	Engel, 1967
<i>Argopecten irradians</i> juveniles (scallop)	1.4–8,900 mGy d ⁻¹ for 3 mo	No deleterious effects	Baptist, <i>et al.</i> , 1976
<i>Mercenaria mercenaria</i> juveniles (clam)	1.4–8,900 mGy d ⁻¹ for 14 mo	deleterious effects only at 3,800–8,900 mGy d ⁻¹ dose rate	Baptist <i>et al.</i> , 1976

^aFor this report, conversions are taken to be given by: 1 Gy = 100 rad \approx 105R.

Abbott and Mix, 1979) have generated data of limited value for the current analysis.

Exposure of coho salmon *Oncorhynchus kisutch* to 4.2 mGy d^{-1} during embryogenesis produced an increased incidence of opercular defects in smolts. However, chinook salmon *Oncorhynchus tshawytscha* exposed to 5.1 mGy d^{-1} had higher body weights at the time of their release to the environment from the University of Washington rearing facility (Donaldson and Bonham, 1964). Measurements of growth rates in smolts from various F_1 generation crosses between control and irradiated fish were inconclusive (Donaldson and Bonham, 1970; Woodhead 1984). Significantly decreased growth rates in chinook salmon smolts subjected to $\geq 95 \text{ mGy d}^{-1}$ as embryos were observed in a later study (Hershberger *et al.*, 1978; Woodhead, 1984), but no growth-stimulating effects occurred at lower dose rates. No consistent pattern of growth or developmental effects were produced by exposing guppy *Poecilia reticulata* embryos or juveniles to tritiated water in the dose rate ranges $100\text{--}1,000 \text{ mGy d}^{-1}$ and $1,000\text{--}2,100 \text{ mGy d}^{-1}$, respectively, but males exposed to the highest dose rate as embryos were nearly double (1.95x) the weight of controls by 21 weeks of age (Erickson, 1973). No effect of embryonic exposure at 20 mGy d^{-1} from tritiated water on growth (length) of rainbow trout *Salmo gairdneri* larvae was found in fish 76–149 d old, even though irradiated fish were significantly longer than controls at 35 d (Strand *et al.*, 1973b). Larval eye diameter in the brook stickleback *Gasterosteus aculeatus* was reduced after total absorbed doses to the embryo $\geq 14 \text{ Gy}$ (Walden, 1973; also see Ichikawa and Suyama, 1974).

Intraspecific competition enhanced the effects of ^{60}Co gamma radiation on *Daphnia pulex* populations (Marshall, 1962; 1966; 1967). In the absence of competition, the intrinsic rate of population increase fell to zero (population extinction) at $12,600 \text{ mGy d}^{-1}$ (Marshall, 1962). In populations self-limited by competition for food, *Daphnia pulex* were able to tolerate much less exposure to chronic irradiation, only $3,940 \text{ mGy d}^{-1}$. However, net biomass production remained constant up to the maximum tolerable dose rate (Marshall, 1966). Cropping, up to 65 percent/week, lessened the effects of chronic irradiation on the populations (not the individuals) and increased the maximum tolerable dose rate to $5,990 \text{ mGy d}^{-1}$ (Marshall, 1967). Freshwater snails *Physa heterostrophus* subjected to lifetime exposures to $240\text{--}6,000 \text{ mGy d}^{-1}$ showed effects on egg hatching and adult growth rates at dose rates $\geq 2,400 \text{ mGy d}^{-1}$ (Cooley and Miller, 1971). No hatching occurred at the highest dose rate employed ($6,000 \text{ mGy d}^{-1}$). Effects on the growth rate of marine invertebrates chronically exposed to ^{60}Co gamma radiation have been observed only at

dose rates $\geq 3,800 \text{ mGy d}^{-1}$ (Engel, 1967; Baptist *et al.*, 1976), but, since lifetime exposures were not employed in these studies, this dose rate is not the threshold for appearance of such radiation effects.

Natural Populations

Studies of chronic irradiation on growth and development in natural populations appear to have been limited to effects on fish (Table 3.9). An increased frequency of abnormalities in embryos of mosquitofish *Gambusia affinis* from White Oak Lake was recorded by Blaylock (1969). The dose rate to *Gambusia* when collected in 1965 has been estimated at 3.5 mGy d^{-1} (Trabalka and Allen, 1977). No deleterious effects on growth rate of adult pike *Esox lucius* (Pitkyanen *et al.*, 1978) or juvenile roach *Rutilus rutilus* (Peshkov *et al.*, 1978),

TABLE 3.9—Effects on development in fish from natural populations exposed to chronic irradiation^a

Organism and Life stage exposed	Exposure regime	Observations	Reference
<i>Gambusia affinis</i> all life stages (mosquitofish)	White Oak Lake, Oak Ridge, TN, contaminated since 1943; 1960 dose rate $\geq 10 \text{ mGy d}^{-1}$ falling to 3.5 mGy d^{-1} in 1965	Increased frequency of abnormal embryos from White Oak Lake fish in 1965	Blaylock, 1969; Trabalka and Allen, 1977
<i>Esox lucius</i> all life stages (pike)	Ural Lake (USSR) contaminated in late 1950s; dose rate in early 1970s 4.5 mGy d^{-1} ; ^{90}Sr and ^{137}Cs in lake water at 3.7 and 0.37 kBq L^{-1} , respectively	No effect on growth rate of adults; selection of controls and limited data about ecosystem confound interpretation of results	Pitkyanen <i>et al.</i> , 1978
<i>Rutilus rutilus</i> all life stages (roach)	Ural Lake (USSR) contaminated in the 1950s; early 1970s dose rate $7\text{--}15 \text{ mGy d}^{-1}$, ^{90}Sr and ^{137}Cs in lake water at 5.9 and 0.14 kBq L^{-1} , respectively	No effect on growth rate of juveniles; questionable controls and aberrant chemistry of system confound data interpretation	Peshkov <i>et al.</i> , 1978

^aFor this report, conversions are taken to be given by: $1 \text{ Gy} = 100 \text{ rad} \approx 105 \text{ R}$; $1 \text{ Bq} = 2.7 \times 10^{-11} \text{ Ci}$.

respectively, occurred in two lakes in the Soviet Urals contaminated with ^{90}Sr and ^{137}Cs . The dose rate to the pike (first lake) was estimated to be 4.5 mGy d^{-1} and to the roach (in the second lake), $7\text{--}15 \text{ mGy d}^{-1}$, at the time of the studies (early 1970s). However, as in the case of White Oak Lake (Section 3.3), dose rates in the Ural lakes were much higher and the spectrum of radionuclides more diverse at the time of contamination (in the late 1950s) (Trabalka et al., 1980a). Consequently, fish populations in all three systems have tolerated much greater earlier exposures to radiation (and, potentially, to other factors; see previous discussion in Section 3.3).

3.5 Summary

Deleterious effects of chronic irradiation have not been observed in natural populations at dose rates $\leq 10 \text{ mGy d}^{-1}$ over the entire history of exposure to ionizing radiation. Radiation exposure regimes at the time that studies were conducted (1960s and 1970s) have sometimes been recorded by reviewers (and authors) without recognition of the potential impacts of earlier exposures to anthropogenic radiation levels orders of magnitude greater (and of the resulting radiation-induced genetic load accumulated). This has led to some confusion about the potential threshold for observation of deleterious effects in natural systems. Although researchers have almost universally recognized the difficulties in obtaining suitable controls for natural population studies, few have been successful in overcoming these problems. In no case, including examples cited from research on White Oak Lake populations, can results be attributed exclusively to effects of ionizing radiation. By the same token, it is not possible to eliminate completely the possibility of deleterious effects on natural populations at dose rates $\leq 10 \text{ mGy d}^{-1}$, given the limited number (and limitations on interpretation) of studies that have been conducted to date.

Taking the combined results from laboratory and field studies, it appears that reproductive and early developmental systems of vertebrates are most sensitive to chronic irradiation, in both aquatic and terrestrial environments (Blaylock and Trabalka, 1978; Egami and Ijiri, 1979; Anderson and Harrison, 1986). Invertebrates appear to be relatively radioresistant (cf Table 3.1 with Table 3.2, and Table 3.7 with Table 3.8; Section 2; Blaylock and Trabalka, 1978; Woodhead, 1984; Anderson and Harrison, 1986). Effects on aquatic organisms, not necessarily detrimental when evaluated in the context of population dynamics (Blaylock and Trabalka, 1978), have been

detected at dose rates in the range 1–10 mGy d⁻¹ (Tables 3.1, 3.3, 3.4, 3.7, and 3.9; note caveats about results on natural populations). Clearly deleterious effects on terrestrial vertebrates which would be detrimental at the population level, e.g., complete sterility in male dogs at 4.0 mGy d⁻¹ after 1 year (Luckey, 1980), appear at similar dose rates (Anderson and Harrison, 1986), but have not been produced in studies of aquatic vertebrates chronically irradiated at dose rates <10 mGy d⁻¹, appearing instead in the 10–100 mGy d⁻¹ range (Tables 3.4 and 3.5). Once again, however, the limitations imposed by the relatively small number of studies conducted precludes a judgment that such effects will not appear in systems chronically irradiated at dose rates <10–100 mGy d⁻¹.

Still less is known about the modification of radiation effects by ecological factors, such as competition, temperature, and stress, but the existing evidence (Tables 3.6 and 3.8; Section 2) indicates the potential for significant enhancement of these effects. On the other hand, stimulatory (hormetic) effects of low radiation doses are also reported by numerous authors (Tables 3.1, 3.4, 3.7, and 3.8; IAEA, 1976; Blaylock and Trabalka, 1978; Luckey, 1980; Sagan, 1987). Such observations are not restricted to radiation effects studies (Trabalka *et al.*, 1980b). However, little is known about the mechanisms involved or the significance of such stimuli in counteracting deleterious radiation effects in natural populations. Whether ionizing radiation at very-low dose rates represents a potentially beneficial or harmful stimulus to individuals, populations, or ecosystems is an unresolved question, yet one deserving considerable further investigation (IAEA, 1976; Sagan, 1987).

The detrimental effects of radiation on populations of aquatic organisms are being considered in this report; therefore, beneficial or hormetic effects should be considered at the same level. Although increases in growth, survival, and other attributes that contribute to the fitness of an organism may be beneficial to an individual organism, the effect could be detrimental at the population or community level. A significant increase in competitive advantage as a result of an increase in size or survival of one organism could affect the stability and diversity of a community, thereby resulting in a potentially adverse effect. However, the same statement could be made about many human attempts to improve characteristics of animal and plant species and to optimize the productivity of fisheries.

Yet any prediction of the effects of radiation on fish and other populations of aquatic organisms must be made within the perspective of the reproductive rate of the species and the value of one individual to the population. In most aquatic organisms, in which reproductive rates are generally very high and on which selective

pressures are strong, the value of one or even thousands of individual organisms to the population is rather insignificant insofar as the long-term structure and fate of the population is concerned.

The same criteria cannot be applied to populations of aquatic species with low fecundity (e.g., elasmobranchs and marine mammals) or to human populations. In the absence of any data on somatic effects of chronic irradiation on low-fecund species, it is not possible to make predictions of radiation dose-effect relationships. However, low-fecund species are not prominent components of freshwater ecosystems. (At the low dose rates existing in marine environments [$\leq 2.4 \text{ mGy d}^{-1}$ reported by Woodhead, 1984], the effect on such populations is likely to be quite small in comparison with fishing, the principal stress [IAEA, 1976].) In human populations, a great value is placed on individuals, and many with relatively low adaptive values are maintained in the population, often at considerable cost. Thus, radiation exposure limits for human populations are based on stochastic effects, linearly extrapolated from data obtained at relatively high dose rates and set such that the annual average risk to an individual is within, or below, the range of 1×10^{-6} – 1×10^{-5} judged acceptable by the International Commission on Radiological Protection (ICRP, 1977). NCRP reaffirms these annual limits and considers a risk of 1×10^{-7} as a boundary below which risk is dismissed from consideration (NCRP, 1987).

In aquatic populations where less than one percent of the viable zygotes are normally expected to mature and reproduce, it would be incorrect to project that all developmental and reproductive effects reported from studies conducted at dose rates in the 10–100 mGy d^{-1} range (Tables 3.5 and 3.7) would necessarily be harmful to the exposed populations. Recruitment in fish populations is not often related to the total number of eggs and offspring produced, but more typically to the availability of food for the young fish (except at the edges of the geographic range of a species, where environmental conditions are of major importance) (IAEA, 1976). Thus, it seems highly likely that chronic irradiation at dose rates in the lower portion of the 10–100 mGy d^{-1} range, in particular, would not have a significant effect on the exposed populations unless these were already at risk due to overexploitation (e.g., fishing) or to exposures to other environmental stressors.

It also follows that even if the conservative assumption is made that the effects reported in Tables 3.4 and 3.9 for chronically exposed natural populations are totally attributable to the radiation levels current at the time of the studies, 0.6–15 mGy d^{-1} , no significant impacts on these or similarly exposed populations would be predicted. This has, in fact, been observed for White Oak Lake popula-

tions (Cooley, 1973b; Trabalka and Allen, 1977) and would be expected even in the case of the Soviet Ural lake roach population (Table 3.4), since the growth rate of juveniles was excellent (Pechkurenkov and Pokrovskaya, 1978; Peshkov et al., 1978). Long-term observations of the plaice *Pleuronectes platessa* fishery in the Irish Sea off Windscale also indicate that there has been no measurable effect of low-level irradiation on this population, which has been exposed for over 20 years (IAEA, 1976)—the average dose rate to plaice in 1967–1969 was 0.05 mGy d^{-1} (Woodhead, 1984).

4. Cytogenetic and Genetic Effects

Two of the most sensitive endpoints that can be used to study the effects of radionuclide releases on aquatic organisms are the induction of genetic and cytogenetic damage in exposed organisms (IAEA, 1979). Several reviews of this subject have been published in recent years (IAEA, 1976; Kligerman, 1979; Anderson and Harrison, 1986), and no attempt will be made to review all published research in this field. Instead, major areas of research will be examined, problems described, and an evaluation of our present state of knowledge on the effects of ionizing radiation on the genetic material of aquatic organisms will be presented.

4.1 Cytogenetic Studies

There are two main problems in studying the effects of radiation on the genome of aquatic organisms: (1) many have karyotypes consisting of large numbers of small chromosomes, and (2) most tissues of mature organisms have low mitotic activity and slow cell cycles. In combination, these two factors make studies of cytogenetic damage in aquatic organisms tedious, and due to the limited number of cells that are scoreable in many instances, results obtained may have wide confidence intervals. This limits the value of the data, especially for extrapolation to the low doses associated with environmental exposures.

Because of the technical problems encountered in using standard cytogenetic procedures (i.e., metaphase chromosome analysis) with aquatic organisms, several investigators have used analysis of chromosome bridges and fragments at anaphase to quantify cytogenetic effects of ionizing radiations. This method is only feasible for rapidly dividing tissues because there is no satisfactory way to accumulate sufficient anaphase cells for analysis. Thus, this methodology can only be used in embryos or very young organisms, which have tissues with high mitotic activity. This methodology is not deemed to be as accurate or reliable as metaphase chromosome analysis because it is prone to technically-induced artifacts.

Vertebrates

Much of the research on the genotoxic effects of ionizing radiations has focused on the induction of chromosome damage in fish exposed to high doses, and these studies will not be discussed here. Other studies examined chromosome bridge formation in the early cleavage stages of fish embryo development in fertilized eggs exposed to water-borne radionuclides. Tsytugina (1973) and Migalovskaya (1973) reported evidence of chromosome damaging effects from low activities of ^{90}Sr - ^{90}Y , ^{89}Sr , and ^{91}Y . However, changes in radionuclide concentrations over several orders of magnitude from 3.7 to $3.7 \times 10^5 \text{ Bq L}^{-1}$ produced only small changes in chromosome damage. In contrast to these studies, Pechkurenkov (1973) failed to observe cytogenetic damage following exposure of "good quality" developing roach *Rutilus rutilus lacustris* eggs to 5.6×10^6 - $1.2 \times 10^1 \text{ Bq L}^{-1}$ ^{90}Sr - ^{90}Y . Suyama et al. (1980) showed that *Limanda yokohamae* eggs reared in water containing 3.7×10^{10} - $3.7 \times 10^{11} \text{ Bq L}^{-1}$ tritium for 22 hours from the onset of fertilization had statistically significant increases in cells with chromosome bridges. Similar eggs reared in water containing 3.7×10^5 - $3.7 \times 10^7 \text{ Bq L}^{-1}$ of ^{90}Sr - ^{90}Y displayed no increase in damage; however, the interpretation of this difference is difficult in the absence of dose rate estimates.

Standard metaphase chromosome analysis is generally accepted as a sensitive and accurate means to quantify genetic damage induced by ionizing radiation. Kligerman *et al.* (1975) proposed that a model aquatic organism having a karyotype with certain desirable characteristics be chosen for studies of the effects of various clastogenic agents in aquatic environments. Of prime importance was the use of a small, hardy species having a karyotype consisting of a small number of relatively large chromosomes. They chose the central mudminnow *Umbra limi* for their studies on the effects of x rays on chromosome breakage. Although this study could be considered crude by today's standards since no dose-response curve was obtained and the aberrations were not well categorized, the investigators showed that chromosome damage could be accurately visualized in vivo in an appropriate fish species using standard metaphase chromosome techniques. Soon afterwards, Woodhead (1976) showed that chromosomes of cultured ovarian cells from a tropical fish *Ameioba splendens* responded in a similar fashion to human and toad lymphocytes following ^{60}Co gamma irradiation.

Subsequently, other investigators examined the effects of ionizing radiations in vivo on gill and spleen cells (Mong and Berra, 1979) and in vitro in lymphocytes (Suyama and Etoh, 1983) of *Umbra*

limi. Mong and Berra (1979) reported a dose-dependent increase in aberrant metaphases in fish irradiated with 3.3–9.4 Gy of x-rays.

Suyama and Etoh (1983) x-irradiated lymphocytes from mudminnows with 0.48–1.9 Gy and scored dicentrics with acentric fragments 5 days later. They introduced the important step of using 5-bromo-2'-deoxyuridine in the culture medium to ensure that only first division metaphases were scored for damage. This precludes scoring later division cells that may have lost aberrations during cell cycling and thus, increases the sensitivity of the analysis. Their data showed a significant increase in dicentric formation at the lowest exposure (0.48 Gy) examined. In addition, the dose-response curve obtained was similar to those found in fish by Woodhead (1976) and by de Boer *et al.* (1977), Bianchi *et al.* (1979) and Edwards *et al.* (1980) for human lymphocytes.

