Assessment of the Risks to Fish and Wildlife from Exposure to Ionizing Radiation

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Contents

Executive Summary

The nuclear industry has operated under the premise that measures taken to protect individual humans from harmful effects of exposure to ionizing radiation will also be protective of populations of fish and wildlife. Different agencies, including the International Atomic Energy Agency (IAEA) and the United States Department of Energy (DOE) have reviewed the available literature of radiological effects and evaluated the doses of radiation that plants and wild animals would be exposed to, if the levels of protection for humans in the same general area were not exceeded. From these evaluations, technical dose limits for protection of ecological resources have been proposed:

- 400 µGy/hr (1 Rad/d or 10 mGy/d) for the protection of aquatic animals;
- 400 µGy/hr (1 Rad/d or 10 mGy/d) for the protection of terrestrial plants;
- 40 µGy/hr (0.1 Rad/d or 1 mGy/d) for the protection of terrestrial animals.

This document reviews the literature of radiological effects to biota and assesses whether or not the technical dose limits proposed by DOE are adequately protective or not. In order to understand and interpret that literature, general information about ionizing radiation, its effects on living organisms, and how radiation effects are measured and evaluated is also presented. Different regulatory approaches proposed by a number of different jurisdictions are also reviewed. Particular attention is given to the "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" developed by DOE, and is already in use to assess radiological risks to biota at DOE sites and facilities.

This document does not recommend any regulatory proposals, because regulatory responsibility for managing releases of radioactive materials to the environment has been assigned to the Division of Environmental Remediation. The literature review documented here supports the contention that levels of protection specified in 6NYCRR Part 380 for the protection of individual members of public will also protect populations of fish and wildlife.

One of the primary purposes of this document is to provide a tool for estimating ecological risks from a nuclear accident or inadvertent, unregulated release of radioactive material. It recommends the integration of DOE technical dose limits with European Union predicted-no-effect-dose/dose rates to suggest different zones of radiological risk around a site of accidental or inadvertent radiological contamination, where different levels of risk could be predicted to sensitive organisms. This approach does not address the issue of potential impacts to human health from the consumption of aquatic organisms exposed to either doses of ionizing radiation or radioactive material following such an accident or significant inadvertent release (i.e., ingestion pathway analysis).

1. Statement of Purpose

The purpose of this document is to make available to DEC staff and the public, information about the risks to fish and wildlife and their habitat from ionizing radiation. This document has no regulatory standing; that is, it does not recommend or propose independently-derived standards or criteria for the protection of fish and wildlife from the effects of ionizing radiation in different media (i.e., soil, air, or water). Regulation of most[1] discharges of radioactive material into the Waters of the State or disposal of radioactive wastes or other materials in landfills or other sites is the responsibility of the Division of Environmental Remediation (DER), and is documented in 6NYCRR Part 380. The primary objective of those regulations is to protect members of the public from exposure to ionizing radiation (from sources of other than background) that exceed the standards of protection described in 6NYCRR Subpart 380.5(1).

The information in this document will be useful for Division of Fish, Wildlife, and Marine Resources (DFWMR) staff and others in assessing the ecological risks from unregulated or inadvertent discharges or releases of radioactive material, such as from accidents. This document will also identify levels of concern that could be used to identify potential radiological hazards to fish and wildlife, or to trigger non-regulatory actions such as heightened monitoring or research studies.

The nuclear industry has operated under the premise that levels of protection for individual human beings are also protective of populations of aquatic organisms as well as terrestrial plants and animals. This document will explore the validity of that principle, and will review and assess the levels of protection provided by Department of Energy (DOE) Standard (STD) 1153-2002, "A Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" (DoE 2002) as well as protective values proposed by other countries/jurisdictions.

This document will not provide a comprehensive review of the effects of radiation to aquatic and terrestrial biota. For that information, the numerous literature summaries that were reviewed in the preparation of this document should be consulted (IAEA 1992; Rose 1992; Eisler 1994; UNSCEAR 1996; and the FREDERICA Database). Appendix A does include a summary of data reviewed that is a representative sampling of the data available regarding the effects of ionizing radiation to fish, wildlife, and habitat.