Another approach tried by several investigators is to score micronuclei in peripheral blood erythrocytes after exposing animals to ionizing radiations. Micronuclei are either acentric pieces of chromosomes produced by a chromosome breakage event or entire chromosomes detached from the spindle that are extruded from the nucleus during division and appear as DNA-specific stained bodies in the cytoplasm in an interphase cell. In order for micronuclei to be produced, the cells must divide at least once following the chromosome damaging event.

The scoring of micronuclei is a sensitive technique for measuring chromosome damage in mammalian bone marrow cells, but it has been found to be a much less sensitive measure of chromosome damage when peripheral blood erythrocytes are examined following acute exposures to known or suspected clastogens. There are two major reasons for this: (1) the spleen of some mammalian species will efficiently remove micronuclei-bearing erythrocytes from the circulatory system (Scholm *et al.*, 1975) and (2) the very large background of erythrocytes that have not divided at least once after treatment and are thus not at risk for micronuclei formation greatly reduces the percentage of micronucleated cells.

Siboulet *et al.* (1984) x-irradiated larvae of the newt *Pleurodeles waltl* with doses from 0.06–12 Gy. The authors claim that there was a significant increase in micronuclei induction in the peripheral blood erythrocytes at doses as low as 0.06 to 1.5 Gy. The dose-response curve became curvilinear with a peak at 6 Gy, and the response then decreased (the slope became negative) from 6–12 Gy. While the statistical significance of the data can be questioned at doses of 0.24 Gy and below, the method does appear to be quite sensitive. There was a substantial decline in response within 10 days after acute exposure to radiation, and 18 days after irradiation with 1.2 Gy, the

micronucleus frequency had returned to baseline levels. It should also be noted that the sensitivity of the assay depends upon using larvae at the stage when rapid division is taking place in the peripheral blood erythrocytes. The same group (Jaylet *et al.*, 1986) mention that they examined the blood of two fish *Brachydanio rerio* and *Cyprinus carpio* every other day for a period of 4 weeks (dose not reported) and failed to detect a significant increase in micronucleus formation. Similarly, Hooftman and de Raat (1982) found the micronucleus assay to be insensitive for detecting chromosome damage in the peripheral blood erythrocytes of *Umbra pygmaea* after ethyl methanesulphonate exposures.

Invertebrates

Cytological studies on the effects of radiation on aquatic invertebrates were reviewed by Anderson and Harrison (1986), Woodhead (1984), Kligerman (1979), and Blaylock and Trabalka (1978). Blaylock (1966a, b, 1973) used the giant salivary gland chromosomes from the larvae of the midge *Chironomus tentans* to investigate the frequency of chromosome aberrations in a natural population of midges that inhabited White Oak Lake that served as the final settling basin for radioactive waste from the Oak Ridge National Laboratory. A higher frequency of induced chromosome aberrations was detected in the irradiated population, which was receiving approximately 1,000 times background level of radiation (6.3 mGy d^{-1}), than in control populations. However, ten years later, the radiation dose had decreased from 6.3 to 0.3 mGy d^{-1} , and an increased frequency of induced chromosome aberrations was not detected.

In laboratory experiment, in which *Chironomus riparius* spent their entire larval and pupal stage in tritiated water, the frequency of chromosome aberration induced by β -radiation from incorporated tritium was not different from the frequency induced by an equivalent dose from external chronic gamma radiation from a ^{60}Co -source (Blaylock, 1971). The dose response curve for two-hit chromosome aberrations for β -radiation from incorporated tritium was similar to that of external γ -radiation. In these experiments a concentration of $4.6 \times 10^{-6} \text{ Bq mL}^{-1}$ of tritium in which the larval and pupal stage accumulated a dose of approximately 7.6 Gy over a 20 day period (approximately 0.015 Gy h^{-1}) was the lowest concentration at which an increased frequency of chromosome aberrations was detected (Blaylock, 1973).

Harrison *et al.* (1987) and Anderson (personal communication) have used the marine worm *Neanthes arenaceodentata* to determine

chromosome damage and sister chromatid exchanges following gamma radiation exposures. Using methods developed by Pesch and Pesch (1980), they showed that it was possible to detect a significant increase in chromosome aberrations in worms exposed to 2.0 Gy or higher x-radiation. However, due to large variations in baseline aberration frequencies, significant increases in aberrations could not be detected at doses ≤ 1.7 Gy.

Harrison *et al.* (1987) also investigated the effects of ^{60}Co irradiation on sister chromatid exchange induction in the nereid worm larvae. Although the authors state they found a significant increase in sister chromatid exchange frequency in cells from larvae exposed to 0.6 Gy, there was no dose-related increase in sister chromatid exchange frequency, and the controls showed almost as great a range of values as did the irradiated animals. These results are not surprising given that ionizing radiations are relatively poor inducers of sister chromatid exchange (Bianchi *et al.*, 1979; Crossen and Morgan, 1979; Littlefield *et al.*, 1979).

These studies and the limited number of studies covered in the previously cited reviews do not suggest a reason to suspect that aquatic invertebrates are more sensitive to radiation than other organisms. Most of the evidence suggest that the sensitivity of chromosome breakage by radiation in aquatic organisms is similar to that of other organisms.

4.2 Genetics

The number of genetic studies on aquatic organisms is limited; therefore, most of the reviews on the subject have attempted to establish whether the radiation induced mutation rate in aquatic organisms is similar to that of other organisms. Providing that the relative sensitivity of the mutation rate can be established, results of the numerous radiation genetic studies on other organisms can be used to help predict the genetic effects of radiation on aquatic organisms. In comparison to the number of studies on other organisms, data on radiation mutation rates in aquatic organisms are limited. Reviews by Schröder (1973, 1979), IAEA (1976), Blaylock and Trabalka (1978), and Woodhead (1984), on the effects of radiation on aquatic organisms include studies on radiation-induced mutation rate. In the most recent comprehensive review, Woodhead (1984) concluded that aquatic organisms show a sensitivity to the induction of mutations by radiation similar to that of other organisms and that the mutation rate ranged from 10^{-3} – 10^{-4} per gamete per 10 mGy.

In a series of studies involving the irradiation of spermatozoa and eggs of the rainbow trout *Salmo gairdneri* Newcombe and McGregor (1967) and McGregor and Newcombe (1972a) found a significant increase in the frequencies of eye malformations and body deformities at an acute dose of 0.25 Gy. A dose-response relation was established for the frequencies of malformations, which indicated that approximately 0.54 Gy would be required to double the rate observed in controls (McGregor and Newcombe, 1972b). Woodhead (1984) pointed out that in studies by Blaylock (1971, 1973) on *Chironomus riparius* the frequency of chromosome breaks for acute irradiation was 1.8×10^{-2} per gamete per Gy while the chronic exposures yielded estimates of 1 to 2×10^{-3} per gamete per Gy. Chronic exposure, which is more representative of conditions in a contaminated environment, indicated a substantially lower sensitivity for the induction of chromosome breaks. The most plausible explanation for the difference in sensitivity is probably the different degrees of repair that are possible during chronic exposures.

Studies by Blaylock (1969) and Trabalka and Allen (1978) on the mosquitofish *Gambusia affinis* population that inhabited White Oak Lake indicated that over a period of years the frequency of recessive and deleterious genes had increased in the genome of the species. However, the increased genetic load did not appear to have a detectable effect on the success of the *Gambusia* population that had received a chronic dose of 0.6 mGy d^{-1} . This *Gambusia* population had been exposed to higher dose rates for many generations although the dose rate subsequently declined to 0.6 mGy d^{-1} (60 mrad d^{-1}) (Trabalka and Allen, 1977). These results are consistent with the conclusions reached in reviews by Blaylock and Trabalka (1978) and reports from IAEA (1976) and NRCC (1983) that the increased mutation rate from an exposure of 10 mGy d^{-1} or less will not have a significant deleterious effect at the population level. These results are also consistent with the discussion on the genetic effects of radiation on aquatic organisms found in IAEA (1976) specifying that dominant lethal mutations, which often include chromosome aberrations, will be expressed in the generation after irradiation exposure; however, recessive lethals and deleterious genes will accumulate in the gene pool of the population and be expressed in future generations.

4.3 Summary

It would not be unreasonable to surmise that the genome of aquatic organisms should act similarly to that of mammalian species upon

exposure to ionizing radiations. However, good cytogenetic data to support such a concept are lacking. As can be inferred from the above review, there are presently no sensitive, reliable, or adequate means to determine cytogenetic effects of ionizing radiations on aquatic organisms. Although some attempts appear to be promising, no practical system now exists.

One can question whether monitoring cytogenetic or even genetic damage in somatic cells is of value for protecting the aquatic environment from deleterious effects of ionizing radiations. Chromosome aberrations usually lead to cell death. Although chromosome aberrations are correlated with somatic mutation and events associated with the induction of neoplasia, in most environmental exposures cell death or even individuals lost through the development of tumors would not likely have a significant effect on the aquatic community.

On the other hand, if one accepts the premise that monitoring cytogenetic or genetic damage in the somatic cells of an organism is a good measure of damage to the genetic material of the germ cells that could significantly affect the viability of aquatic communities, then one must conclude that cytogenetic monitoring of aquatic organisms is of value for setting radiation standards for aquatic environments.

Although radiation standards can be set using data obtainable from other organisms such as mammals and the limited cytogenetic data that exist on aquatic organisms, at present no good system for monitoring cytogenetic effects of low-level ionizing radiation in aquatic environments exists.

Considering the high fecundity and expected large loss of individual members of aquatic populations before attainment of sexual maturity, mutations induced by levels of radionuclides present in even the more heavily contaminated aquatic environments would be expected to have a negligible effect on community structure. However, this conclusion should be tempered by the fact that the data base is quite limited in both the number of studies and the number of species examined. In addition, our understanding of the complex ecological interactions that take place in an aquatic ecosystem is far from complete.

5. Criteria for the Protection of Populations of Aquatic Organisms

In the human context, the standards for exposure to ionizing radiation are based upon acceptable doses to individuals. Two sets of standards have been established, one for the general public and the other for those exposed to additional radiation in the work place. For the former the annual dose limit is 1 mSv; for occupational exposure, the annual dose limit is 50 mSv (NCRP, 1987). Within these limits, good practice requires that the exposure should be as low as reasonably achievable (ICRP, 1977; NCRP, 1987).

In contrast, for endemic aquatic organisms it is the response of the population rather than the individual which is of consequence, and especially the capacity of the population to maintain itself through reproduction and competition in the face of the stress imposed by increased radiation exposure. Thus, there can be effects in the individual, which, if there are no consequences at the population level, may be considered acceptable. Because maintenance of endemic populations is the important requirement, it is the preservation of fertility and fecundity which is necessary, and it is precisely these components of the life cycle which have been found to be most sensitive to the effects of chronic irradiation. As the natural fecundity differs between species and even populations of these species, this factor may be of considerable importance in assessing the response of a population to radiation exposure; a highly fecund species is likely to be more resilient than those with low fecundity. If exposures are limited to protect fertility and fecundity, it is most unlikely that other effects expressed in individuals will be detrimental to the population.

Experimental studies under carefully controlled conditions in the laboratory have shown detectable effects on fecundity at dose rates down to 0.4 mGy h^{-1} (Section 3). Although detectable effects have been reported in natural populations exposed to comparable dose rates the effects observed are, at least partially, attributable to much greater dose rates delivered to the populations in the decades preceding the studies. Significant effects on fecundity have been observed

in aquatic vertebrates only at dose rates $>1 \text{ mGy h}^{-1}$ (Woodhead, 1984) in controlled studies. Mortality effects appear only at much higher dose rates. Thus, it is likely to be very difficult to detect deleterious effects at the population level until the $0.4\text{--}1 \text{ mGy h}^{-1}$ dose rate has been exceeded. Exceptions to this might possibly be highly valued and/or threatened species with low fecundity.

The data reviewed in Sections 2 through 4 may be summarized as follows: consistently damaging effects of irradiation during the development of salmon eggs *Oncorhynchus tshawytscha* did not become apparent until the dose rate over the 81-day development period reached 4 mGy h^{-1} . Studies under carefully controlled conditions in the laboratory have shown detectable histological effects on the gonads of small tropical fish at dose rates down to 0.4 mGy h^{-1} .

In the contaminated environment of White Oak Lake an increased frequency of chromosome aberrations was found in midge larvae *Chironomus tentans* at a dose rate of $2.6 \times 10^{-1} \text{ mGy h}^{-1}$ but not at $1.3 \times 10^{-2} \text{ mGy h}^{-1}$. As the cytogenetic response is probably one of the most sensitive indicators of radiation-induced damage, if it is undetectable there is little possibility of damage at higher levels of organization. In view of the relatively high fecundity and high larval and juvenile mortality in most aquatic populations, it is improbable that mutations induced by increased radiation exposure in even the most heavily contaminated environments could have any detectable effect at the population or community level.

In the same lake, a statistically significant increased incidence of dead embryos of mosquitofish *Gambusia affinis* was found after the population had been exposed to time-varying dose rates in the range of $2.5 \times 10^{-2}\text{--}4.0 \times 10^{-1} \text{ mGy h}^{-1}$ (Blaylock, 1969; Trabalka and Allen, 1977), and it was concluded that this was a consequence of genetic effects; however, this had no detectable effects at the population level. In the northeast Irish Sea, it has been shown that average, long-term exposures of $2.1 \times 10^{-3} \text{ mGy h}^{-1}$, with a maximum of $1.5 \times 10^{-2} \text{ mGy h}^{-1}$ (Woodhead 1984), have had no detectable effects on the commercially exploited population of plaice *Pleuronectes platessa*. The dose rates at which no effects have been found are of the order of the maximum permissible dose rate for radiation workers, i. e., $2.5 \times 10^{-2} \text{ mSv h}^{-1}$, and it is probable that any effects of the increased radiation exposure have been accommodated with the reserve reproductive capacity of the populations.

There remains a factor of 33 between the highest environmental dose rate at which no radiation effects have been observed ($2 \times 10^{-2} \text{ mGy h}^{-1}$) and the lowest dose rate at which some effects have been detected in carefully controlled experiments (0.4 mGy h^{-1}). Although it is potentially possible that there could be very minor radiation

effects within this range, their significance in the natural environment where other factors are operative, is probably minimal. On the basis of the evidence presented in this report, it seems reasonable to conclude that a dose rate to protect endemic aquatic populations should be set within this range.

Studies of contaminated environments have shown that point discharges of radionuclides generate a varying dose field in the receiving environment and that the mean population exposure is less than the exposure at the point of discharge either because the population of sedentary organisms exist throughout the varying dose field, or because mobile organisms experience a time-varying dose rate as they migrate within the environment. Thus, if the dose rate standard is applied to the organisms subjected to the maximum dose rate in the contaminated environment, then the average exposure of any population, or subpopulation of organisms will be less, and often much less than the standard. For this reason it is suggested that a maximum dose rate of 0.4 mGy h^{-1} would provide protection for endemic populations of aquatic organisms in environments receiving discharges of radioactive effluent.

There is an indication that a limit of $5.7 \times 10^{-3} \text{ mSv h}^{-1}$ equivalent to 50 mSv a^{-1} has been adopted in the USSR to provide protection for fish populations (Gusev and Pavlovskij, 1982). However, Izriel (1987) in considering the environmental effects of the Chernobyl accident considers that chronic exposures of 0.4 mGy h^{-1} to aquatic biota, while causing some fractional changes, maintains ecological stability at the population and organism level.

Adoption of a reference level of 0.4 mGy h^{-1} appears to represent a reasonable compromise based on current information, i. e., considering both the nature of the effects observed at this dose rate and the limited amount of information on effects of radiation in natural populations, including interactions between ionizing radiation and ecological conditions. Populations exposed to dose rates approaching 0.4 mGy h^{-1} may also be at risk from other factors such as over-exploitation or environmental stressors which might, in combination, result in an undesirable impact. In such circumstances, it would seem highly desirable to conduct a comprehensive ecological evaluation of the radiation exposure regime along with the other factors in order to determine the potential consequences. Thus, it is suggested that where the results of radiological modelling and/or dosimetric measurements indicate that a radiation dose of 0.1 mGy h^{-1} will be exceeded, such an evaluation should be conducted.

6. Environmental Dosimetry

The practice of discharging of radioactive wastes into terrestrial or aquatic environments entails a potential for the contamination, to a greater or lesser degree, of the many components of these environments. Such contamination inevitably results in increased exposure of populations of wild organisms, and it is quite clear that the impact of the increased exposure can only be assessed if the magnitude of the incremental radiation dose can be determined. This simple statement belies the potential complexity of the requirement in practice. An organism may be irradiated externally by radionuclides in any or all of the air, water, soil (sediment) and vegetation, and internally by radionuclides accumulated within the body by inhalation or by direct absorption through gills or the integument in aquatic organisms and by ingestion of food and water. In the latter case it is likely that the various radionuclides would be differentially distributed among the organs and tissues within the organism. In addition, the relative significance of internal and external sources can be markedly altered by the size and behavior of the organisms. Thus, the radiation sources and the consequential radiation dose rate are inhomogeneous in space and time both for a particular radionuclide and for different radionuclides. An additional requirement is the definition of the sensitive target (or targets) for which the radiation exposure must be estimated. The choice depends upon a number of factors including the potential magnitudes of dose rate and total dose, the radiosensitivities of the organisms and whether it is the individual organism or the population which is deemed to be at risk. These considerations, which are not necessarily all-inclusive, indicate the magnitude of the problem, and it is clear that informed simplifications and generalizations have to be introduced to reduce it to manageable proportions.

In general, the discharge of radioactive wastes into the environment, because it will be controlled to limit human exposure, will result in only long-term, low-dose-rate exposure of animals and plants. In these circumstances, acute mortality can be discounted and the very small degree of morbidity and mortality which could be attributed to the incremental irradiation is unlikely to be detectable against the natural mortality in the populations. Thus, it is the local population of a species, and its ability to maintain itself through

reproduction, which is of concern. The usual approach is to select organisms which are significant and representative of the particular environment under investigation. That is, either they might be important in terms of the community structure, or they might have behavior which leads to above average exposure to external sources of irradiation, or their eating habits might lead to significantly higher than usual concentrations of radionuclides within the body. Again, these criteria are not all-inclusive, but examination of the situation will usually allow the identification of a few critical species of organisms for which dose estimates should be made, and which will provide an adequate basis for an overall assessment of the radiation regime and its potential consequences. In summary, the problem of dosimetry may be reduced to:

- (a) Making assessments for a few critical or significant species of organisms, i.e., indicator species and
- (b) Making estimates of the average exposure of the whole body of the organism while ensuring that this is representative of the exposure to the gonads, (i.e., there is not increased differential accumulation of radionuclides either within, or adjacent to, these organs). In addition, it is useful to make estimates of the dose rate to the most highly exposed tissue or organ within the organism.

6.1 Methods

As it turns out, it is rarely practicable to obtain estimates of the radiation dose rate to organisms in a contaminated, but otherwise natural, environment by direct measurement. The difficulties include the problematic evaluation of the potential importance of α - and β -emitting radionuclides accumulated within the organism, the estimation of errors introduced by the variations in external exposure due to the behavior of mobile animals in a nonuniformly contaminated environment, and the logistical complications imposed by the requirement for a capture-recapture program if a passive dosimeter (e.g., LiF) is to be used for *in situ* measurements.

These factors enforce dependence upon modelling methods to determine the behavior of the radionuclides when released to the environment and for the estimation of the radiation dose rate to selected targets from the predicted distributions of the radionuclides external to, and within, the animals. A number of approaches have been developed for particular situations and may be capable of adaptation for more general application.

CRITR

This set of models and the associated computer codes (Soldat *et al.*, 1974) was developed for application to discharges of effluent into surface waters. It provides a simplified means of calculating the concentrations of radionuclides in water, sediment, and two groups of organisms using a restricted number of parameters relating to the discharge and the receiving water body.

The concentration of radionuclide i in water at the point of interest is given by:

$$C_w(P)_i = 10^{-3} \times \frac{Q_i N_i}{F} M_p \exp(-\lambda_i t_p) \text{ (Bq L}^{-1}\text{)} \quad (6-1)$$

where

- Q_i is the release rate of radionuclide i (Bq s⁻¹);
- N_i is the reconcentration factor of radionuclide i due to recycling of contaminated water, if applicable (dimensionless);
- F is the flow rate of the liquid effluent (m³ s⁻¹);
- M_p is the mixing ratio at the point of interest (i.e., a dimensionless dilution factor);
- t_p is the transit time to the point of interest (s); and
- λ_i is the decay constant for radionuclide i (s⁻¹).

The concentration of radionuclide i in sediment at the point of interest is given by:

$$C_s(P)_i = \frac{K_i C_w(P)_i}{\lambda_i} (1 - \exp[-\lambda_i t_s]) \text{ (Bq kg}^{-1}\text{)} \quad (6-2)$$

where

- K_i is the radionuclide transfer rate from water to sediment (L kg⁻¹s⁻¹);
- $C_w(P)_i$ is the concentration of radionuclide i in the water at the point of interest (Bq L⁻¹);
- t_s is the time for which the sediment has been exposed to contaminated water; effectively the operating life of the facility (s); and
- λ_i is the decay constant of the radionuclide i (s⁻¹).

There are two points which must be made in connection with this expression: first, the transfer of radionuclides is in one direction only—water to sediment—and because transfer in the opposite sense would also be expected, the value of λ_i should be larger than that due to radioactive decay alone; second, the analysis has been greatly simplified by assuming a time-independent water concentration.

The two groups of representative organisms for which concentrations are estimated are, respectively, fully aquatic (waterweeds, molluscs, crustacea, and fish) and semi-aquatic (herbivorous ducks, herons, muskrats, and raccoons). For the first group the equilibrium body burden of a radionuclide is simply determined from the water concentration by application of an appropriate concentration factor. Thus, the concentration of radionuclide i is given by:

$$C_B(P)_i = C_w(P)_i CF_i \text{ (Bq kg}^{-1}\text{)} \quad (6-3)$$

where

$C_w(P)_i$ is the water concentration of radionuclide i at the point of interest obtained from Eq. 6-1; and

CF_i is the concentration factor of radionuclide i in the organism of interest.

For the second group of organisms, the main source of internal radionuclides is the consumption of organisms of the first group, and the concentration of radionuclide i is given by:

$$C_k(P)_i = \frac{I_{ki} f_{ki} (1 - \exp[-\lambda_{ek} t])}{\lambda_{ek} m_k} \text{ (Bq kg}^{-1}\text{)} \quad (6-4)$$

where

I_{ki} is the ingestion rate of radionuclide i by organism k via the consumption of contaminated organisms of the first group (Bq s^{-1});

f_{ki} is the proportion of the intake of radionuclide i which is absorbed across the gut into the body of organism k (dimensionless);

λ_{ek} is equal to $\lambda_i + \lambda_{ki}$, the sum of the physical decay constant and the biological excretion rate in organism k for radionuclide i (s^{-1});

m_k is the body mass of organism k (kg); and

t is the period of exposure(s).

This expression implies a much greater knowledge of the feeding behavior of the secondary organisms and their metabolism of radionuclides than is, in fact, available. Assumptions have to be made concerning the values appropriate for the radionuclide intake and excretion rates and the absorbed fraction.

For both groups of organisms, the contaminant radionuclides are assumed to be uniformly distributed throughout the body. To determine the absorbed dose rate from such contamination an estimate is required of the fraction of the emitted energy which is absorbed within the body. For organisms of the size being considered, this fraction is taken to be unity for both α - and β - particles emitted by

the radionuclides of interest, which have ranges less than, or of the order of, the organisms' dimensions. For γ -rays the situation is quite different because the absorption mean free path is generally greater than the organisms' dimensions except at very low energies and the absorbed fraction can be substantially less than unity, but increasing with body size. A set of values for the effective absorbed energy per disintegration from all contributing radiation types is given in the model for a series of muscle-tissue spheres of different radii (1.4–20 cm) for 136 radionuclides (including short-lived daughters where appropriate). Each organism is then approximated by an equivalent sphere, e. g., crustacean, 2 cm radius; raccoon, 14 cm radius; etc.

The dose rates to the two groups of organisms from internal contamination are given by:

$$D_B = 5.76 \times 10^{-4} C_B (P)_i E_i (\mu\text{Gy h}^{-1}) \quad (6-5)$$

and

$$D_k = 5.76 \times 10^{-4} C_k (P)_i E_i (\mu\text{Gy h}^{-1}) \quad (6-6)$$

respectively, where

where

- $C_B (P)_i$ and $C_k (P)_i$ are the body burdens of radionuclide i estimated from Equations 6-3 and 6-4 (Bq kg^{-1}); and
- E_i is the effective absorbed energy per disintegration for radionuclide i in the sphere taken to be equivalent to the organisms of interest (MeV).

To calculate the dose rate from contaminated sediment, the radionuclide concentration (Eq. 6-2) is converted to an effective surface concentration given by:

where

$$C'_s (P)_i = 100 \tau_i C_w (P)_i W (1 - \exp [-\lambda_i t_s]) (\text{Bq m}^{-2}) \quad (6-7)$$

where

- τ_i is equal to $\frac{0.693}{\lambda_i}$, the half-life of radionuclide i (s); and

W is a factor set equal to 2 for small organisms and 1 for large organisms to take account of the finite extent of the sediment source which is effective in irradiating the organism (dimensionless).

The numerical factor includes the conversion of Bq kg^{-1} integrated over a depth of 2.5 cm to Bq m^{-2} , and the value of K which is apparently independent of the radionuclide. This empirical expres-

sion was derived from observations made over a number of years on Columbia River sediments (Nelson, 1965; Toombs and Culter, 1968).

The dose rate is then:

$$D_s = C'_s (P)_i U_p D_{is} (\mu\text{Gy h}^{-1}) \quad (6-8)$$

where

U_p is the occupancy factor, i.e., the proportion of the total available time spent on contaminated sediment at point P (dimensionless); and

D_{is} is the dose rate from radionuclide i at a height of 1 m ($\mu\text{Gy h}^{-1}/\text{Bq m}^{-2}$). For small organisms which are effectively much closer to the source, the dose rate is increased by a factor of 2 over that at 1 m and this is included in the value of 2 assumed for the shore width factor W .

The dose rate from the contaminated water is given by:

$$D_w = \frac{C_w(P)_i}{K_p} U_p D_{iw} (\mu\text{Gy h}^{-1}) \quad (6-9)$$

where

U_p is the proportion of the total available time spent swimming on the surface (e.g., ducks) or totally immersed (e.g., fish, ducks) (dimensionless);

D_{iw} is the dose rate from radionuclide i in the water ($\mu\text{Gy h}^{-1}/\text{Bq L}^{-1}$) and

K_p is a factor equal to 1 for immersion, and equal to 2 for organisms on the surface or at the sediment-water interface.

Tabulated values of D_{is} and D_{iw} are given for the geometry of the human body, and these are assumed to apply to the organisms under consideration. Their use will, in fact, somewhat underestimate the dose rate to the smaller organisms.

These models have been developed and generalized so as to be applicable at any site and are, therefore, particularly useful for preoperational assessments, although site-specific information should be employed whenever possible. Once discharges have commenced, the models should be further developed to use the data which become available from research programs and monitoring surveys, and in particular the measurements of radionuclide concentrations in the various components of the environment.

EXREM III and BIORAD

These approaches have been developed from the starting point of a unit concentration of a radionuclide in water, with the concentra-

tion in the organisms being determined simply by the application of a concentration factor (Eq. 6-3) (Trubey and Kaye, 1973). No means of estimating the concentrations of radionuclides in sediment are given.

The dose rate to the organisms from the radionuclides in the water is derived from the mean dose rate in an effectively infinite (i.e., dimensions much greater than the radiation attenuation length), uniformly contaminated source, i.e., $D(\infty)$ defined as:

$$D(\infty) = 5.76 \times 10^{-4} \bar{E}_i (\mu\text{Gy h}^{-1}/\text{Bq L}^{-1}) \quad (6-10)$$

where

\bar{E}_i is the mean energy of all radiations emitted per disintegration (MeV).

Due to their very limited range, the contribution to the whole body dose rate from α particles emitted by nuclides in the water is taken to be zero. For β particles which have ranges up to a few centimeters in water, the contribution is taken to be $0.5 D_\beta(\infty)$, while for the much more penetrating γ -radiation it is taken to be $D_\gamma(\infty)$. Values of the immersion dose rate derived in this way are listed for a number of radionuclides (including contributions from radioactive daughters where appropriate) which are of significance in the nuclear fuel cycle. (The units are mrad a^{-1} for a water concentration of $1 \mu\text{Ci mL}^{-1}$, but can be converted to $\mu\text{Gy h}^{-1}/\text{Bq L}^{-1}$ by multiplying by the factor 3.08×10^{-11}).

The dose rate from radionuclides accumulated within the organisms from water at a unit concentration of 1 Bq L^{-1} is given by:

$$D = 5.76 \times 10^{-4} CF_i E_i (\mu\text{Gy h}^{-1}) \quad (6-11)$$

where

E_i is the effective absorbed energy per disintegration for radionuclide i in an appropriate geometry (MeV) and

CF_i is the concentration factor of radionuclide i in the organism of interest.

In deriving the values of E_i , the radius of the sphere taken to be equivalent to the organisms has been assumed to be 30 cm except for the muskrat for which a value of 10 cm was used. For most organisms, the effective radius would be less than 30 cm so that the method provides an overestimate of the dose rate. The required concentration factors and the resulting absorbed dose rates for freshwater (water weed, invertebrates, fish, and muskrat) and marine (algae, molluscs/crustaceans, fish, and waterfowl/shore birds) organisms have been given. (Again a simple unit conversion is necessary).

The potentially significant exposure of the organisms from contaminated sediment is not considered by this model.

Point Source Dose Distributions

For any extended source of ionizing radiation, the dose rate at a specified point can be obtained by the integration of an appropriate point source dose function over the source geometry. Although it is possible to derive theoretical expressions from first principles, these are frequently complex due to the multiplicity of absorption and scattering phenomena which must be considered. For ease of computation, simple empirical expressions have been described (IAEA, 1976; 1979).

The point source dose functions are:

For α -radiation

$$D_{\alpha} = \frac{4.59 \times 10^{-2}}{\rho r^2} (A + Br^2) (\mu\text{Gy h}^{-1} \text{ Bq}^{-1}) \quad (6-12)$$

where

- ρ is the density of the medium (g cm^{-3});
- r is the distance between the point source and the target point (μm); and
- A and B are parameters dependent upon the emission energy of the α -particle, and for which expressions are provided.

For β -radiation

$$D_{\beta} = \frac{k}{(\rho \nu r)^2} \left\{ a \left[1 - \frac{(\rho \nu r)}{c} \exp \left(1 - \frac{\rho \nu r}{c} \right) \right] + \rho \nu r \exp \left(1 - \frac{\rho \nu r}{c} \right) \right\} (\mu\text{Gy h}^{-1} \text{ Bq}^{-1}) \quad (6-13)$$

where

- $\left[1 - \frac{\rho \nu r}{c} \exp \left(1 - \frac{\rho \nu r}{c} \right) \right] \equiv 0$ for all $r \geq \frac{c}{\rho \nu}$
- $k = \frac{4.59 \times 10^{-2} \rho \nu^3 \bar{E}_{\beta} n_{\beta}}{ac(3-e) + e} (\mu\text{Gy h}^{-1} \text{ Bq}^{-1})$
- r is the distance between the point source and the target point (cm);
- e is the exponential constant;
- ν is an energy dependent apparent absorption coefficient ($\text{cm}^2 \text{g}^{-1}$);
- a and c are energy dependent parameters; and
- n_{β} is the proportion of disintegrations producing β -particles of mean energy \bar{E}_{β} MeV (dimensionless).

Expressions to provide values for a , c , and v as a function of \bar{E}_β are given.

For γ -radiation and for small organisms where attenuation and scattering may be neglected

$$D_\gamma = 4.59 \times 10^{-2} \frac{\mu}{\rho} \frac{E_\gamma n_\gamma}{r^2} \left[1 - \exp \left(-\frac{2.30 r}{r_e (0.3 E_\gamma)} \right) \right] (\mu\text{Gy h}^{-1} \text{ Bq}^{-1}) \quad (6-14)$$

where

$\frac{\mu}{\rho}$	is the mass energy absorption coefficient of the target material at E_γ MeV ($\text{cm}^2 \text{g}^{-1}$);
n_γ	is the proportion of disintegrations producing a γ -ray of energy E_γ MeV (dimensionless);
r	is the separation of the point source and the target point (cm); and
$r_e (0.3 E_\gamma)$	is the range in the target material of an electron with initial energy $0.3 E_\gamma$ MeV (cm).

For larger organisms, where absorption and scattering become significant, i.e., those with dimensions greater than a few cm, a factor to take account of these processes must be applied. Thus

$$D'_\gamma = D_\gamma B \exp(-\mu_T r) (\mu\text{Gy h}^{-1} \text{ Bq}^{-1}) \quad (6-15)$$

where

B	is an empirical build-up factor at energy E_γ MeV (dimensionless); and
μ_T	is the total γ -ray absorption coefficient at energy E_γ MeV (cm^{-1})

Values of B have been tabulated (Loevinger et al., 1956). Fortunately, Monte Carlo calculations have been made which include absorption and scattering for a number of geometries which can be adapted for aquatic organisms (Brownell et al., 1968; Ellett and Humes, 1971). The results are given in terms of the absorbed fraction Φ (see Appendix A) defined as:

$$\Phi = \frac{\text{photon energy absorbed by target}}{\text{photon energy emitted by source}}$$

The mean dose rate is then

$$D_\gamma = 5.76 \times 10^{-4} \frac{E_\gamma n_\gamma \Phi}{m} (\mu\text{Gy h}^{-1} \text{ Bq}^{-1}) \quad (6-16)$$

where m is the mass of the target (kg).

In the particular case where the source and target volumes are coincidental

$$D_{\gamma} = 5.76 \times 10^{-4} E_{\gamma} n_{\gamma} \Phi \text{ (}\mu\text{Gy h}^{-1}\text{/Bq kg}^{-1}\text{)} \quad (6-17)$$

This approach has the advantage that it can be applied to any combination of source and target geometries although it is usual to simplify these so that analytical (as opposed to numerical) integration is possible. Examples of the application of this approach to practical problems have been given by several authors (Woodhead, 1970; 1973; 1974; 1979; 1986; Pentreath *et al.*, 1973; Hetherington *et al.*, 1976; Hoppenheit *et al.*, 1980).

These models and codes have been applied (see Appendix A) to estimate dose rates to aquatic organisms at four North American sites where aquatic environments have been contaminated in the past as the result of nuclear fuel cycle activities. It should be emphasized that the radionuclide concentrations and distributions were not specifically collected for environmental dosimetry but were those available as monitoring and research data in the literature. The data are insufficiently detailed to permit the estimation of dose rates to specific target organs, (e.g., the gonads) so the whole body was taken as the target tissue. The resultant estimated dose rates were primarily calculated to provide comparisons between the models considered in this report. The agreement between CRITR and PSDD was excellent. Reasonable agreement was indicated between BIORAD and CRITR and PSDD, where comparable dose rates were calculated.

6.2 Summary

The methods for environmental dosimetry which have been briefly described were developed for specific purposes and should not be judged outside that context. CRITR would appear to be the most useful approach to adopt in the preoperational assessment phase of any waste management project when no observational data are available on the distributions of radionuclides in the environment. Through the application of simple models requiring a minimum of parameters, it allows the prediction of the distributions of the discharged radionuclides in the various compartments of the aquatic environment. If the values of the parameters are chosen conservatively, but not unduly so, to take account of uncertainties, reasonably realistic upper-bound estimates of the dose rates to a variety of organisms can be obtained. These provide the only basis upon which an assessment of the impact of a discharge on wild populations can

be made. The potential significance of the incremental radiation exposure may be evaluated in a number of ways:

- (a) The estimated dose rates may be compared with the variation in the natural radiation background or, indeed, the natural background itself.
- (b) Comparisons may be made with the dose rates which have been shown to produce significant detrimental effects on populations of organisms in laboratory or field studies.
- (c) If, and when, limits are set on the incremental dose rates for the purposes of environmental protection, comparisons may be made against these criteria.

Where any or all of these comparisons show that the incremental dose rates are relatively small, then it may be concluded that the discharge could be permitted from the point of view of environmental protection, but with the requirement that there should be a continuing program of monitoring and research (at least, in the short term) to confirm the conclusion.

If the comparisons show the incremental dose rates to be relatively high, then a first step could be a site-specific research program to improve the detailed understanding of the behavior of the radionuclides in the environment so that refined and more realistic estimates of the incremental exposure can be made. However, the cost of this should be set against the cost of improved waste management methods and reduced discharges, and rational deployment of resources made.

Once discharges have commenced, research and monitoring programs make available detailed, site-specific data on the behavior and distribution of the radionuclides, and this should permit improved estimates of the incremental exposure of the aquatic organisms. Any of the dosimetric methods which have been described could be used, but it should be noted that the point source dose distribution functions require data of considerable detail on the radionuclide distributions to realize their full potential. Thus, the data should be collected with their possible utilization for environmental dosimetry clearly in mind. If this is not done, the simplifications imposed by the paucity of the information rapidly reduce the different approaches to a common level. The point source dose functions are particularly useful for application to small organisms (e.g., developing embryos) (Woodhead, 1979; Hoppenheit et al., 1980) or where it is necessary to estimate the dose rate to a radiosensitive tissue (e.g., the gonad) from a nonuniform radionuclide distribution (Pentreath et al., 1973).