2. Types of incidents/exposures

There are numerous ways in which fish and wildlife could be exposed to ionizing radiation. Solid wastes with low levels of radioactivity that meet the standards for human protection can be disposed of in landfills, and similar liquid or soluble wastes can be discharged into lakes, rivers and streams. Those types of releases are carefully and stringently regulated so as not to be harmful to individual humans, but their potential for ecological impact has not been evaluated.

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[1] The Federal government retains regulatory responsibility for uranium utilization facilities, including nuclear power plants and spent fuel recycling facilities (West Valley).

New York State has promulgated standards for the protection of individual members of the public not to exceed a total effective dose equivalent of 1 mSv (0.1 rem) in a year (DOH Regulation 10 NYCRR Part 16.7(a)(1)(i) and (ii), also 6NYCRR Part 380.5.(1)(a)(1)), or that the dose in any unrestricted area in the environment does not exceed 0.02 mSv (0.002 rem) in any one hour (6NYCRR Part 380.5 (1)(a)(2)).

Regulated discharges, releases, and emissions are not the sources of ionizing radiation that are of the greatest concern for the protection of fish and wildlife. Rather, it is unregulated, inadvertent releases that are of the greatest concern. Radioactive materials licensees, including nuclear power plants or storage facilities for radioactive material could have unknown leaks that allow the unregulated release of radioactive material into the air, water, or groundwater. Another possible source of concern would be a leak or accident during the transport of spent nuclear fuel, or large quantities of radioactive material, or radioactive waste. Finally, nuclear accidents of catastrophic (Windscale (Sellafield) England, 1957; Kyshtym, southeastern Urals, 1957; Chernobyl, Ukraine 1986; and most recently, Fukushima-Daiichi, 2011) or near-catastrophic (Three Mile Island, Harrisburg, Pennsylvania, 1979) proportions have occurred throughout the world. A large scale accident can produce quantities of radioactive fallout that could be dispersed over hundreds or even thousands of square miles.

3. Radiological Units

This document uses the standard international (SI) units adopted by the International Commission on Radiological Protection (ICRP) in 1977 (Eisler 1994). However, because historical units may be more familiar to many readers, whenever radiological values are expressed in SI units, the value in historical units will follow in parentheses. A comparison of the current SI (standard international) units with more historical units (curies, roentgens, rads, rems, etc.) with conversion factors is included in Appendix B. A description of the SI units follows:

- Becquerel (Bq) is the unit of radiological activity. It is equivalent to 1 disintegration per second. Concentrations of radioactive material in water or other media is usually stated in terms of Bq/L (pCi/L), Bq/kg (pCi/kg) or even Bq/m^2 (pCi/m^2) rather than units of mass; that is, the concentration of ^{90}Strontium in water would be described as Bq/L (pCi/L) as opposed to a number of µg/L. The specific activity of a radionuclide is the number of becquerels, or disintegrations per second, per unit of mass. Using the specific activity, levels of radioactivity can be converted to mass units.

- Coulomb/kilogram (C/kg) is the unit of radiological intensity. This value, which was seldom used in the literature reviewed for the preparation of this document, is a measure of the energy emitted by a radiological source as measured in free air, but it does not provide any information about the dose of ionizing radiation received by an exposed organism.

- Gray (Gy) is a measure of absorbed dose. It is equivalent to one joule/kilogram.

- Sievert (Sv) is a measure of total effective dose equivalent, and like the Gy, is equivalent to one joule/kilogram. A Sievert is the sum of each type of absorbed dose resulting from exposure to different types of radiation times the RBE (relative biological effectiveness) weighting factor for each radiation. The RBE factor is discussed in section 6, below. For example, an animal might have received an external dose of 0.2 mGy (0.02 rad) of gamma (γ) radiation and an internal dose of 0.01 mGy (0.001 rad) of alpha (α) radiation. The RBEs for γ and α radiation are 1 and 20, respectively, so the total dose equivalent for the animal would be:

$$(0.2 \text{ mGy} * 1) + (0.01 \text{ mGy} * 20) = (0.2 \text{ mGy}) + (0.2 \text{ mGy}) = 0.4 \text{ mSv}$$
$$(0.02 \text{ rad} * 1) + (0.001 \text{ rad} * 20) = (0.02 \text{ rad}) + (0.02 \text{ rad}) = (0.04 \text{ rem})$$