7. Dose to Aquatic Organisms and Man from Environmental Radioactivity

Radiation protection standards have been expressly developed for the protection of human health; however, it has been generally accepted and adopted by those involved with radiation standards that by "*protecting humans we are protecting the environment.*" Statements for general acceptance of this philosophy are found in the 1972 BEIR (Biological Effects of Ionizing Radiation) Report (National Academy of Sciences, 1972) which states that:

"Evidence to-date indicates that probably no other living organisms are very much more radiosensitive than man so that if man as an individual is protected, then other organisms as populations would be most unlikely to suffer harm."

A similar statement can be found in the recommendations of the International Commission on Radiological Protection, (ICRP, 1977):

"Although the principal objective of radiation protection is the achievement and maintenance of appropriately safe conditions for activities involving human exposure, the level of safety required for the protection of human individuals is thought likely to be adequate to protect other species, although not necessarily individual members of those species. The commission therefore believes that if man is adequately protected then other living things are also likely to be sufficiently protected."

Although this viewpoint has been generally accepted, it has not previously been seriously challenged nor formally defended.²

²More recently the ICRP has modified its statement on the subject as follows:

"The Commission believes that the standard of environmental control needed to protect man to the degree currently thought desirable will ensure that other species are not put at risk. Occasionally, individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species (ICRP, 1991)."

It is well documented that radionuclides in the environment can be expected to produce similar or even substantially higher doses to certain organisms than to people inhabiting and deriving sustenance from the same environment. Therefore, the risk of radiation effects (discounting variations in radiosensitivity, lifespan, etc.) would appear as high or higher for natural biota than for humans. However, there is a basic difference in how we perceive risks for humans compared to other species. For humans, a great value is placed on an individual member, and the loss of one or a few individuals from radiation exposure is viewed as a catastrophic event. On the contrary, most other species are viewed more as a population than as identifiable individuals, and the loss of a few individuals, or in the case of many aquatic organisms, thousands of individuals, can be accepted without producing a noticeable or deleterious effect on the population (Section 3). This philosophy is expressed, although not clearly, in the previous statements from the BEIR Report (NAS, 1972) and ICRP-26 (ICRP, 1977) and perhaps explains the general acceptance of the statement that if "*humans are protected from radiation then the environment is protected.*"

In Sections 2, 3, and 4 we have reviewed the effects of radiation on aquatic organisms from the cellular level to the population level, and in Section 5 we have made recommendations for standards to protect the environment. Therefore, by calculating a dose rate to aquatic biota in a contaminated environment from which humans receive a limiting dose of 1 mSv a^{-1} , the potential effects of radiation on aquatic organisms and populations can be assessed against the radiation standards established for the protection of human health.

A number of approaches have been taken in calculating dose rates to aquatic organisms in the environment. These approaches are discussed in Section 6 of this report. For the present scenario the BIORAD computer code (Trubey and Kaye, 1973) can be used to calculate the internal and external dose rate to fish. Under the assumptions of the models used in BIORAD, only the concentration of the individual radionuclides in water is required to calculate a dose rate to the biota.

It is assumed that the concentrations of radioactivity in water, sediment and biota are at equilibrium and that the concentration of radioactivity in the water is constant. The effective dose equivalent to humans from this environment is assumed to have reached a limiting value of 1 mSv a^{-1} . The question is whether the concentration of radioactivity in the tissue of the fish and the surrounding environment is sufficient to produce a radiation dose high enough to result in a detrimental effect on the fish populations.

The assumption is that a dose of 1 mSv a^{-1} to humans is acquired via the following pathways: eating fish (100 g d^{-1}), drinking water (2 L d^{-1}) and exposure to sediments for 2000 h a^{-1} (IAEA, 1982). This would imply maximum use of the aquatic ecosystem resulting in a maximum credible dose. The percentage of the dose 1 mSv a^{-1} obtained via each pathway varies for each radionuclide. Therefore, doses were calculated for each pathway and each radionuclide to determine the contribution from each pathway to the total dose. Doses to humans were calculated for a concentration of 37 Bq L^{-1} in water using the following equations and dose conversion factors (ICRP-30) for ingestion and exposure to contaminated ground from the New Standard Guide EXXX (ASTM, 1986):

Drinking Water

$$D_w = C_w CR_w Dc_w \quad (7-1)$$

where:

- D_w is the annual effective dose equivalent (Sv a^{-1}) from consumption of water
- C_w is the concentration in water (Bq L^{-1}) taken to be 37 Bq L^{-1}
- CR_w is the consumption rate of water (L a^{-1})
- Dc_w is the dose conversion factor (Sv Bq^{-1})

Consuming Fish

$$D_f = C_w BF CR_f Dc_f \quad (7-2)$$

where:

- D_f is the annual effective dose equivalent in (Sv a^{-1}) from eating fish
- C_w is the concentration in water (Bq L^{-1})
- BF is the bioaccumulation factor for fish (L Kg^{-1})
- CR_f is the consumption rate of fish (kg a^{-1})
- Dc_f is the dose conversion factor for ingestion (Sv Bq^{-1})

Exposure to Sediment

$$D_s = C_w K_d Dc_s \quad (7-3)$$

where:

- D_s is the annual effective dose equivalent (Sv a^{-1})
- C_w is the concentration in water (Bq Kg^{-1})
- K_d is the sediment distribution coefficient (L Kg^{-1})
- Dc_s is the dose conversion factor (Sv a^{-1}) per (Bq g)

The estimated doses from drinking water, consuming fish, and being exposed to external radiation from contaminated sediments are given for each radionuclide in Tables 7.1, 7.2, and 7.3, respectively. These estimated doses can be used to determine the percentage of the dose contributed by each of these pathways (Table 7.4). In addition, the total dose rate estimated for the combination of the

TABLE 7.1—Dose to humans from drinking water contaminated with 37 Bq L⁻¹ of the following radionuclides

Radionuclide	C _w (Bq L ⁻¹)	CR _w (L a ⁻¹)	Dc _w (Sv Bq ⁻¹)	D _w (Sv a ⁻¹)
³ H	3.70E+01	7.30E+02	1.70E-11	4.59E-07
³² P	3.70E+01	7.30E+02	2.08E-09	5.62E-05
⁶⁰ Co	3.70E+01	7.30E+02	6.97E-09	1.88E-04
⁹⁰ Sr	3.70E+01	7.30E+02	3.60E-08	9.72E-04
⁹⁹ Tc	3.70E+01	7.30E+02	3.38E-10	9.13E-06
¹³¹ I	3.70E+01	7.30E+02	1.44E-08	3.89E-04
¹³⁷ Cs	3.70E+01	7.30E+02	1.35E-08	3.65E-04
²²⁶ Ra	3.70E+01	7.30E+02	3.05E-07	8.24E-03
²³⁵ U	3.70E+01	7.30E+02	6.84E-08	1.85E-03
²³⁸ U	3.70E+01	7.30E+02	6.32E-08	1.71E-03
²³⁹ Pu	3.70E+01	7.30E+02	1.19E-07	3.21E-03
²⁴¹ Am	3.70E+01	7.30E+02	5.94E-07	1.60E-02

C_w = concentration in water

CR_w = consumption rate of water

Dc_w = dose conversion factor for ingestion

D_w = effective dose equivalent from drinking water

TABLE 7.2—Dose to humans from eating fish from water contaminated with 37 Bq L⁻¹ of the following radionuclides

Radionuclide	C _w (Bq L ⁻¹)	BF (L kg ⁻¹)	CR _f (kg a ⁻¹)	Dc _f (Sv Bq ⁻¹)	D _f (Sv a ⁻¹)
³ H	3.70E+01	1.00E+00	3.65E+01	1.70E-11	2.30E-08
³² P	3.70E+01	1.00E+05	3.65E+01	2.08E-09	2.81E-01
⁶⁰ Co	3.70E+01	2.00E+01	3.65E+01	6.97E-09	1.88E-04
⁹⁰ Sr	3.70E+01	5.00E+00	3.65E+01	3.60E-08	2.43E-04
⁹⁹ Tc	3.70E+01	1.50E+01	3.65E+01	3.38E-10	6.85E-06
¹³¹ I	3.70E+01	1.50E+01	3.65E+01	1.44E-08	2.92E-04
¹³⁷ Cs	3.70E+01	4.00E+02	3.65E+01	1.35E-08	7.29E-03
²²⁶ Ra	3.70E+01	5.00E+01	3.65E+01	3.05E-07	2.06E-02
²³⁵ U	3.70E+01	1.00E+01	3.65E+01	6.84E-08	9.24E-04
²³⁸ U	3.70E+01	1.00E+01	3.65E+01	6.32E-08	8.54E-04
²³⁹ Pu	3.70E+01	3.50E+02	3.65E+01	1.19E-07	5.62E-02
²⁴¹ Am	3.70E+01	2.50E+01	3.65E+01	5.94E-07	2.01E-02

C_w = concentration in water

BF = bioaccumulation factor—data taken from Table 4-2, NRCC (1983)

CR_f = consumption rate of fish

Dc_f = dose conversion factor for ingestion

D_f = effective dose equivalent for eating fish

TABLE 7.3—Dose to humans from exposure to external radiation from contaminated sediments for 2000 h a⁻¹ with an occupancy factor of 0.228

Radionuclide	C _w (Bq L ⁻¹)	K _d (L Kg ⁻¹)	D _s (Sv a ⁻¹)/(Bq Kg ⁻¹)	(Sv a ⁻¹)
³ H	3.70E+01	0	0	0
³² P	3.70E+01	2.00E+02	0	0
⁶⁰ Co	3.70E+01	3.00E+04	6.08E-06	1.55E+00
⁹⁰ Sr	3.70E+01	2.00E+03	0	0
⁹⁹ Tc	3.70E+01	2.00E+02	4.54E-13	7.67E-10
¹³¹ I	3.70E+01	2.00E+02	0	0
¹³⁷ Cs	3.70E+01	3.00E+04	1.36E-06	3.45E-01
²²⁶ Ra	3.70E+01	5.00E+02	4.19E-06	1.77E-02
²³⁵ U	3.70E+01	5.00E+02	2.41E-07	1.02E-03
²³⁸ U	3.70E+01	5.00E+02	3.43E-08	1.45E-04
²³⁹ Pu	3.70E+01	1.00E+05	1.80E-10	1.52E-04
²⁴¹ Am	3.70E+01	3.00E+04	1.29E-08	3.27E-03

C_w = concentration in waterK_d = sediment distribution coefficientD_s = dose conversion factorD_h = annual mean effective dose equivalent from exposure to sediment

$$C_w \times K_d \times D_s \times 0.228$$

TABLE 7.4—Percentage dose contributed from each pathway

Radionuclide	D _f (Sv a ⁻¹)	D _w (Sv a ⁻¹)	D _s (Sv a ⁻¹)	D _t (Sv a ⁻¹)	D _f (%)	D _w (%)	D _s (%)
³ H	2.30E-08	4.59E-07	0	4.82E-07	4.76	95.24	0.00
³² P	2.81E-01	5.62E-05	0	2.81E-01	99.98	0.01	0.00
⁶⁰ Co	1.88E-04	1.88E-04	1.55E+00	1.55E+00	0.01	0.01	99.80
⁹⁰ Sr	2.43E-04	9.72E-04	0	1.22E-03	20.00	80.00	0.00
⁹⁹ Tc	6.85E-06	9.13E-06	7.67E-10	1.60E-05	42.8	57.1	0.000
¹³¹ I	2.92E-04	3.89E-04	0	6.81E-04	42.88	57.12	0.00
¹³⁷ Cs	7.29E-03	3.65E-04	3.45E-01	3.53E-01	2.06	0.11	97.83
²²⁶ Ra	2.06E-02	8.24E-03	1.77E-02	4.65E-02	44.30	17.72	38.06
²³⁵ U	9.24E-04	1.85E-03	1.02E-03	3.79E-03	24.40	48.81	26.91
²³⁸ U	8.54E-04	1.71E-03	1.45E-04	2.71E-03	31.51	63.10	5.50
²³⁹ Pu	5.62E-02	3.21E-03	1.52E-04	5.96E-02	94.29	5.38	.0026
²⁴¹ Am	2.01E-02	1.60E-02	3.27E-03	3.94E-02	51.02	40.61	8.30

D_t = Total dose rate from the three pathways

three pathways considered (i.e. D_w + D_f + D_s) may be used to calculate (by simple proportion) the concentration in water for each radionuclide which would result in a total dose rate of 1 mSv a⁻¹ to humans.

The dose to freshwater fish from a concentration of a radionuclide in its tissue and from the surrounding water can be calculated by using a computer code developed by Trubey and Kaye (1973). The computer code and the method for using it can be found in the

Canadian report on radioactivity in the environment (NRCC, 1983) and is described in Section 6.

Using output from the BIORAD computer code and the following equation, a dose was estimated for fish from internal emitters and external exposure based on the water concentration of the radionuclide which would deliver a total dose equivalent rate of 1 mSv as - 1 to humans.

$$D_w = 2.7 \times 10^{-10} [C_w BR_e + C_w BR_i] \text{ (mGy a}^{-1}\text{/Bq L}^{-1}\text{)} \quad (7-4)$$

Where

D_w is the total dose in (mGy a⁻¹)

C_w is the concentration of radionuclides in water (Bq L⁻¹)

BR_e is the BIORAD dose conversion factor for external dose (mGy a⁻¹) (Table 4.7; NRCC, 1983)

BR_i is the BIORAD dose conversion factor for internal dose (mGy a⁻¹) (Table 4.4; NRCC, 1983)

Results of the calculations for internal and external dose to fish are given in Table 7.5. Note that the BIORAD dose conversion factors have been converted to SI units (NRCC dose conversion factor times $2.7 \times 10^{-13} = \text{Gy a}^{-1}\text{/BqL}^{-1}$).

Many of the radionuclides released into aquatic ecosystems accumulate in sediment to such a degree that sediments are often referred to as sinks. Aquatic organisms in contact with the sediments receive a substantially higher dose from sediment than from other sources. To calculate a dose rate to aquatic organisms from sediment, the

TABLE 7.5—*External and internal absorbed dose to fish from water at a concentration that would produce a dose equivalent of 1 mSv a⁻¹ in humans for the sum of the three aquatic pathways considered*

Radionuclide	C_w (Bq mL ⁻¹)	External Dose (Gy a ⁻¹)	Internal Dose (Gy a ⁻¹)
³ H	7.68E+04	1.21E-03	2.95E-03
³² P	1.32E-01	2.32E-01	4.63E-02
⁶⁰ Co	2.40E-02	3.12E-07	3.62E-06
⁹⁰ Sr	3.04E+01	8.21E-05	8.21E-04
⁹⁹ Tc	2.31E+03	5.61E-04	1.62E-02
¹³¹ I	5.44E+01	1.34E-04	1.76E-03
¹³⁷ Cs	1.05E-01	3.48E-07	1.25E-04
²²⁶ Ra	7.96E-01	5.37E-08	2.15E-02
²³⁵ U	9.70E-00	1.42E-05	2.25E-02
²³⁸ U	1.37E+01	2.74E-04	2.96E-02
²³⁹ Pu	6.19E-01	1.39E-08	5.91E-02
²⁴¹ Am	9.37E-01	2.00E-07	6.83E-03

C_w = Nuclide concentration in water which would deliver a total dose equivalent rate of 1 mSv a⁻¹ to humans derived from Table 7.4.

concentration of the radionuclides in sediment is needed. In the present scenario the concentration was obtained by multiplying the derived water concentration in Table 7.4 by the distribution coefficient (K_d) for sediment. The K_d (concentration of the radionuclide in sediment/concentration of the radionuclide in water) varies with the composition of the sediment and also varies according to chemical state of the radionuclide (e.g., valence, complex status, etc.), and may not be constant with time even for a given radionuclide. Thus a range of K_d values would be more appropriate than a single value; however, for purposes of simplification for calculating a generic dose to aquatic biota, the K_d 's given in IAEA Safety Series No. 57 (IAEA, 1982) were used, unless more conservative values were justifiable.

Methods for calculating a dose rate to aquatic biota from β - and γ -radiation at the sediment-water interface are given in IAEA Technical Report Series No. 172, (IAEA, 1976) and Section 6 of this report. The following equations were used to calculate β - and γ -dose rates to aquatic biota at the sediment water interface:

$$D_s = (0.5) (5.76 \times 10^{-4}) E_i C_w K_d \quad (7-5)$$

where

D_s is the dose to organisms at the sediment-water interface ($\mu\text{Gy h}^{-1}$)

0.5 is the fraction of the dose rate from an infinite volume source at the sediment-water interface.