4. Radiation Primer

A nuclide is an atom with a nucleus characterized by its number of protons and neutrons. The atomic number is the number of positive charges in the nucleus. A proton has a positive charge of 1, so the atomic number is the same as the number of protons. The mass number is the total number of protons and neutrons in a nucleus. The mass number is written either after the element name or as a superscript to the left of an element's symbol. For example, a carbon nuclide with six protons and six neutrons would be designated as Carbon-12 or ^{12}C. Isotope is an older term used to identify different nuclides that have the same atomic number (protons or positive charges) but different mass numbers; that is, they differ in the number of neutrons. A radionuclide is a radioactive nuclide that disintegrates spontaneously to a daughter nuclide with the emission of characteristic subatomic particles and gamma or X-ray photons (Eisler 1994).

In alpha (α) decay, the disintegrating nucleus emits a particle consisting of 2 protons and 2 neutrons known as an α particle. The mass number (of the disintegrating nuclide) is reduced by 4 and the atomic number is reduced by 2. The range of an α particle in tissue is short, ≤ 100 µm (Pentreath and Woodhead 2001), although they have very high linear energy transfer (LET) values, thus imparting a comparatively large amount of energy over a very short distance (see section 6, below on Relative Biological Effectiveness). External doses of α particle radiation rarely present much risk to fish or wildlife, because α particles generally do not penetrate skin, scales, hair, or fur. Internal doses of this type of radiation can be very dangerous, however, if the α emitting radionuclide is inhaled or ingested, or absorbed (such as being taken in through gills).

Beta (β) particles are electrons produced by the disintegration of a neutron into an electron, a proton, and an antineutrino (Eisler 1994). The disintegration increases the atomic number (of the disintegrating nuclide) by 1 without changing the mass number. The range of a β particle in tissue can be up to 2 cm (Pentreath and Woodhead 2001), so both internal and external doses of most β particles can be harmful.

The decay and emission of α or β particles leaves the resultant daughter nuclei in an excited state that is deactivated by the emission of photons, as gamma (γ) rays or X-rays. These rays are generally very penetrating, but have low LET values (that is, they impart very small amounts of energy at any given point), particularly compared to α particles. Some rays, particularly some X-rays, have such low energy that they are not particularly penetrating.

The half-life is the time it takes for half of the nuclei in a given mass of a radionuclide to spontaneously decay into a daughter nuclide. Some radionuclides persist for millions of years, while some only last for a few seconds. Rarely does a radionuclide decay directly into a stable end product. Instead, radioactive decay proceeds through a "decay series" of daughter radionuclides. Decay series can also vary from millions of years to seconds or minutes. For example Strontium-90 decays with a half-life of 28.8 years into Yttrium-90 which in turn decays with a half-life of 64 hours into the stable end product Zircon-90 (Eisler 1994).

The important characteristics of 23 radionuclides are described in Appendix C.

5. Dosimetry

Dosimetry is the estimation, by direct measurement or by calculation, of the dose rates experienced by organisms from exposure to various radionuclides in the environment (IAEA 1992). Barnthouse (1995) reports that the greatest single uncertainty affecting radiological dose calculations for biota is in dosimetry; the calculation of the total radiation dose received by an organism from both internal and external sources. Numerous techniques have been developed for measuring radiation dose rates, but these are appropriate primarily for measuring external doses of γ radiation. They cannot be used to measure internal doses of α and β radiation, particularly to small targets such as the gonads, or the apical meristems of a growing plant (IAEA 1992).

The geometry, that is, length, width, height, shape, and density of the exposed target organism can influence and alter the absorbed dose that the organism actually receives. For example, Pentreath and Woodhead (2001) propose five types of dosimetric models for describing the geometry of different organisms exposed to radiation, and six different "cases" to describe how the organisms are exposed to media contaminated with radionuclides, such as, completely surrounded by contaminated media (air, water, soil), or dwelling at the interface of two contaminated media, such as the water-sediment interface.

Most of the generalized risk assessment models that have been used to support the position that measures to protect individual humans are also protective of fish and wildlife use very conservative assumptions regarding dosimetry. For example, IAEA (1992) and DOE (2002) use the basic assumption that the geometry of all exposed organisms can be described as ellipsoid, that the whole body experiences the same dose, and there is no self shielding (i.e., one part of the body shields other parts from exposure to radiation). These assumptions may not be very conservative for evaluating the external doses of γ radiation, however, they can be very conservative for estimating the internal dose from α and β radiation, whose range of effects in tissue is only about 100 microns and 2 cm, respectively.

Other conservative assumptions related to dosimetry include the assumption that all of the decay energy is retained in the exposed tissue (i.e., 100% absorption), that radionuclides in media are evenly distributed and infinite in extent, and that organisms are continuously exposed (DoE 2002).

6. Relative Biological Effectiveness (RBE)

The damage caused by ionizing radiation to living organisms is related to its LET, or Linear Energy Transfer. LET is defined as the amount of locally-absorbed energy per unit length (Eisler 1994). A typical unit for measuring LET is, for example, thousands of electron volts per micrometer (or micron) (keV/µm). Alpha (α) particles can produce observable damage at much lower absorbed doses than beta (β) particles or gamma (γ) rays. Thus, to determine the total effective absorbed dose equivalent in Sieverts, a total absorbed dose (in Gy) for a given type of radiation must be multiplied by a modifying factor called the relative biological effectiveness (RBE) weighting factor. RBEs are determined by comparing the radiological effects caused by one type of radiation to that caused by a standard or reference source, usually γ rays from Cesium-137 or Cobalt-60, which are assigned an RBE of 1. For human health assessment, α particles are typically assigned an RBE of 20, however, that value could be considered conservative for assessment of the total effective dose equivalent absorbed by a plant or animal. The use of an RBE weighting factor of five has been suggested as an alternative (UNSCEAR 1996). Chambers, et al. (2006) studied α particle RBE weighting factors and also recommended the use of a factor of five for environmental risk assessments. They further indicated that there was a range of uncertainty of 1-10 for population-relevant deterministic endpoints and 1-20 for stochastic endpoints. In contrast, Pentreath and Woodhead (2001) recommended the use of a weighting factor of 40 for α particle radiation when no experimental data were available.

Most of the literature reviewed recommended the use of an RBE of one for β particles. However, Copplestone, et al. (2001) proposed an RBE of three for mono-energetic electrons, or β particles of average energy (i.e., less than 10 keV). This is an important recommendation because tritium (^3H) or tritiated water (^3H_2O) is a commonly observed β emitter that would fall into this category. Tritium occurs both in nature and as one of the most abundant radionuclides in effluents from current light-water cooled reactors under normal conditions (Copplestone, et al. 2001). The use of an RBE of one for β particles from tritium could result in an underestimate of the absorbed dose equivalent received by a plant or animal from a β emitter, although the majority of the scientific literature considers an RBE of 1 for β particles in general to be appropriate.

7. Summary of the Sensitivity of Biota to Radiation

For plants, woody species are the most sensitive (UNSCEAR 1996). Pine trees (that is, *Pinus* species) are the most sensitive terrestrial plants to radiation, and that deciduous evergreen forests, tropical rain forests, herbaceous rock-outcrop communities, and abandoned cropland species were increasingly less sensitive (Eisler 1994). The least sensitive plants are simple plants such as mosses and lichens. Among the various plant components, dry seeds are the least sensitive to ionizing radiation and apical meristems are the most sensitive (UNSCEAR 1996). Among animals, mammals are the most sensitive animals, followed by birds, fish, amphibians, reptiles, crustaceans, insects, and mollusks, although there is considerable overlap in the range of sensitivities (Rose 1992; Eisler 1994; UNSCEAR 1996). For soil-dwelling invertebrates, the earthworm was the most radiosensitive organism (IAEA 1992). Among aquatic organisms, fish are the most sensitive to the effects of radiation, and developing fish embryos are particularly so

(UNSCEAR 1996). Appendix A contains a representative sampling of radiation toxicity data from UNSCEAR (1996) and IAEA (1992).

8. Effects of Radiation on Biota

The immediate harmful effects of ionizing radiation occur at the molecular and cellular level, however, when severe enough, impacts to communities, populations, individual whole organisms, organs, or tissues can be traced back to those molecular and cellular level responses. Conversely, effects at the molecular and cellular levels do not necessarily translate into harm at higher levels of biological organization (UNSCEAR 1996). In other words, even though an individual animal or plant sustains significant damage at the cellular level, the whole organism or population might be relatively unaffected.