E_i is the mean energy per decay (MeV)

C_w is the concentration in water (Bq L^{-1})

K_d is the sediment distribution coefficient (L kg^{-1})

$$D_{\beta} = (0.5) (5.76 \times 10^{-4}) E_i C_w K_d \quad (7-6)$$

Where

D_{β} is the β -dose to the surface of the organism in contact with sediment ($\mu\text{Gy h}^{-1}$)

0.5 is the fraction of β -dose rate from an infinite volume source

E_i is the mean energy per decay (MeV)

C_w is the concentration of the water (Bq L^{-1})

K_d is the sediment distribution coefficient (L kg^{-1})

The estimated dose from β - and γ -radiation to aquatic biota at the sediment-water interface are given for the different radionuclides in Tables 7.6 and 7.7, respectively. For comparative purposes these doses are calculated on a yearly basis. The doses would be overestimations for fish because even bottom-dwelling fish do not spend all of their time in contact with the sediment. These doses would be more realistic for benthic organisms that spend most of their time

TABLE 7.6—*Estimate of absorbed dose to fish from beta radiation in sediments at a water concentration which would produce a dose equivalent of 1 mSv a⁻¹ in humans for the three pathways considered^a*

Radionuclide	C _w (Bq mL ⁻¹)	K _d (mL kg ⁻¹)	E _i (MeV)	Dose (Gy a ⁻¹)
³ H	7.68E+04	0	5.68E-03	0
³² P	1.32E-1	2.00E+02	6.95E-01	4.64E-05
⁶⁰ Co	2.46E-02	3.00E+04	9.65E-02	1.76E-04
⁹⁰ Sr	3.04E-1	2.00E+03	1.13E-00	1.74E-01
⁹⁹ Tc	2.31E+3	2.00E+02	1.01E-01	1.18E-01
¹³¹ I	5.44E+1	2.00E+02	1.90E-01	5.23E-03
¹³⁷ Cs	1.05E-1	3.00E+04	2.49E-01	1.98E-03
²²⁶ Ra	7.96E-01	5.00E+02	1.37E-00	1.38E-03
²³⁵ U	9.70E-00	5.00E+02	2.11E-01	2.59E-03
²³⁸ U	1.37E+01	5.00E+02	8.89E-01	1.54E-02
²³⁹ Pu	6.19E-01	1.00E+05	6.65E-03	1.04E-3
²⁴¹ Am	9.32E-01	3.00E+04	5.19E-02	3.69E-3

^aHypothetical organism (fish) is assumed to reside at the sediment-water interface

C_w = concentration of water
 K_d = sediment distribution coefficient
 E_i = mean energy per decay

TABLE 7.7—*Estimate of absorbed dose to fish from gamma radiation in sediments at a water concentration which would produce a dose of 1 mSv a⁻¹ in humans for the three pathways considered^a*

Radionuclide	C _w (Bq L ⁻¹)	K _d (Bq kg ⁻¹)	E _i (MeV)	Dose (Gy a ⁻¹)
³ H	7.68E+04	0	0	0
³² P	1.32E-01	2.00E+02	0	0
⁶⁰ Co	2.40E-02	3.00E+04	2.5E-00	4.55E-03
⁹⁰ Sr	3.04E+01	2.00E+03	0	0
⁹⁹ Tc	2.31E+03	2.00E+02	0	0
¹³¹ I	5.44E+01	2.00E+02	3.80E-01	1.05E-02
¹³⁷ Ca	1.05E-01	3.00E+04	5.64E-01	4.49E-03
²²⁶ Ra	7.96E-01	5.00E+02	1.72E-00	1.73E-03
²³⁵ U	9.70E-00	5.00E+02	1.80E-01	2.21E-03
²³⁸ U	1.37E+01	5.00E+02	2.78E-02	4.82E-04
²³⁹ Pu	6.19E-01	1.00E+05	7.96E-04	1.25E-04
²⁴¹ Am	9.37E-01	3.00E+04	3.24E-02	2.30E-3

^aHypothetical organism (fish) is assumed to reside at the sediment-water interface

C_w = concentration of water
 K_d = sediment distribution coefficient
 E_i = mean energy per decay

on or near the sediment surface. In the present scenario, however, we are considering irradiation of fish because some of the developmental embryonic stages of the teleosts are considered the most sensitive to radiation (Sections 2 and 3.)

The doses for α -radiation from sediments were not calculated. Alpha particles emitted by radionuclides in sediment do not contribute to whole body dose but can cause significant exposure at the skin and lining of the digestive system of those organisms that ingest sediments. External and internal doses for α -emitting radionuclides were included in the dose calculated for fish using the BIORAD computer code (Table 7.5). Doses to biota at the sediment-water interface would be $0.5 D(\infty)$, see Section 6 of this report. However, because this exposure would not contribute significantly to the whole body dose, and because the external and internal doses calculated with the BIORAD computer code are conservative estimates of the dose to fish, the contribution from α -emitting radionuclides in sediment was not included.

Another problem with calculating alpha dose for aquatic biota is the differential response to equal absorbed doses of different radiations. Quality factors (Q) have been determined to account for the differences in relative biological effectiveness (RBE) for α -, β -, and γ -radiation; however, these are currently defined only for the purpose of human radiation protection. Factors equivalent to Q for aquatic organisms are required in order to modify the calculated absorbed dose and thus give a measure of the biologically effective dose in aquatic organisms. Because the soft tissue compositions of humans and other organisms are generally similar in water content and basic cell structure, similar values of RBE would be expected for the different radiation types. However, because of the conservatism built into the BIORAD computer code, quality factors were not considered.

The sum of the total radiation doses to fish from water, internal emitters, and sediment for each radionuclide considered is given in Table 7.8. Of these radionuclides the largest estimated dose received by fish is from ^{90}Sr and the greatest percentage of the dose from ^{90}Sr is from β -radiation from sediment. As stated previously, this is an overestimation of dose to fish from sediment because the fish would not be in constant contact with the sediment. In addition, a less conservative model, for example the model described by Woodhead in IAEA (1976) and also discussed in Section 6 would reduce the estimated dose rate to fish still further.

The limiting value of the contamination of the environment is inversely related to the intensity of its use by humans. In other words, in estimating radiation doses to humans, the more aquatic pathways considered and the greater the consumption of aquatic food the smaller the estimated dose will be to aquatic organisms in the contaminated environment. If the estimated dose to humans was calculated alternatively on the basis of a lower consumption rate (20 g d^{-1}) of fish, the estimated dose to the biota would be greater

TABLE 7.8—Estimated absorbed dose to fish inhabiting an environment in which humans would receive a dose equivalent of 1 mSv a^{-1} ^a

Radionuclide In Water	Concentration Fish (Bq kg ⁻¹)	Biorad internal dose (Gy a ⁻¹)	Biorad external dose (Gy a ⁻¹)	External dose from sediment		Total dose rate (Gy a ⁻¹)
				gamma radiation (Gy a ⁻¹)	beta radiation (Gy a ⁻¹)	
³ H	7.68E+04	3.95E-03	1.21E-03	0	0	5.16E-03
³² P	1.32E+04	4.63E-02	2.32E-07	0	4.64E-05	4.63E-02
⁶⁰ Co	4.80E-01	3.62E-06	3.12E-07	4.55E-03	1.76E-04	4.73E-03
⁹⁰ Sr	1.52E-02	8.21E-04	8.21E-05	0	1.74E-01	1.74E-01
⁹⁹ Tc	3.49E+04	1.62E-02	5.61E-04	0	1.18E-01	1.35E-01
¹³¹ I	8.16E-02	1.76E-03	1.34E-04	1.05E-02	5.23E-03	1.76E-02
¹³⁷ Cs	4.20E-01	1.25E-04	3.46E-07	4.49E-03	1.98E-03	6.60E-03
²²⁶ Ra	3.98E-01	2.15E-02	5.37E-08	1.73E-03	1.38E-03	2.46E-02
²³⁵ U	9.70E-01	2.25E-02	1.42E-05	2.21E-03	2.59E-03	2.73E-02
²³⁸ U	1.37E-02	2.96E-02	2.74E-04	4.82E-04	1.54E-02	4.58E-02
²³⁹ Pu	2.17E-02	5.91E-02	1.39E-08	1.25E-04	1.04E-03	6.02E-02
²⁴¹ Am	2.34E-02	6.83E-03	2.00E-07	2.30E-03	3.69E-03	1.28E-02

^aFrom consuming contaminated fish and drinking water and being exposed to external radiation from sediments

than if 100 g d^{-1} of fish were being consumed or if additional pathways were being considered. However, the assessment of radiation doses to humans carried out in establishing standards for releases of radionuclides to aquatic ecosystems, is usually based on maximum potential exposures from all pathways.

In the present scenario in which humans receive a limiting dose of 1 mSv a^{-1} from an aquatic environment, the maximum estimated dose to fish for a single radionuclide was 174 mGy a^{-1} (0.5 mGy d^{-1}) (Table 7.8). In the more typical case where a mixture of radionuclides is present in the environment and delivers a combined dose rate of 1 mSv a^{-1} to man, the dose rate to fish would be correspondingly less.

Based on the recommendations in Section 5 of this report and the conclusion that a dose of 10 mGy d^{-1} or less will not have a detectable deleterious effect on aquatic populations, it can be concluded that under these conditions, if man is protected by limiting the exposure via the aquatic pathways to 1 mSv a^{-1} , populations of aquatic organisms in this environment should also be protected from deleterious effects of radiation.

8. Conclusions

The discharge of the low-level radioactive effluents into the aquatic environment has resulted in chronic, low dose rate exposure to aquatic organisms. The fate of individual organisms is, generally, not the major concern but rather the response and maintenance of endemic populations.

Experimental studies to date have shown that fertility and fecundity (gametogenesis) of the organisms and embryonic development are probably the most sensitive components of the radiation response, and it is precisely these attributes which are of importance in determining the fate of the population.

It must be recognized that increased radiation exposure is but one of the many stresses imposed upon aquatic populations by human activities. However, determination of the mode of interaction of radiation, whether it be antagonistic, additive, or synergistic, with other environmental contaminants or stressors, is extremely difficult to assess under conditions of chronic exposure.

The review of the three dosimetry models and associated codes given in Section 6 indicate that they are all appropriate for estimating radiation dose rates to aquatic organisms. CRITR would appear to be the most useful approach to adopt in any preoperational assessment phase of planned waste management activities that will result in radionuclides being released to the aquatic environment. The Point Source Dose Distribution (PSDD) functions require considerable detail on radionuclide distributions to realize their full potential and would be most appropriately applied in contaminated environments where data have been collected specifically for environmental dosimetry. These functions are particularly useful for application to small sized organisms, (e.g., developing embryos), or where it is necessary to estimate the dose rate to a radiosensitive tissue, such as the gonads, from a non-uniform radionuclide distribution. BIORAD is less useful since the potentially significant exposure of an organism to contaminated sediments is not considered.

The majority of the estimated whole body doses received by aquatic organisms at all the sites examined was more than two orders of magnitude below the proposed standard of 0.4 mGy h^{-1} (Section 5). However there are a few maximum values that approach that value and exceed the dose of 0.1 mGy h^{-1} at which it is proposed that a

more detailed evaluation be undertaken. These occur in man-made bodies of water associated with waste management activities and have no direct connection with bodies of natural water at these sites.

For example, the internal dose values from ^{90}Sr in turtles exposed in seepage basins at the Savannah River Plant exceeded the proposed evaluation dose. However, these seepage basins have been fenced to preclude the ingress or egress of turtles, and a permanent closure plan for these basins is being developed. At Gable Mountain Pond at the Hanford Plant, insects, molluscs and small fish externally exposed to ^{137}Cs from contaminated sediments also exceeded the proposed evaluation dose. Again remediation and restoration action has been taken; the pond has been drained and the sediments covered with clean soil.

From the information reviewed in this report, it appears that a chronic dose rate of no greater than 0.4 mGy h^{-1} (1 rad d^{-1}) to the maximally exposed individual in a population of aquatic organisms would ensure protection for the population. If modelling and/or dosimetric measurements indicate a level of 0.1 mGy h^{-1} , then a more detailed evaluation of the potential ecological consequences to the endemic population should be conducted.

Radiation dose rates to aquatic biota were estimated by using a combination of aquatic pathways for which the total exposure is 1 mSv a^{-1} . These estimates indicate that the ICRP statement, "if man is adequately protected then other living things are also likely to be sufficiently protected," is reasonable, at least within the generic scenario considered here. While a value of 1 mSv for human exposure was used in the scenario for estimating these doses, the prevailing radiation protection philosophy is to limit human exposure to well below the acceptable levels under the ALARA principle (As Low as Reasonably Achievable). Such a philosophy also minimizes the impact upon the natural population of aquatic organisms sharing the environment with man.

Appendix A

Dose Rate Estimates to Aquatic Biota at Example Sites.

There are a number of sites, three of which are DOE-operated and one in Canada, which can be used as examples of the application of dosimetry methods and models described in Section 6 for the assessment of the potential radiation doses to aquatic organisms. These are:

Gable Mountain Pond, Hanford Plant, Washington
White Oak Lake, Oak Ridge National Laboratory, Tennessee
Pond B, Savannah River Plant, South Carolina
Beaverlodge Uranium Mining Area, Saskatchewan, Canada

It must be emphasized that the data available in the literature on the distribution and concentrations for the radionuclides in these environments were not collected specifically for the purposes of dosimetry. This has required that a number of assumptions be made, generally in the direction of simplification. The data are insufficiently detailed to permit the estimation of dose rates to specific target organs (e.g., the gonads), and the whole body is taken to be the target. In the absence of indications to the contrary, the radionuclide concentrations given for organisms have been assumed to be mean whole body concentrations. Where concentrations have been given for specific tissues, these have been converted to mean whole body concentrations using simple assumptions. In the case of fish from White Oak Lake, the given sample weights have been converted to proportions of body weight; for the fish from Beaverlodge Lake, the skin, muscle, bone, liver, and gonad have been assumed to be the following proportions, respectively, of the whole body weight—0.01, 0.50, 0.29, 0.1 and 0.1. Where the radionuclide concentration has been given in terms of sample dry weight, these have been converted to fresh weight using the following dry/wet weight values:

sediment	0.75
soft tissue, e.g., fish fillet, bird muscle	0.20
molluscs	0.20
bird carcass, less muscle	0.40

For Gable Mountain Pond there are no data for water, but the concentrations for a variety of radionuclides are given for sediment, goldfish, molluscs, insects and birds (muscle:carcass—0.2:0.8) (Cushing and Watson, 1974). These are summarized in Table A.1.

Data have been published for radionuclide concentrations in water, fish and sediment from White Oak Lake (Blaylock, 1987). For fish, the data refer to one or more of the following tissues from carp, bluegill, or largemouth bass. Where data are given for muscle (fillet) alone, this has been assumed to be equivalent to the mean whole body concentration. In bluegill, the activity in the gut has been adjusted to an equivalent mean whole body concentration assuming that the gut is 5.8% of the body mass. For ^{90}Sr , the concentration in the carcass (the major part of the body) is greater than that in the fillet, and the former has been conservatively assumed to be the required whole body value. The data are given in Table A.2.

The data available for Pond B and the Seepage Basins at Savannah River Plant are summarized in Table A.3 (Brisbin, 1988; Whicker et al., 1989).

For the mine tailings sites in northern Saskatchewan data have been published for certain of the natural decay series radionuclides in a variety of environmental materials (Swanson, 1985). Information for two sites, Beaverlodge Lake and Tailings Creek, has been selected to be the basis of estimates of the radiation dose rates prevailing in the region. Because data are given only for ^{210}Pb , ^{226}Ra and total uranium, a number of assumptions have to be made concerning the state of equilibrium between these longer-lived radionuclides and the various daughters. These assumptions vary with the environmental material under consideration, and are summarized in Table A.4, along with the derived concentrations in the environmental materials.

The geometrical models which have been adopted to represent the various organisms for the application of the point source dose distribution functions are summarized in Table A.5. The derived absorbed fractions, F , as a function of γ -ray energy are given in Figs. A.1 through 5. As noted in Section 6.1, the dose rate from internal contamination is

$$D_{\gamma} = 5.76 \times 10^{-4} E_{\gamma} n_{\gamma} \Phi \text{ (}\mu\text{Gy h}^{-1}\text{/Bq kg}^{-1}\text{)} \quad (\text{A-1})$$

It follows that the dose rate from the nuclides in water away from the sediment is

$$D_{\gamma} = 5.76 \times 10^{-4} E_{\gamma} n_{\gamma} (1 - \Phi) \text{ (}\mu\text{Gy h}^{-1}\text{/Bq kg}^{-1}\text{)} \quad (\text{A-2})$$

TABLE A.1—Radionuclide concentrations at Gable Mountain Pond, Hanford

Material	Nuclide	Published concentrations pCi g ⁻¹ dry	Derived concentrations Bq kg ⁻¹ wet
Sediment 0–15 cm	¹³⁷ Cs	$8.80 \times 10^{-1} - 7.96 \times 10^4$	$2.44 \times 10^3 - 2.21 \times 10^6$ ($1.11 \times 10^5 - 1.01 \times 10^8$ Bq m ⁻²)
	²³⁸ Pu	$5.00 \times 10^{-4} - 8.80 \times 10^{-2}$	$1.39 \times 10^{-2} - 2.44$
	^{239/240} Pu	$1.10 \times 10^{-2} - 8.00$	$3.05 \times 10^{-1} - 2.22 \times 10^2$
	²⁴¹ Am	$9.00 \times 10^{-3} - 1.40$	$2.50 \times 10^{-1} \times 3.89 \times 10^1$
Goldfish	⁹⁰ Sr	1.00 – 3.50	$7.40 - 2.59 \times 10^1$
	¹³⁷ Cs	$6.48 \times 10^1 - 1.66 \times 10^3$	$4.80 \times 10^2 - 1.23 \times 10^4$
	²³⁸ Pu	8.00×10^{-4}	5.92×10^{-3}
	^{239/240} Pu	7.00×10^{-3}	5.18×10^{-2}
	²⁴¹ Am	7.00×10^{-3}	5.18×10^{-2}
Molluscs	¹³⁷ Cs	$1.37 \times 10^1 - 3.70 \times 10^3$	$1.01 \times 10^2 - 2.74 \times 10^4$
	²³⁸ Pu	2.80×10^{-3}	2.07×10^{-2}
	^{239/240} Pu	3.30×10^{-2}	2.44×10^{-1}
	²⁴¹ Am	4.60×10^{-2}	3.40×10^{-1}
Insects			
Large	¹³⁷ Cs	$7.30 - 1.43 \times 10^1$ (pCi g ⁻¹ wet)	$2.70 \times 10^2 - 5.29 \times 10^2$
Small	¹³⁷ Cs	$3.10 - 2.78 \times 10^2$	$2.29 \times 10^1 - 2.06 \times 10^6$
Duck			
Muscle	¹³⁷ Cs	$3.20 - 1.88 \times 10^2$	Average whole body
Carcass	¹³⁷ Cs	$8.00 \times 10^1 - 7.89 \times 10^1$	$1.42 \times 10^1 - 1.21 \times 10^3$
Coot			
Muscle	¹³⁷ Cs	$1.23 \times 10^1 - 1.15 \times 10^3$	Average whole body
Carcass	¹³⁷ Cs	$2.96 \times 10^1 - 8.68 \times 10^2$	$3.69 \times 10^2 - 1.20 \times 10^4$

*See page A-1 for dry/wet conversion factors.

TABLE A.2—Radionuclide concentrations at White Oak Lake, Oak Ridge

Material	Nuclide	Published concentrations pCi L ⁻¹	Derived concentrations Bq L ⁻¹
Water	³ H	3.50×10^5	1.30×10^4
	⁶⁰ Co	6.30×10^1	2.33
	⁹⁰ Sr	3.00×10^2	1.11×10^1
	¹³⁷ Cs	4.20×10^1	1.55
Fish		dpm g ⁻¹ (except where noted)	Bq kg ⁻¹ wet
Carp fillet	⁶⁰ Co	$0.00 - 6.60 \times 10^{-1}$	$0.00 - 1.10 \times 10^1$
	¹³⁷ Cs	$4.12 - 3.03 \times 10^1$ Bq kg ⁻¹	$6.87 \times 10^1 - 5.05 \times 10^2$
	⁹⁰ Sr	$1.60 - 8.30 \times 10^1$	$1.60 - 8.30 \times 10^1$
Bluegill fillet gut fillet gut fillet carcass	⁶⁰ Co	0.00 - 1.78	Average whole body
	⁶⁰ Co	$5.40 \times 10^{-1} - 2.82 \times 10^1$	$5.22 \times 10^{-1} - 5.70 \times 10^1$
	¹³⁷ Cs	$1.12 \times 10^1 - 3.85 \times 10^1$	$1.97 \times 10^2 - 7.64 \times 10^2$
	¹³⁷ Cs	$1.15 \times 10^1 - 1.27 \times 10^2$ Bq kg ⁻¹	
	⁹⁰ Sr	$9.70 - 3.20 \times 10^1$	$5.90 \times 10^2 - 1.10 \times 10^3$
	⁹⁰ Sr	$5.90 \times 10^2 - 1.10 \times 10^3$	
Largemouth Bass fillet	⁶⁰ Co	0.00 - 1.4	$0.00 - 2.33 \times 10^1$
	¹³⁷ Cs	$2.35 \times 10^1 - 9.38 \times 10^1$	$3.92 \times 10^2 - 1.56 \times 10^3$
	⁹⁰ Sr	$3.00 - 1.80 \times 10^1$	$3.00 - 1.80 \times 10^1$
Sediment 0–20 cm		pCi g ⁻¹ wet	
	⁶⁰ Co	1.50×10^2	5.55×10^3 (2.53×10^5 Bq m ⁻²)
	¹³⁷ Cs	1.55×10^3	5.75×10^4 (2.62×10^6 Bq m ⁻²)

TABLE A.3—Radionuclide concentrations at the B Pond and Seepage Basins, Savannah River

Material	Nuclide	Published concentrations Bq L ⁻¹	Derived concentrations Bq L ⁻¹
Water	⁹⁰ Sr	1.40×10^{-1}	1.40×10^{-1}
	¹³⁷ Cs	7.60×10^{-1}	7.60×10^{-1}
	²³⁸ Pu	2.20×10^{-5}	2.20×10^{-5}
	^{239/240} Pu	8.80×10^{-6}	8.80×10^{-6}
	²⁴¹ Am	2.30×10^{-5}	2.30×10^{-5}
	²⁴⁴ Cm	6.40×10^{-4}	6.40×10^{-4}
Benthic invertebrates		pCi g ⁻¹	Bq kg ⁻¹
insect larvae	¹³⁷ Cs		$4.75 \times 10^1 - 1.76 \times 10^3$
Molluscs	¹³⁷ Cs	5.00	1.85×10^2
Fish	¹³⁷ Cs	$5.50 \times 10^1 - 2.20 \times 10^2$	$2.04 \times 10^3 - 8.14 \times 10^3$
Birds	¹³⁷ Cs	3.98×10^1	1.47×10^3
Alligators	¹³⁷ Cs	1.32×10^1	4.89×10^2
Turtles			
Pond B	¹³⁷ Cs	$1.00 \times 10^1 - 4.47 \times 10^1$	$3.70 \times 10^2 - 1.65 \times 10^3$
	⁹⁰ Sr	$1.50 \times 10^1 - 4.81 \times 10^1$	$5.55 \times 10^2 - 1.78 \times 10^3$
Seepage basin	¹³⁷ Cs	$6.48 - 2.61 \times 10^2$	$2.40 \times 10^2 - 9.64 \times 10^3$
	⁹⁰ Sr	$2.19 \times 10^2 - 1.00 \times 10^4$	$8.09 \times 10^3 - 3.70 \times 10^6$
		Bq g ⁻¹ dry	Bq kg ⁻¹ wet
Sediment surface	¹³⁷ Cs	3.00×10^1	2.25×10^4 (1.02×10^6 Bq m ⁻²)
Average 0–11 cm and 0–17 cm	⁹⁰ Sr	2.50×10^{-1}	1.88×10^2 (8.55×10^3 Bq m ⁻²)

TABLE A.4—Radionuclide concentrations at Beaverlodge Lake and Tailings Creek, Saskatchewan

TABLE 2.7. Radionuclide concentrations at Beaverlodge Lake and Tailings Creek, Saskatchewan				
Material	Nuclide	Concentrations		Assumptions made concerning daughter radionuclides
		Beaverlodge Lake	Tailings Creek	
Bq L⁻¹				
Water	²¹⁰ Pb	3.0×10^{-2}	3.1	²¹⁰ Bi and ²¹⁰ Po in equilibrium Daughters ²²² Ra to ²¹⁴ Po present at the concentration of ²¹⁰ Pb given above, i.e. there has been some loss of radon ²³⁴ Th and ²³⁴ U in equilibrium ²³¹ Th in equilibrium
	²²⁶ Ra	6.0×10^{-2}	4.3	
	²³⁸ U	4.17	5.31×10	
	²³⁵ U	1.95×10^{-1}	2.48	
Bq kg⁻¹				
Sediment	²¹⁰ Pb	1.58×10^3	1.04×10^3	²¹⁰ Bi and ²¹⁰ Po in equilibrium Daughters ²²² Rn to ²¹⁴ Po in equilibrium Daughters ²³⁴ Th to ²³⁰ Th in equilibrium Daughters ²³¹ Th to ²⁰⁷ Te in equilibrium
	²²⁶ Ra	9.5×10^2	8.2×10^2	
	²³⁸ U	1.35×10^3	1.5×10^2	
	²³⁵ U	6.29×10^{-1}	4.02	
Small insects and larvae	²¹⁰ Pb	—	$4.0 \times 10^1 - 7.2 \times 10^2$	²¹⁰ Bi and ²¹⁰ Po in equilibrium Assume complete loss of ²²² Rn and no daughters to ²¹⁴ Po present ²³⁴ U alone present at equilibrium No daughters present
	²²⁶ Ra	—	$7.0 - 8.5 \times 10^2$	
	²³⁸ U	—	$3.26 \times 10^1 - 3.16 \times 10^2$	
	²³⁵ U	—	$1.52 - 1.48 \times 10^1$	
Small fish (whole)	²¹⁰ Pb	1.0	2.0×10^1	²¹⁰ Bi and ²¹⁰ Po in equilibrium Daughters ²²² Rn to ²¹⁴ Po present at one half the equilibrium value, i.e. there is a 50% loss of radon ²³⁴ Th and ²³⁴ U in equilibrium ²³¹ Th in equilibrium
	²²⁶ Ra	4.0	$2.0 \times 10^1 - 6.0 \times 10^1$	
	²³⁸ U	7.66	$2.43 \times 10^1 - 3.30 \times 10^1$	
	²³⁵ U	3.57×10^{-1}	1.13 - 1.54	
Large fish				
	Skin			
Muscle	²¹⁰ Pb	$4.2 - 1.1 \times 10^1$	—	
	²¹⁰ Pb	1.0 - 1.3	—	

Bone	^{210}Pb	$3.0 \times 10^1 - 4.0 \times 10^1$	—	^{210}Bi in equilibrium
Liver	^{210}Pb	$2.0 \times 10^1 - 1.0 \times 10^2$	—	
Gonad	^{210}Pb	$2.0 \times 10^1 - 6.0 \times 10^1$	—	
<u>Bq kg⁻¹</u>				
Equivalent whole body concentration for γ -ray dosimetry		$1.32 \times 10^1 - 2.84 \times 10^1$	—	
Gonad	^{210}Po	$7.0 - 2.0 \times 10^1$	—	
Skin	^{226}Ra	$5.0 - 2.4 \times 10^1$	—	Daughters ^{222}Rn to ^{214}Po present at one half the equilibrium value, i.e. there is a 50% loss of radon
Muscle	^{226}Ra	$2.0 \times 10^{-1} - 1.1$	—	
Bone	^{226}Ra	$3.0 \times 10^1 - 8.0 \times 10^1$	—	
Liver	^{226}Ra	$5.5 - 8.0$	—	
Gonad	^{226}Ra	$1.0 \times 10^1 - 3.0 \times 10^1$	—	
Equivalent whole body concentration for γ -ray dosimetry		$1.04 \times 10^1 - 2.78 - 10^1$	—	
Skin	^{238}U	$2.4 - 3.7$	—	
Muscle	^{238}U	$7.41 \times 10^{-1} - 9.88 \times 10^{-1}$	—	^{234}Th and ^{234}U in equilibrium
Bone	^{238}U	$1.63 \times 10^2 - 1.69 \times 10^2$	—	
Liver	^{238}U	$2.56 \times 10^1 - 4.64 \times 10^1$	—	
Gonad	^{238}U	$5.93 - 8.15$	—	
Equivalent whole body concentration for γ -ray dosimetry		$5.0 \times 10^1 - 5.6 \times 10^1$	—	
Skin	^{235}U	$1.12 - 1.74$	—	
Muscle	^{235}U	$3.46 \times 10^{-2} - 4.61 \times 10^{-2}$	—	
Bone	^{235}U	$7.58 - 7.87$	—	^{231}Th in equilibrium
Liver	^{235}U	$1.19 - 2.17$	—	
Gonad	^{235}U	$2.76 \times 10^{-1} - 3.80 \times 10^{-1}$	—	
Equivalent whole body concentration for γ -ray dosimetry		$2.38 - 2.59$	—	

TABLE A.5—Geometrical models adopted to represent the organisms for the Point Source Dose Distribution method

Organism	Mass kg	Length of the major axes of the ellipsoid cm
Small insects and larvae at Gable Mountain Pond, Savannah River, Beaverlodge Lake and Tailings Creek	1.6×10^{-5}	$0.62 \times 0.31 \times 0.16$
Large insects and molluscs at Gable Mountain Pond and molluscs at Savannah River	1.0×10^{-3}	$2.5 \times 1.2 \times 0.62$
Goldfish at Gable Mountain Pond and small fish at Beaverlodge Lake and Tailings Creek	2.0×10^{-3}	$3.1 \times 1.6 \times 0.78$
Large fish at White Oak Lake and Beaverlodge Lake	1.0	$45 \times 8.7 \times 4.9$
Alligators at Savannah River	2.0	$57 \times 11 \times 6.1$
Turtles at Savannah River	1.0	$17 \times 13 \times 8.6$
Duck and coot at Gable Mountain Pond	5.5×10^{-1} at a density of 0.8 g cm ⁻³ for solid tissue; plus 5×10^{-2} kg feathers for a density of 0.33 g cm ⁻³ ; overall dimensions	$15 \times 11 \times 7.6$ $21 \times 16 \times 11$

and at the sediment-water interface is

$$D_{\gamma} = 2.88 \times 10^{-4} E_{\gamma} n_{\gamma} (1 - \Phi) (\mu\text{Gy h}^{-1}/\text{Bq kg}^{-1}) \quad (\text{A-3})$$

Due to radioactive decay, the relatively short duration of the discharges and limited mixing in the river or lake bed, the sediment rarely represents a uniformly contaminated, semi-infinite source of γ -radiation from waste radionuclides. Thus, the dose rate from the sediment at the sediment-water interface is less than

$$D_{\gamma} = 2.88 \times 10^{-4} E_{\gamma} n_{\gamma} (1 - \Phi) (\mu\text{Gy h}^{-1}/\text{Bq kg}^{-1}) \quad (\text{A-4})$$

In those cases where the distribution of the nuclides within the sediment column has been considered, it has been found that the dose rate is given approximately by:

$$D_{\gamma} = 1.44 \times 10^{-4} E_{\gamma} n_{\gamma} (1 - \Phi) (\mu\text{Gy h}^{-1}/\text{Bq kg}^{-1}) \quad (\text{A-5})$$

i.e., $0.25 \times D(\infty)$. In those cases where detailed information concern-

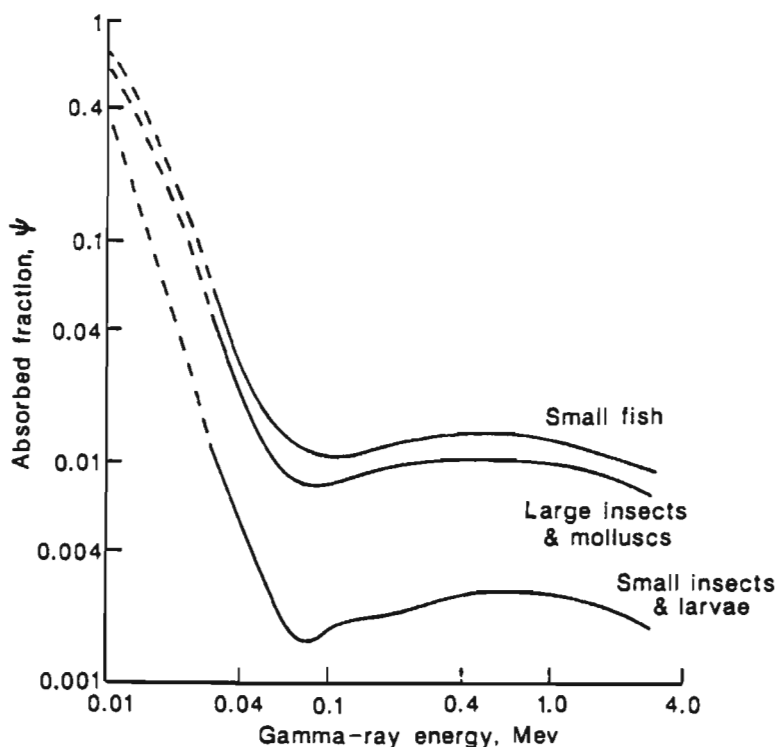


Fig. A.1 Derived absorbed fractions as a function of γ -ray energy (small fish, large insects and molluscs, and small insects and larvae).

ing the distribution of the nuclides in the sediment is available, this can be accounted for explicitly (IAEA, 1976; Woodhead, 1984).

The point source β -dose function described in Section 6.1 has been integrated over the geometries given in Table A.5 assuming a uniform nuclide distribution to obtain the dose rate at the center as a fraction of $D_{\beta}(\infty)$. The results are shown in Fig. A.6 as a function of maximum β -particle energy for the three smaller geometries; for the four larger geometries, the β -dose rate at the center is $D_{\beta}(\infty)$ independent of β -particle energy.

Application of an α -dose distribution function requires information on the micro-distribution of the α -emitting nuclides on a spatial scale of tens of μm ; this is rarely available and the α -dose rate is taken to be $D_{\alpha}(\infty)$.

In the case of the model adopted to represent the birds, it is assumed that all the contamination is in the solid tissue and the lower average tissue density is taken into account (Woodhead, 1986).

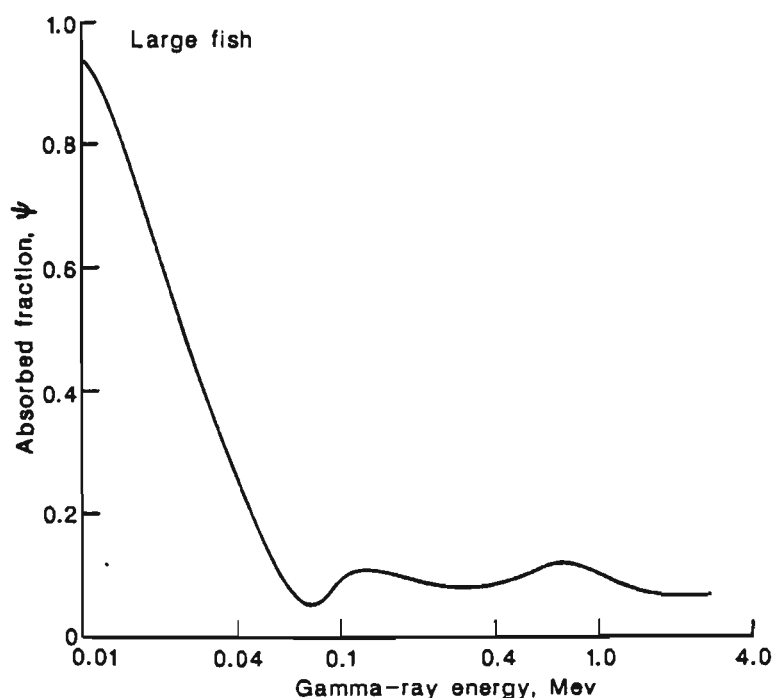


Fig. A.2 Derived absorbed fractions as a function of γ -ray energy (large fish).

Application of dosimetry models

Gable Mountain Pond

No discharge data are available for this site so that it is not possible to apply, in detail, all the CRITR methods described in Section 6.1. However, estimates of the dose rates from internal contamination and radionuclides in the sediment can be made from the data given in Table A.1. The results are given in Table A.6. The estimates of the dose rates derived from the point source dose functions are also given in Table A.6.

The application of the EXREM III and BIORAD approach requires values for the concentrations of the radionuclides in the water. Because these data are not available for Gable Mountain Pond this method cannot be applied.

There is good agreement between the two approaches set out in Table A.6 which can be applied in most instances. The discrepancies in the results for the external dose rate from sediment (a factor of 2) are simply a consequence of the assumptions concerning the source

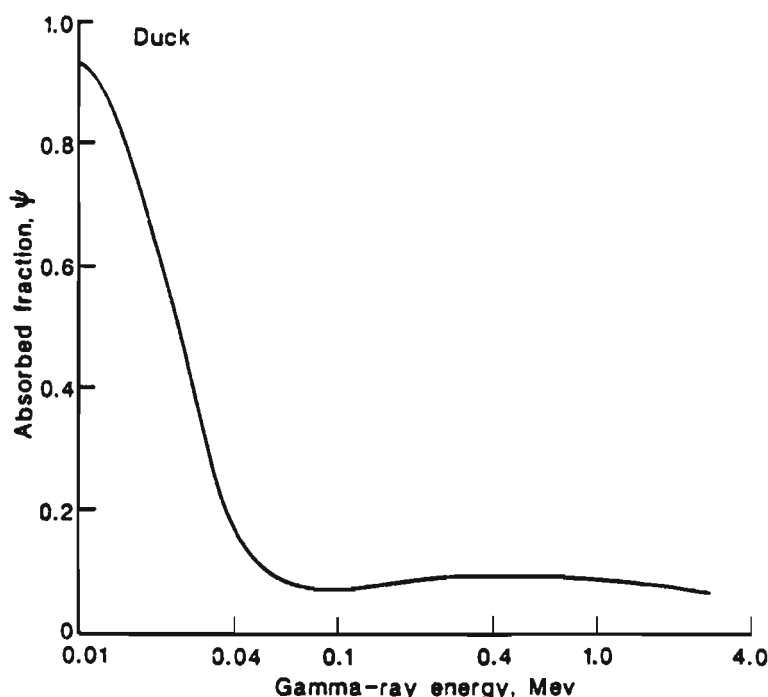


Fig. A.3 Derived absorbed fractions as a function of γ -ray energy (duck).

distribution i.e., a uniformly contaminated, surface layer of thickness 2.5 cm with zero concentration at greater depths (CRITR), and a depth-dependent source distribution for which, approximately

$$D_{\gamma} = 0.25 \times D_{\gamma}(\infty) \text{ [point source dose functions]}. \quad (\text{A-6})$$

For the small insects, the latter approach also includes a contribution from β -particles which is not included in CRITR.