The harmful effects of radiation can be stochastic or non-stochastic (deterministic). Stochastic effects are probabilistic in nature, where the *likelihood*, but not necessarily the *severity* of an adverse effect occurring is a function of dose; that is, the greater the dose the more likely it is that the harmful effect will occur (Upton 1983). Examples of stochastic effects include cancer induction or the production of non-lethal, heritable mutations (Copplestone, et al. 2001). Non-stochastic, or deterministic effects reflect tissue damage resulting from the collective injury or killing of many cells in an affected organ. There is a threshold dose that must be exceeded before harmful effects to the organism are observed (Upton 1983). Copplestone, et al. (2001) comments that there is a consensus of opinion that stochastic effects, other than heritable genetic damage, are likely to be of little relevance to non-human biota.

Exposure can be acute or chronic. Acute refers to short exposure to relatively high doses of radiation usually received in minutes or hours, and is measured as total absorbed dose in Grays. As with toxic chemicals, lethal, acute radiation exposures can be described with the LD_{50}, or dose that is lethal to 50% of exposed organisms (Rose 1992). With radiation, exposure-related deaths can continue to occur for an extended period of time after the exposure has ended; as much as 30 days. The period of time during which exposed animals were observed for delayed mortality is usually indicated by a subscript, so that the LD_{50} following a 30 day post-exposure observation period would be written as the $LD_{50/30}$. The period of delayed mortality for cold-blooded animals, such as fish, can be twice as long as for mammals, or as much as 60 days ($LD_{50/60}$) (IAEA 1992).

Chronic exposure generally refers to extended, or lifetime exposure, and is usually described as dose rate, in units such as milli-Grays per day (mGy/d), Grays per year (Gy/y or Gy/a[2]), or micro-Grays per hour (µGy/hr). Extending or protracting the period of time during which an animal receives a given dose of ionizing radiation can reduce the extent of injury (UNSCEAR 1996; IAEA 1992). In other words, a 10 Gy (1,000 rad) dose of radiation received in a few hours or days does much more damage than the same 10 Gy (1,000 rad) total dose received over several weeks or months. This is because cells have an inherent ability to repair damage from

[2] "a" is an abbreviation for Julian year that appears frequently in the radiological literature. A Julian year consists of 365.25 days (Wilkins 1989), which is a more precise measurement than year as abbreviated "yr", which is generally considered to mean 365 days.

ionizing radiation, and the lower the dose rate, the greater the cell's ability to "keep up" with the damage. Also, at lower dose rates, the impacted tissue has correspondingly greater time and ability to replace lethally-damaged cells and to maintain the cell population at a new level (UNSCEAR 1996). For example, Rose (1992) reported a single dose of 57 Gy (5,700 rad) to granary weevils reduced their survival from 97 to 5 days, but survival was totally unaffected when the same cumulative dose was given in five daily fractions.

Reproduction is a more sensitive parameter than mortality, and the minimum dose rate required to depress reproduction rates may be less than 10% of the dose required to produce direct mortality (UNSCEAR 1996).

A. Molecular and Cellular Level Impacts

Ionizing radiation is named as such because radiation injury is related to the production of ions (atoms with an electrical charge because of the loss or gain of electrons) within the cell (Eisler 1994). Effects result from the passage of one or more ionizing particles through, or close to, the affected gene or chromosome. Typical genetic effects include point mutations and changes in chromosome number and structure. The changes in genes and chromosomes are associated with lesions in DNA which include single-strand and double-strand breaks, crosslinks, and various alterations in sugar and base moieties (Upton 1983).

The sensitivity of cells is directly related to their reproductive capacity and indirectly to their degree of differentiation. Cells that frequently undergo mitosis are the most radiosensitive, and cells that do not divide are the most radio-resistant. Cell death occurs when, after irradiation, a cell fails to pass through more than one or two mitoses. Cellular reproductive death is especially important in rapidly dividing tissues such as bone marrow, gut lining, skin, and germinal epithelium (i.e., tissues that produce gametes). Embryos and fetuses are particularly susceptible to ionizing radiation, and very young animals are consistently more radiosensitive than adults (Eisler 1994). Adult insects are far less sensitive to radiation than vertebrates. One reason for this difference is that there is very little cell division and differentiation occurring in adult insects (IAEA 1992).