It should also be noted that the dose rates estimated for the small organisms are of the same order as those measured by in-situ thermoluminescent dosimeters (Cushing and Watson, 1974).

White Oak Lake

The results of applying the three dosimetry approaches to the data available for White Oak Lake are given in Table A.7. The dose rate to the muskrat from contaminated water is based on the assumption that the animal spends one-quarter of its time swimming at the

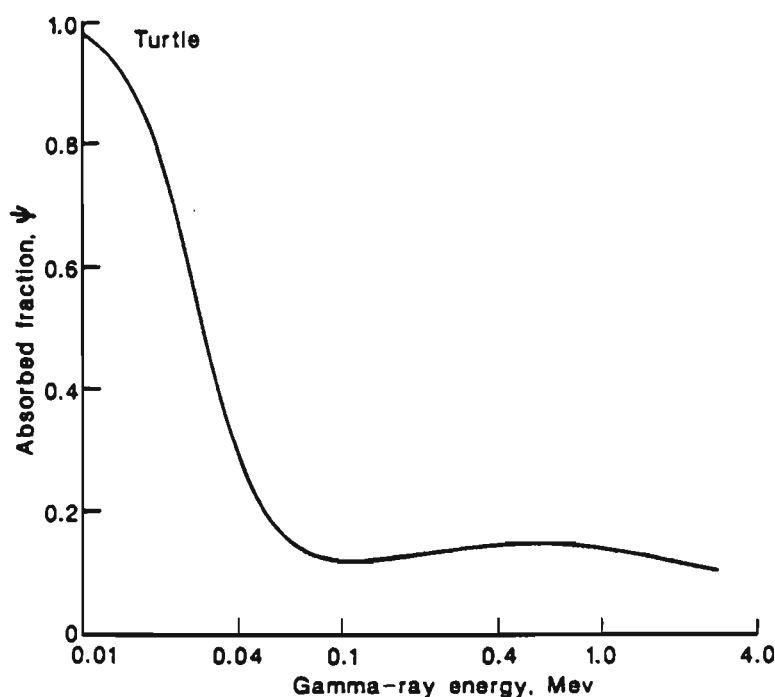


Fig. A.4 Derived absorbed fractions as a function of γ -ray energy (turtle).

surface, thus the dose factors provided by EXREM III and BIORAD have been reduced by a factor of eight. In most cases the agreement between the three methods is fairly good. The most notable exceptions are the dose rates to fish from internal contamination with ^3H and ^{60}Co where the results from EXREM III and BIORAD appear high relative to the results from the other two approaches.

Savannah River

The estimates of the dose rates to the animals in the Savannah River environment are given in Table A.8. In applying the CRITR method the effective radii of ducks, turtles and alligators have been assumed to be 5 cm, 5 cm, and 10 cm, respectively. Where comparisons are possible, there is a substantial measure of agreement between the results obtained from the three different approaches. Exceptions are the dose rates to fish from internal ^{137}Cs and to molluscs from ^{90}Sr in water and sediment.

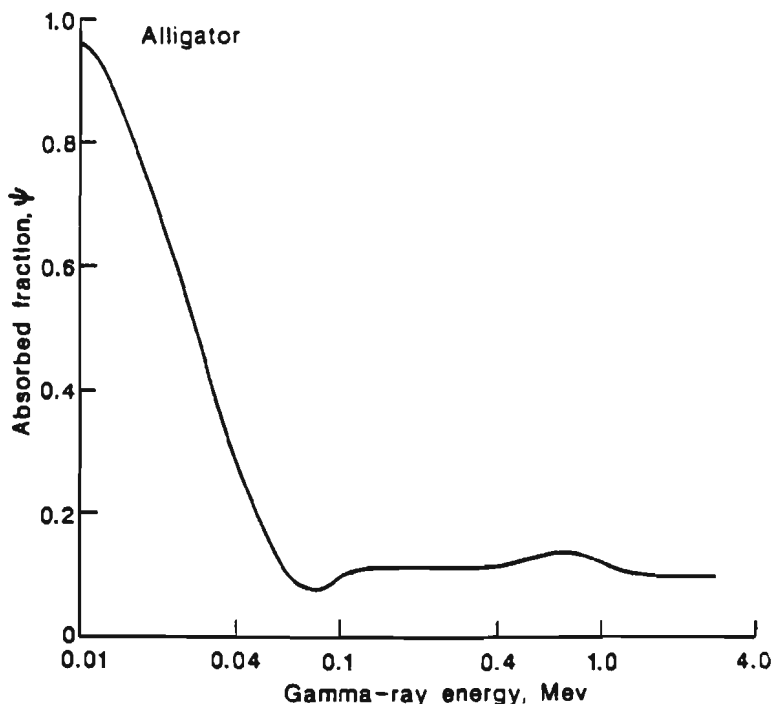
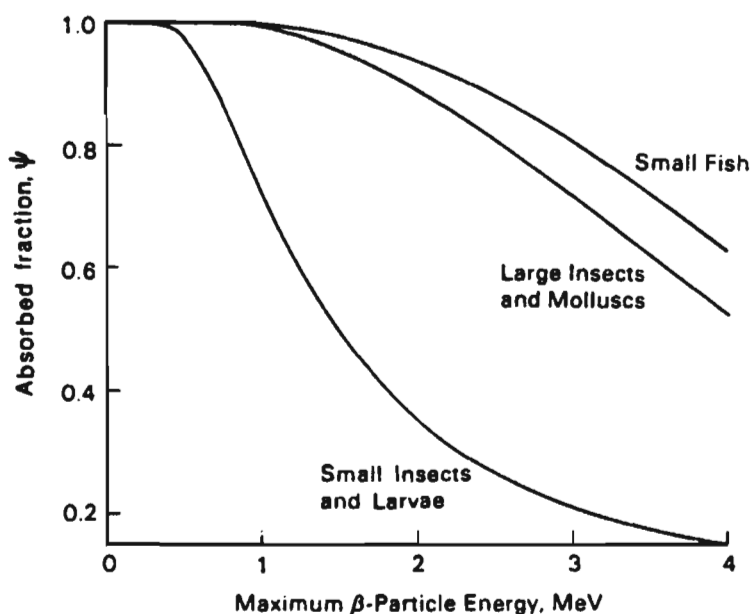


Fig. A.5 Derived absorbed fractions as a function of γ -ray energy (alligator).

Beaverlodge Lake and Tailings Creek

The CRITR method cannot be applied in this situation because dose factors are not supplied for the series of natural radionuclides. These dose factors are provided by EXREM III and BIORAD; however, the assumptions underlying the values given are probably inapplicable at this site. In some cases, the dose factors (e.g., ^{238}U) are inconsistent with the explicit assumptions described in the text concerning the state of equilibrium attained within decay series over the 30-year period of discharge. For these reasons the dose estimates given in Table A.9 are based on point source dose functions alone. Clearly, the results obtained for these natural series radionuclides are very dependent upon the assumptions made concerning the state of radioactive equilibrium pertaining at each step in the decay chain and this conclusion underlines the necessity for a complete set of radionuclide concentration data if credible estimates of the dose rate are to be made.

All the results presented in Tables A.6 through A.9 are in terms of the absorbed dose; however, some of the dose is delivered by



α -particles which might be considerably more effective in producing biological damage. The limited data available indicate that the relative biological effectiveness of various radiation types is similar in aquatic organisms to that found for mammals (Woodhead, 1984).

Thus it seems reasonable to assume that a quality factor of 20 could be applied to the absorbed dose from α -particles to obtain an estimate of the biologically effective dose. This would have a particularly significant effect on the magnitude of the total biologically effective dose rate at the mine tailings site in northern Saskatchewan.

TABLE A.6—Estimates of dose rates at Gable Mountain Pond, Hanford*

Source	Method	Organism	Nuclide	CRTR $\mu\text{Gy h}^{-1}$	EXREM III & BIORAD $\mu\text{Gy h}^{-1}$	Point Source Dose Functions $\mu\text{Gy h}^{-1}$
Internal		Small insects	^{137}Cs	N/A	N/A	$4.7 \times 10^{-3} - 4.2 \times 10^{-1}$
		Large insects	^{137}Cs	NA	N/A	$4.0 \times 10^{-2} - 7.8 \times 10^{-2}$
		Molluscs	^{137}Cs	$1.5 \times 10^{-2} - 4.1$	N/A	$1.5 \times 10^{-2} - 4.1$
			^{238}Pu	6.6×10^{-5}	N/A	6.7×10^{-5}
			$^{239/240}\text{Pu}$	7.2×10^{-4}	N/A	7.4×10^{-4}
			^{241}Am	1.1×10^{-3}	N/A	1.1×10^{-3}
		Small fish	^{137}Cs	$7.1 \times 10^{-2} - 1.8$	N/A	$7.2 \times 10^{-2} - 1.8$
			^{90}Sr	$4.9 \times 10^{-3} - 1.7 \times 10^{-2}$	N/A	$4.4 \times 10^{-3} - 1.6 \times 10^{-2}$
			^{238}Pu	1.9×10^{-5}	N/A	1.9×10^{-5}
			$^{239/240}\text{Pu}$	1.5×10^{-4}	N/A	1.6×10^{-4}
			^{241}Am	1.6×10^{-4}	N/A	1.7×10^{-4}
		Duck	^{137}Cs	$2.6 \times 10^{-3} - 2.2 \times 10^{-1}$	N/A	$2.5 \times 10^{-3} - 2.1 \times 10^{-1}$
		Coot	^{137}Cs	$6.7 \times 10^{-2} - 2.2$	N/A	$6.5 \times 10^{-2} - 2.1$
External Sediment		Small insects	^{137}Cs		N/A	$2.3 \times 10^{-1} - 2.1 \times 10^2$
		Large insects	^{137}Cs		N/A	$2.1 \times 10^{-1} - 1.9 \times 10^2$
		Molluscs	^{137}Cs	$1.3 \times 10^{-1} - 1.2 \times 10^2$	N/A	$2.1 \times 10^{-1} - 1.9 \times 10^2$
		Small fish	^{137}Cs		N/A	$2.1 \times 10^{-1} - 1.9 \times 10^2$

*no water data available

TABLE A.7—Estimate of dose rates at White Oak Lake, Oak Ridge tes at Gable Mountain Pond, Hanford*

Method					
Source	Organism	Nuclide	CRITR $\mu\text{Gy h}^{-1}$	EXREM III & BIORAD $\mu\text{Gy h}^{-1}$	Point Source Dose Functions $\mu\text{Gy h}^{-1}$
Internal	Invertebrates	^3H	N/A	7.6×10^{-2}	N/A
		^{60}Co	N/A	4.0×10^{-1}	N/A
		^{137}Cs	N/A	5.3×10^{-2}	N/A
		^{90}Sr	N/A	7.2×10^{-1}	N/A
	Fish	^3H	4.3×10^{-2}	7.6×10^{-2}	4.3×10^{-2}
		^{60}Co	7.8×10^{-3}	4.2×10^{-2}	8.9×10^{-3}
		^{137}Cs	1.0×10^{-2}	2.1×10^{-1}	$1.3 \times 10^{-2} - 2.9 \times 10^{-1}$
			2.3×10^{-1}		
		^{90}Sr	$1.0 \times 10^{-3} -$ 7.2×10^{-1}	3.4×10^{-2}	$1.0 \times 10^{-3} - 7.2 \times 10^{-1}$
	Muskrat	^3H	N/A	7.6×10^{-2}	N/A
		^{60}Co	N/A	7.9×10^{-2}	N/A
		^{137}Cs	N/A	3.0×10^{-1}	N/A
		^{90}Sr	N/A	1.5×10^{-1}	N/A
External Water	Invertebrates	^3H	N/A	2.3×10^{-2}	N/A
		^{60}Co	N/A	3.4×10^{-3}	N/A
		^{137}Cs	N/A	5.7×10^{-4}	N/A
		^{90}Sr	N/A	3.4×10^{-3}	N/A
	Fish	^{60}Co	2.9×10^{-3}	3.4×10^{-3}	3.4×10^{-3}
		^{137}Cs	4.2×10^{-4}	4.7×10^{-4}	5.3×10^{-4}
	Muskrat	^{60}Co	N/A	4.2×10^{-4}	N/A
		^{137}Cs	N/A	5.9×10^{-5}	N/A
External Sediment	Fish	^{60}Co	1.2	N/A	2.0
		^{137}Cs	3.0	N/A	4.9

TABLE A.8—Estimates of the dose rates at B Pond and Seepage Basins, Savannah River

Method					
Source	Organism	Nuclide	CRITR $\mu\text{Gy h}^{-1}$	EXREM III & BIORAD $\mu\text{Gy h}^{-1}$	Point Source Dose Functions $\mu\text{Gy h}^{-1}$
Internal	Invertebrates	^{137}Cs	N/A	2.6×10^{-2}	2.4×10^{-1}
		^{90}Sr	N/A	9.1×10^{-3}	N/A
		^{238}Pu	N/A	7.5×10^{-5}	N/A
		$^{239/240}\text{Pu}$	N/A	2.7×10^{-5}	N/A
		^{241}Pu	N/A	7.8×10^{-4}	N/A
		^{244}Cm	N/A	2.2×10^{-2}	N/A
	Molluscs	^{137}Cs	2.7×10^{-2}	N/A	2.7×10^{-2}
	Fish	^{137}Cs	$3.0 \times 10^{-1} - 1.2$	1.0×10^{-3}	$3.8 \times 10^{-1} - 1.5$
		^{90}Sr	N/A	4.3×10^{-4}	N/A
		^{238}Pu	N/A	2.5×10^{-4}	N/A
		$^{239/240}\text{Pu}$	N/A	9.5×10^{-5}	N/A
		^{241}Am	N/A	1.9×10^{-5}	N/A
		^{244}Cm	N/A	5.5×10^{-4}	N/A
	Ducks	^{137}Cs	2.7×10^{-1}	N/A	2.6×10^{-1}
	Muskrat	^{137}Cs	N/A	1.5×10^{-1}	N/A
		^{90}Sr	N/A	1.9×10^{-1}	N/A
		^{238}Pu	N/A	1.1×10^{-7}	N/A
		$^{239/240}\text{Pu}$	N/A	4.1×10^{-8}	N/A
		^{241}Am	N/A	5.5×10^{-6}	N/A
		^{244}Cm	N/A	1.6×10^{-4}	N/A
	Turtles				
	Pond B	^{137}Cs	$6.7 \times 10^{-1} - 3.0 \times 10^{-1}$	N/A	$7.2 \times 10^{-2} \times 3.2 \times 10^{-1}$
		^{90}Sr	$3.6 \times 10^{-1} - 1.2$	N/A	$3.6 \times 10^{-1} - 1.2$
	Turtles				
	Seepage Basins	^{137}Cs	$4.4 \times 10^{-2} - 1.8$	N/A	$4.7 \times 10^{-2} - 1.9$
		^{90}Sr	$5.3 - 2.4 \times 10^2$	N/A	$5.3 - 2.4 \times 10^2$

TABLE A.8—Continued

Source	Method	Organism	Nuclide	CRITR $\mu\text{Gy h}^{-1}$	EXREM III & BIORAD $\mu\text{Gy h}^{-1}$	Point Source Dose Functions $\mu\text{Gy h}^{-1}$
External Water		Alligators	^{137}Cs	1.1×10^{-1}	N/A	9.4×10^{-2}
		Invertebrates	^{137}Cs	N/A	2.8×10^{-4}	1.3×10^{-4}
			^{90}Sr	N/A	4.3×10^{-5}	2.9×10^{-5}
		Molluscs	^{137}Cs	1.4×10^{-4}	N/A	1.3×10^{-4}
			^{90}Sr	2.8×10^{-6}	N/A	1.0×10^{-5}
		Fish	^{137}Cs	2.1×10^{-4}	2.3×10^{-4}	2.6×10^{-4}
		Ducks	^{137}Cs	1.0×10^{-4}	N/A	1.3×10^{-4}
		Muskrat	^{137}Cs	N/A	2.3×10^{-4}	N/A
		Turtles				
		Pond B	^{137}Cs	2.1×10^{-4}	2.3×10^{-4}	2.6×10^{-4}
External Sediment		Alligators	^{137}Cs	2.1×10^{-4}	2.3×10^{-4}	2.6×10^{-4}
		Invertebrates	^{137}Cs	N/A	N/A	2.0
			^{90}Sr	N/A	N/A	3.9×10^{-2}
		Molluscs	^{137}Cs	1.4	N/A	1.9
			^{90}Sr	6.0×10^{-4}	N/A	1.3×10^{-2}
		Fish	^{137}Cs	1.2	N/A	1.9
		Ducks	^{137}Cs	1.2	N/A	1.9
		Turtles	^{137}Cs	1.2	N/A	1.9
		Alligators	^{137}Cs	1.2	N/A	1.9

TABLE A.9—Estimates of the dose rates at Beaverlodge Lake and Tailing Creek, Saskatchewan

Source	Organisms	Nuclide	Point Source Dose Functions	
			Beaverlodge Lake $\mu\text{Gy h}^{-1}$	Tailings Creek $\mu\text{Gy h}^{-1}$
Internal	Insects	^{210}Pb & daughters	—	$1.3 \times 10^{-1} - 2.4$
		^{226}Ra & daughters	—	$2.0 \times 10^{-2} - 2.4$
		^{238}U	—	$8.0 \times 10^{-2} - 7.8 \times 10^{-1}$
		^{234}U	—	$9.1 \times 10^{-2} - 8.8 \times 10^{-1}$
		^{235}U	—	$4.0 \times 10^{-3} - 3.9 \times 10^{-2}$
	Small fish	^{210}Pb & daughters	3.4×10^{-3}	6.7×10^{-2}
		^{226}Ra & daughters	3.5×10^{-2}	$1.7 \times 10^{-1} - 5.2 \times 10^{-1}$
		^{238}U & daughters	4.5×10^{-2}	$1.4 \times 10^{-1} - 2.0 \times 10^{-1}$
		^{235}U & daughters	9.6×10^{-4}	$3.1 \times 10^{-3} - 4.2 \times 10^{-3}$
	Large fish (gonad)	^{210}Po	$2.2 \times 10^{-2} - 6.2 \times 10^{-2}$	—
		^{210}Pb & daughters	$5.0 \times 10^{-3} - 1.5 \times 10^{-2}$	—
		^{226}Ra & daughters	$2.8 \times 10^{-1} - 8.3 \times 10^{-1}$	—
		^{238}U & daughters	$3.3 \times 10^{-2} - 4.5 \times 10^{-2}$	—
		^{235}U & daughters	$1.4 \times 10^{-2} - 2.0 \times 10^{-2}$	—
External Water	Insects	^{210}Pb & daughters	1.3×10^{-6}	1.3×10^{-4}
		^{226}Ra & daughters	1.9×10^{-6}	1.9×10^{-3}
		^{238}U & daughters	7.4×10^{-4}	9.4×10^{-3}
		^{235}U & daughters	1.0×10^{-5}	1.3×10^{-4}
	Small fish	^{210}Pb & daughters	6.6×10^{-8}	6.9×10^{-6}
		^{226}Ra & daughters	1.6×10^{-5}	1.6×10^{-3}
		^{238}U & daughters	1.5×10^{-4}	1.9×10^{-3}
		^{235}U & daughters	1.0×10^{-5}	1.3×10^{-4}
	Large fish (gonad)	^{210}Pb & daughters	5.0×10^{-8}	5.1×10^{-6}
		^{226}Ra & daughters	1.5×10^{-5}	1.5×10^{-3}

TABLE A.9—Continued

Source	Organisms	Nuclide	Point Source Dose Functions	
			Beaverlodge Lake $\mu\text{Gy h}^{-1}$	Tailings Creek $\mu\text{Gy h}^{-1}$
External Sediment		^{238}U & daughters	2.7×10^{-5}	3.4×10^{-4}
		^{235}U & daughters	1.0×10^{-5}	1.3×10^{-4}
	Insects	^{238}U series	6.5×10^{-1}	3.7×10^{-1}
		^{235}U series	1.4×10^{-2}	1.5×10^{-3}
	Small fish	^{238}U series	3.0×10^{-1}	2.3×10^{-1}
		^{235}U series	5.9×10^{-3}	6.6×10^{-4}
	Large fish (gonad)	^{238}U series	2.4×10^{-1}	2.0×10^{-1}
		^{235}U series	5.6×10^{-3}	6.3×10^{-4}

Glossary

alevin: Fish fry on which the yolk is still apparent. Particularly applied to salmonids.

anthropogenic: Caused by or resulting from the acts of man.

effective dose equivalent: The sum over specified tissues of the products of the dose equivalent in a tissue and the weighting factor for that tissue.

eutrophic: Bodies of water rich in nutrients and containing a minimal amount of dissolved oxygen, usually a shallow lake, with abundant organic matter.

euryhaline: Marine organisms capable of withstanding wide variations in osmotic pressure of salinity.

isosmotic: Equal tension or equal osmotic pressure.

karyotype: The chromosome complement of a cell with individual chromosomes arranged in pairs in order of size.

gammaridae: Family of amphipod Crustacea common in marine habitat and in all freshwaters of the world.

genome: The genetic complement of a living organism or a single cell.

nauplii: Unsegmented larvae of crustaceans having a simple median eye and three pairs of appendages.

Neonate: Newly born animal.

neoplasm: A pathological lesion characterized by the progressive or uncontrolled proliferation of cells.

osmoregulation: Regulation of the osmotic pressure of body fluids by controlling the amount of water and/or salt in the body.

poikilothermic: Cold-blooded animals or animals whose body temperature fluctuates with that of the environment.

sister-chromatid exchange: The exchange of segments between sister chromatids during mitosis.

smolt: Smolt are young silvery salmon, which are about years old, that are descending streams leading to salt water.

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The National Council on Radiation Protection and Measurements is a nonprofit corporation chartered by Congress in 1964 to:

1. Collect, analyze, develop, and disseminate in the public interest information and recommendations about (a) protection against radiation and (b) radiation measurements, quantities, and units, particularly those concerned with radiation protection;
2. Provide a means by which organizations concerned with the scientific and related aspects of radiation protection and of radiation quantities, units, and measurements may cooperate for effective utilization of their combined resources, and to stimulate the work of such organizations;
3. Develop basic concepts about radiation quantities, units, and measurements, about the application of these concepts, and about radiation protection;
4. Cooperate with the International Commission on Radiological Protection, the International Commission on Radiation Units and Measurements, and other national and international organizations, governmental and private, concerned with radiation quantities, units, and measurements and with radiation protection.

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The Council is made up of the members and the participants who serve on the over sixty scientific committees of the Council. The scientific committees, composed of experts having detailed knowledge and competence in the particular area of the committee's interest draft proposed recommendations. These are then submitted to the full membership of the Council for careful review and approval before being published.

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- SC 1 Basic Radiation Protection Criteria
 - SC 1-1 Probability of Causation for Genetic and Developmental Effects
 - SC 1-2 The Assessment of Risk for Radiation Protection Purposes
 - SC 1-3 Collective Dose

- SC 16 X-Ray Protection in Dental Offices
- SC 40 Biological Aspects of Radiation Protection Criteria
 - SC 40-1 Atomic Bomb Survivor Dosimetry
- SC 46 Operational Radiation Safety
 - SC 46-2 Uranium Mining and Milling—Radiation Safety Programs
 - SC 46-4 Calibration of Survey Instrumentation
 - SC 46-5 Maintaining Radiation Protection Records
 - SC 46-8 Radiation Protection Design Guidelines for Particle Accelerator Facilities
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 - SC 46-10 Assessment of Occupational Doses from Internal Emitters
 - SC 46-11 Radiation Protection During Special Medical Procedures
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 - SC 57-2 Respiratory Tract Model
 - SC 57-6 Bone Problems
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 - SC 63-2 Criteria for Radiation Instruments for the Public
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 - SC 64-6 Screening Models
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The NCRP has found its relationships with these organizations to be extremely valuable to continued progress in its program.

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The NCRP seeks to promulgate information and recommendations based on leading scientific judgment on matters of radiation protection and measurement and to foster cooperation among organizations concerned with these matters. These efforts are intended to serve the public interest and the Council welcomes comments and suggestions on its reports or activities from those interested in its work.

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- 10 *Nonionizing Radiation Bioeffects: Cellular Properties and Interactions* by Herman P. Schwan (1986) [Available also in *Nonionizing Electromagnetic Radiations and Ultrasound*, see above]
- 11 *How to be Quantitative about Radiation Risk Estimates* by Seymour Jablon (1987) [Available also in *New Dosimetry at Hiroshima and Nagasaki and its Implications for Risk Estimates*, see above]
- 12 *How Safe is Safe Enough?* by Bo Lindell (1988) [Available also in *Radon*, see above]
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- 14 *Radiation Protection and the Internal Emitter Saga* by J. Newell Stannard (1990)

NCRP Commentaries

- | No. | Title |
|-----|---|
| 1 | <i>Krypton-85 in the Atmosphere—With Specific Reference to the Public Health Significance of the Proposed Controlled Release at Three Mile Island</i> (1980) |
| 2 | <i>Preliminary Evaluation of Criteria for the Disposal of Transuranic Contaminated Waste</i> (1982) |
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| 4 | <i>Guidelines for the Release of Waste Water from Nuclear Facilities with Special Reference to the Public Health Significance of the Proposed Release of Treated Waste Waters at Three Mile Island</i> (1987) |

- 5 *A Review of the Publication, Living Without Landfills*
 (1989)
- 6 *Radon Exposure of the U.S. Population—Status of the*
 Problem (1991)

NCRP Reports

No.	Title
8	<i>Control and Removal of Radioactive Contamination in Laboratories</i> (1951)
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ography* (1978)
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pounds Incorporated in Genetic Material* (1979)
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Response Relationships for Low-LET Radiations*
(1980)
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Radionuclides* (1980)
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Neutron Generators* (1983)
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 (1989)
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 Radiation (1989)
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- Volume X. NCRP Reports Nos. 64, 65, 66, 67
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The following NCRP Reports are now superseded and/or out of print:

No.	Title
1	<i>X-Ray Protection</i> (1931). [Superseded by NCRP Report No. 3]
2	<i>Radium Protection</i> (1934). [Superseded by NCRP Report No. 4]
3	<i>X-Ray Protection</i> (1936). [Superseded by NCRP Report No. 6]
4	<i>Radium Protection</i> (1938). [Superseded by NCRP Report No. 13]
5	<i>Safe Handling of Radioactive Luminous Compounds</i> (1941). [Out of Print]
6	<i>Medical X-Ray Protection Up to Two Million Volts</i> (1949). [Superseded by NCRP Report No. 18]
7	<i>Safe Handling of Radioactive Isotopes</i> (1949). [Superseded by NCRP Report No. 30]
9	<i>Recommendations for Waste Disposal of Phosphorus-32 and Iodine-131 for Medical Users</i> (1951). [Out of Print]
10	<i>Radiological Monitoring Methods and Instruments</i> (1952). [Superseded by NCRP Report No. 57]
11	<i>Maximum Permissible Amounts of Radioisotopes in the Human Body and Maximum Permissible Concentrations in Air and Water</i> (1953). [Superseded by NCRP Report No. 22]
12	<i>Recommendations for the Disposal of Carbon-14 Wastes</i> (1953). [Superseded by NCRP Report No. 81]
13	<i>Protection Against Radiations from Radium, Cobalt-60 and Cesium-137</i> (1954). [Superseded by NCRP Report No. 24]
14	<i>Protection Against Betatron—Synchrotron Radiations Up to 100 Million Electron Volts</i> (1954). [Superseded by NCRP Report No. 51]
15	<i>Safe Handling of Cadavers Containing Radioactive Isotopes</i> (1953). [Superseded by NCRP Report No. 21]

- 16 *Radioactive Waste Disposal in the Ocean* (1954). [Out of Print]
- 17 *Permissible Dose from External Sources of Ionizing Radiation* (1954) including *Maximum Permissible Exposure to Man, Addendum to National Bureau of Standards Handbook 59* (1958). [Superseded by NCRP Report No. 39]
- 18 *X-Ray Protection* (1955). [Superseded by NCRP Report No. 26]
- 19 *Regulation of Radiation Exposure by Legislative Means* (1955). [Out of Print]
- 20 *Protection Against Neutron Radiation Up to 30 Million Electron Volts* (1957). [Superseded by NCRP Report No. 38]
- 21 *Safe Handling of Bodies Containing Radioactive Isotopes* (1958). [Superseded by NCRP Report No. 37]
- 24 *Protection Against Radiations from Sealed Gamma Sources* (1960). [Superseded by NCRP Report Nos. 33, 34, and 40]
- 26 *Medical X-Ray Protection Up to Three Million Volts* (1961). [Superseded by NCRP Report Nos. 33, 34, 35, and 36]
- 28 *A Manual of Radioactivity Procedures* (1961). [Superseded by NCRP Report No. 58]
- 29 *Exposure to Radiation in an Emergency* (1962). [Superseded by NCRP Report No. 42]
- 31 *Shielding for High Energy Electron Accelerator Installations* (1964). [Superseded by NCRP Report No. 51]
- 33 *Medical X-Ray and Gamma-Ray Protection for Energies up to 10 MeV—Equipment Design and Use* (1968). [Superseded by NCRP Report No. 102]
- 34 *Medical X-Ray and Gamma-Ray Protection for Energies Up to 10 MeV—Structural Shielding Design and Evaluation* (1970). [Superseded by NCRP Report No. 49]
- 39 *Basic Radiation Protection Criteria* (1971). [Superseded by NCRP Report No. 91]
- 43 *Review of the Current State of Radiation Protection Philosophy* (1975). [Superseded by NCRP Report No. 91]
- 45 *Natural Background Radiation in the United States* (1975). [Superseded by NCRP Report No. 94]
- 48 *Radiation Protection for Medical and Allied Health Personnel*. [Superseded by NCRP Report No. 105]
- 56 *Radiation Exposure from Consumer Products and Miscellaneous Sources* (1977). [Superseded by NCRP Report No. 95]

- 58 *A Handbook on Radioactivity Measurement Procedures.*
 [Superseded by NCRP Report No. 58, 2nd ed.]

Other Documents

The following documents of the NCRP were published outside of the NCRP Reports and Commentaries series:

"Blood Counts, Statement of the National Committee on Radiation Protection," *Radiology* 63, 428 (1954)

"Statements on Maximum Permissible Dose from Television Receivers and Maximum Permissible Dose to the Skin of the Whole Body," *Am. J. Roentgenol., Radium Ther. and Nucl. Med.* 84, 152 (1960) and *Radiology* 75, 122 (1960)

Dose Effect Modifying Factors In Radiation Protection, Report of Subcommittee M-4 (Relative Biological Effectiveness) of the National Council on Radiation Protection and Measurements, Report BNL 50073 (T-471) (1967) Brookhaven National Laboratory (National Technical Information Service, Springfield, Virginia).

X-Ray Protection Standards for Home Television Receivers, Interim Statement of the National Council on Radiation Protection and Measurements (National Council on Radiation Protection and Measurements, Washington, 1968)

Specification of Units of Natural Uranium and Natural Thorium (National Council on Radiation Protection and Measurements, Washington, 1973)

NCRP Statement on Dose Limit for Neutrons (National Council on Radiation Protection and Measurements, Washington, 1980)

Control of Air Emissions of Radionuclides (National Council on Radiation Protection and Measurements, Bethesda, Maryland, 1984)

Copies of the statements published in journals may be consulted in libraries. A limited number of copies of the remaining documents listed above are available for distribution by NCRP Publications.

Index

- Beaverlodge Lake 63, 64, 75
- Chronic irradiation 24, 25, 26, 27, 28
 - Abnormalities in embryos 24
 - Adult pike 24
 - Community diversity 26
 - Ecological factors 26
 - Fecundity 27
 - Hormetic effect 36
 - Juvenile roach 24
 - Mosquito fish 24
 - Plaice fishery 28
 - Roach population 28
 - Recruitment in fish populations 27
 - Stochastic effects 27
- Criteria for protecting populations 36, 37, 38
 - Dose rates 37, 38
 - Effects on fecundity 36
 - Endemic populations 38
 - Histological effects 37
 - Mortality 37
 - Over exploitation 38
 - Threatened species 37
- Cytogenetic effects 29, 30, 31, 32, 33, 35
 - Ameca splendens* 30
 - β -radiation 32
 - Carp 32
 - Central mudminnow 30
 - Chironomus tentans* 32
 - Chromosome bridges and fragments 29
 - Erythrocytes 31
 - From low levels of radionuclides 30
 - High mitotic activity 29
 - Invertebrates 32
 - Karyotype 29
 - Low-level radiation 35
 - Low mitotic activity 29
 - Lymphocytes from mudminnow 31
 - Marine worms 32
 - Newt 31
 - Tritium 30
 - Micronuclei 31, 32
 - Sister chromatid exchanges 33
 - Vertebrates 30
 - X-rays 30
- Developmental effects 5, 6
 - Acute irradiation 5, 6
 - Brine shrimp 6
 - Carp eggs 6
 - Fish 5
 - Invertebrate eggs 5
 - Salmon embryos 6
- Dose to organisms and man 50
- Dose to humans 52, 53, 54
 - Consuming fish 52, 53
 - Drinking water 52, 53
 - Exposure to sediments 52, 54
- Dose rate estimates to aquatic biota 63, 64
 - Beaverlodge Lake 63, 64
 - Gable Mountain Pond, Hanford 63, 64
 - Pond B, Savannah River Plant 63
 - White Oak Lake 63, 64
 - Seepage basin, Savannah River Plant 64
- Effects of chronic irradiation 9, 19
 - Natural populations 9
- Effects on growth and development 23
- Environmental dosimetry 39, 40, 42, 43, 44, 55, 56, 58, 64, 70
 - α -emitting radiations 42, 45, 46, 58
 - β -radiation 42, 45, 46
 - β dose rate from sediment 56
 - Contaminated water 44
 - Contaminated sediment 43
 - External dose to fish 55
 - Exposure to gonads 40
 - γ -rays 45
 - γ dose rate from sediments 56
 - γ radiation for large organisms 47
 - γ radiation for small organisms 47
 - Indicator species 40
 - Internal and external sources 39
 - Internal dose to fish 55
 - in situ* measurements 40
 - Natural populations 40
 - Sediment-water interface 70
- Environmental factors 7, 8
 - Salinity 7
 - Stable element 8
 - Temperature 7

- Water chemistry 7
- External radiation 3, 20, 23
 - Chinook salmon 23
 - Coho salmon 23
 - Daphnia pulex* 23
 - Effects on growth and development 23
 - Fish 20
 - Intraspecific competition 23
 - Invertebrates 20
 - Rainbow trout 23
 - Snail eggs 23
 - Stickleback 23
- Gable Mountain Pond, Hanford 62, 63, 64, 72
- Genetic effects 29, 33, 34
 - Chronic irradiation 34
 - Deleterious genes 34
 - Dominant lethal mutations 34
 - Doubling dose 34
 - Mutation rate 33
- Laboratory studies 19
 - Fish embryos 19
- LD₅₀ 3
- Methods for environmental dosimetry 41, 44, 45, 46, 48, 51, 55, 61, 73, 74, 75
 - BIORAD 44, 48, 51, 55, 58, 61, 74, 75
 - CRITR 41, 48, 61, 73, 74, 75
 - EXREM III 44, 74, 75
 - Point Source Dose Distribution 46, 48, 61
 - Geometrical models 64
- Mortality 3, 10, 11, 12
 - Alevins 11
 - Blue crab 12
 - Clams 12
 - Embryos 11
 - Fish 10
 - Invertebrates 10, 11
 - LD₅₀ 3
 - Scallops 12
 - snails 12
- Natural populations 14, 17, 18, 24
 - Fish 14
 - Gambusia 17
 - Siberian roach 17
- Snail 18
- Pathological effects 4
 - Cellular 4
- Physiological effects 4, 5, 12, 13
 - Acute irradiation 5
 - Biochemical 4
 - Cellular 4
 - Fecundity 12
 - Chronic irradiation 12
 - Fish 5
 - Hemopoietic activity 12
 - Immune system 12
 - Invertebrate 5
 - Male germ cells 12
 - Osmoregulatory 5
 - Primary spermatogonia 12
 - Snails 13
 - Spermatogenesis 12
- Proposed standards 61
- Protection of populations 36
 - Dose rate 36
 - Effects on individuals 36
 - Human standards 36
- Radiation sensitivity 8
 - Early developmental stages 8
 - Environmental modification 8
 - Invertebrates 8
- Reproductive effects 6, 7, 14, 17, 18, 19
 - Amphiods 7
 - Chinook salmon 7, 18
 - Chronic irradiation 18
 - Daphnia pulex* 19
 - Decrease in mating success 7
 - Decrease in number of germ cells 7
 - Fecundity 18
 - Fish 6, 14
 - Guppy 18
 - Invertebrate 14
 - Laboratory studies 18
 - Medaka 19
 - Rainbow trout 7
 - Polychaete worms 7
 - Snail 7, 19
- Resistant to radiation 4
 - Aquatic organisms 4
 - Terrestrial mammals 4
- Savannah River Plant 62, 74
- White Oak Lake 63, 64, 73