Depending on the dose received, the damage to cells and tissues can include the production of teratogenic effects (i.e., effects on developing embryos), changes in cell population kinetics, cell death, disturbances in endocrine balance, depression of immunity, and other alterations affecting homeostasis (Upton 1983).

B. Radiation Damage to the Whole Organism

In mammals (and probably other animals that have not been studied as thoroughly as mammals) acute, non-stochastic radiological damage or lethality is due primarily to disturbances in the haematopoietic system (production of red blood cells in bone marrow) and the gastro-intestinal mucosa. These cell self-renewal systems contain stem cells, which are highly sensitive to the effects of radiation and are thus the predominant influence on the radiation response. Symptoms become apparent when the end cells (cells produced by the systems) are not replaced. Mammals die from damage to the gastro-intestinal tract within the first 10 days following exposure to relatively high doses of radiation, 10 – 50 Gy (1,000 – 5,000 rads). Mammals die of bone marrow failure within weeks following whole-body exposure to mid-level doses of radiation, ranging from 1.6 – 10 Gy (160 – 1,000 rads). For chronic exposures to dose rates less than 3,800 μGy/hr (≈ 9 rad/d), radiation-induced cancer is responsible for the increased loss of life-span relative to controls (UNSCEAR 1996).

The majority of data available examines the acute effects to different plants and animals of high doses of radiation. There are many fewer studies of the chronic effects of low doses to different organisms (IAEA 1992). Many of the chronic studies of the effects of low doses of ionizing radiation show effects such as chromosomal aberrations and a reduction of reproductive material including viable gametes and embryos. These types of impacts, even to reproductive tissues, are not always or necessarily reflected in changes to population-level parameters of exposed animals (IAEA 1992, UNSCEAR 1996). In other words, individual animals appear to be able to sustain a certain level of impact before the damage is reflected by the population as a whole.

C. Population vs. Individual Effects

Much has been written about the concept that efforts to protect individual human beings from harm will also be protective of *populations* of fish and wildlife (IAEA1992; UNSCEAR 1996; DOE 2002). More specifically, the 1977 recommendations of the International Commission on Radiological Protection (ICRP) contain the following statement:

> "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 all 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" (IAEA 1992).

In 1990, the ICRP modified that statement as follows:

> "The Commission believes that the standard of environmental control needed to protect man to the degree currently thought desirable will insure 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" (IAEA 1992).

This premise has been occasionally challenged, but primarily from the perspective that there is a notable lack of data regarding chronic effects of low levels of radiation (Thompson 1988).
It is clear that discharges of radionuclides to the environment that are protective of individual humans would only result in chronic, low dose exposures to populations of wild organisms (IAEA 1992). In order to protect populations, the focus of ecological risk assessments should be on the potential for chronic, low levels of radiation to have effects on parameters related to the maintenance of the population, such as mortality, fertility, fecundity, growth rate, vigor, and mutation rate. Of course, all these attributes can be influenced by other environmental stressors besides radiation (IAEA 1992).

One example of how a population may be maintained despite adverse effects to individuals is cancer. The fate of a significant percentage of small mammals is to be consumed by larger animals. Thus, even if exposure to a particular level of radiation has the potential to cause cancer in mice, it might be unlikely that the average mouse could avoid predation long enough for cancer to be the ultimate cause of mortality. Even if it was, it is likely that the same mouse would have bred numerous times before succumbing to chronic effects of radiation. From the perspective of population maintenance, it doesn't matter what the ultimate cause of death was, as long as the mouse had adequate opportunities to successfully breed before dying. Low levels of chronic radiation can increase egg mortality amongst exposed fish eggs. But egg mortality is high for most fish anyway. For example, mortality between hatching and maturity of arctic cod is 99.99%; and this level of mortality is not uncommon for highly fecund, pelagic species. That a given fraction of egg mortality was attributable to ionizing radiation is unlikely to alter the overall mortality rate (IAEA 1966) or population dynamics.

Several studies have detected chronic effects such as increased egg mortality, increased embryo mortality, and increased number of abnormal embryos. Despite the fact that these impacts are to reproductive parameters, populations do not always appear to be affected. Trabalka and Allen (1977) examined the reproduction of mosquitofish (*Gambusia affinis*) from White Oak Lake compared to controls. White Oak Lake served as the final settling basin for low level radioactive effluent from Oak Ridge National Laboratory since 1943, although the level of radioactivity declined over two orders of magnitude between 1960 and 1977. Cesium-137 and Cobalt-60 now (i.e., 1977) account for more than 95% of the total radioactivity in the lake, with Strontium-90 accounting for most of the remainder. The measured dose rate to resident mosquitofish was 24.6 µGy/hr (≈ 0.06 rad/d). They found that the fecundity of laboratory-reared mosquitofish collected from the irradiated environment did not differ from controls, although there were a significantly higher number of dead embryos from the irradiated population, relative to controls. Embryo deaths from the irradiated population were attributed to recessive lethal mutations. Despite the occurrence of statistically significant impacts to a reproductive parameter, i.e., number of dead embryos, Trabalka and Allen (1977) describe the wild population of mosquitofish in White Oak Lake as "thriving."

Brown and Templeton (1964) investigated the effects of ionizing radiation on the developing eggs of brown trout and the saltwater plaice. Brown trout eggs were exposed to three radiation dose rates for 58 days, 29 µGy/hr (0.07 rad/d), 280 µGy/hr (0.67 rad/d), and 1,300 Gy/hr (3.1 rad/d). The number of eggs hatching from the 29 µGy/hr (0.07 rad/d) irradiated group was significantly lower than the controls, but the number of eggs hatching from the 280 µGy/hr (0.67 rad/d) group was significantly higher than controls. The number of eggs hatching from the 1,300 µGy/hr (3.1 rad/d) group was not significantly different from controls. The percent of deformed larvae was highest in the 29 µGy/hr (0.07 rad/d) and 1,300 µGy/hr (3.1 rad/d), and lowest in the 280 µGy/hr (0.67 rad/hr) group, which was nearly half as much as the control. Finally, the fry hatching from the 280µGy/hr (0.67 rad/d) group and the 1,300 µGy/hr (3.1 rad/d) group were significantly smaller than the fry from the control and 29 µGy/hr (0.07 rad/d) groups. Similar results were observed for the plaice, although the duration of irradiation was only 18 days. Their conclusions were that at the exposures tested, statistically significant reductions in the proportions hatching could not be demonstrated to occur consistently, nor could the degree of reduction be correlated with increasing dose. They also commented that where radiation doses up to 5 Gy (500 rads) were delivered chronically at a rate of up to 10,000 µGy/hr (24 rad/d) over the development period did not cause significant differences in the proportion hatching or in the number of abnormal larvae produced.

Finally, Donaldson and Bonham (1970) conducted a series of experiments in which chinook salmon (*Oncorhynchus tschawytscha*) eggs were exposed to γ radiation from a Cobalt-60 source throughout their entire development period. When both the control and irradiated fish hatched, they were marked with distinctive fin clips and allowed to migrate to the sea. The number of adult fish recaptured upon their return to spawn was tabulated after three years at sea. This was done for at least two brood cycles by the time the report was prepared. The intensity of exposure was determined to be 0.54 Roentgen/day (0.0225 Roentgen/hr). Unfortunately, Roentgens are a measure of the intensity of a γ ray field in free air, and not a measure of the total dose or dose rate absorbed by the eggs, so this value cannot be directly compared to a absorbed dose rate in µGy/hr. Their findings were that administration of 0.5 Roentgen per day from the time of fertilization up to the feeding stage produced no detected damage to the stock sufficient to reduce the reproductive capacity over a period of slightly more than one generation. Although abnormalities were increased among the young fish, they did not impair the survival of the adults, nor did the abnormalities appear in the adults. The irradiated stock returned in greater number and produced a greater total amount of viable eggs than the controls. Increasing the dose rate two and one-half-fold did not cause a corresponding increase in mortalities or abnormalities of smolts, and the growth rate of the young fish was not retarded (Bonham and Donaldson 1966).

(IAEA 1992) and (UNSCEAR 1996) reported similar findings among mammals, that is, chronic doses of ionizing radiation increase the occurrence of chromosome aberrations, mutations, and occasionally abnormal embryos and offspring, but at low doses, these individual effects do not translate into detectable impacts to populations.

D. The effects of the 1986 Chernobyl accident on fish and wildlife

On April 26, 1986, a catastrophic explosion and fire at the Chernobyl nuclear power plant, 100 km from Kiev in the Ukraine, resulted in an unprecedented release of radioactive material. The total release of radioactive substances was about 14 EBq[3] (378 MCi) (as of 26 April 1986), which included 1.8 EBq (48.6 MCi) of Iodine-131, 0.085 EBq (2.3MCi) of Cesium-137 and other cesium radioisotopes, 0.01 EBq (0.27MCi) of Strontium-90 and 0.003 EBq (0.08 MCi) of plutonium radioisotopes. Noble gases contributed about 50% of the total release of radioactivity (IAEA 2006). In the initial phase, that is, the first 10-20 days after the accident, large acute exposures were delivered to organisms close to the power plant from large quantities of short-lived radionuclides such as Xenon-133 and Iodine-131. During the second phase, which extended through the summer and early autumn of 1986, damaging total doses were accumulated despite an overall decline of radiation levels at the soil surface to 20 – 25% of initial, peak levels. In the third, ongoing phase of chronic exposure, dose rates are less than 10% of initial values and are derived mainly from Cesium-134 and Cesium-137 contamination (UNSCEAR 1996). After the initial period, Cesium-137 became the nuclide of greatest radiological importance, with Strontium-90 being of less importance. For the first years Cesium-134 was also important[4]. Over

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[3] 1 EBq = 10^{18} Bq (Becquerels)

[4] Cesium-134 has a half-life of 2.06 years while Cesium-137 has a half-life of 30.2 years

the longer term (hundreds to thousands of years), the only radionuclides anticipated to be of interest are the plutonium isotopes and Americum-241 (IAEA 2006). The accident occurred in late April, just as wild plant and animal populations were entering the accelerated growth and reproductive phases of their life cycles; that is, when they were the most radiosensitive (UNSCEAR 1996).

In a zone of 500 – 600 ha nearest to the reactor, pine trees were estimated to have received doses in excess of 80 – 100 Gy (8,000 – 10,000 rads). Within two weeks, the trees were dying, but there was some evidence of root survival. Pine trees are among the most sensitive plants to ionizing radiation. Deciduous trees exposed to the same range of doses were only partially damaged. The next zone, of about 3,000 ha, received doses of about 8 – 10 Gy (800 – 1,000 rads). Die-back of vegetative shoot growth was evident and there was damage to needles and buds. In the third zone of about 12,000 ha, pine trees received doses of about 3.5 – 4 Gy (350 – 400 rads). The apparent effects included growth suppression, needle loss, reduced reproductive capacity and genetic damage. Even though some damage remained evident, growth of trees continued and by 1988-89 was apparent even in the second zone (UNSCEAR 1996).

A marked reduction in the number of species among forest litter arthropods was noted in 1986-1989 within the 30 km zone around the reactor site. At areas that had received a total dose of 30 Gy (3,000 rads), no changes were noted in adult animals but the numbers of juveniles were seriously depleted. In the succeeding 2 – 2.5 years, the populations largely recovered, probably owing to immigration (UNSCEAR 1996).

Fish populations seemed unaffected in July-August 1987 and no grossly deformed individuals were noted, although young fish had elevated levels of Cesium-134 and Cesium-137 (Eisler 1994). However, (UNSCEAR 1996) reports that radiation-induced damage was observed in the gonads of fish surviving the accident and in subsequent generations. Over the period 1989-1992 five of the seventy silver carp (*Hypophthalmichthys molitrix*) examined were sterile, and 35% of females and 48% of males showed gonad abnormalities. For bottom-dwelling fish of the 1985-1986 year class, the accumulated dose from internal and external sources was estimated to have reached 10 Gy (1,000 rads) by 1991 (UNSCEAR 1996).

It has been estimated that 90% of the rodents died in areas that received 60 Gy (6,000 rads) and 50% of the rodents died in areas that received 6 – 60 Gy (600 – 6,000 rads). Rodent populations seemed normal in the spring of 1987, and this was attributed to immigration (Eisler 1994).

There has been no report of a local (i.e., isolated) population of a single species having been eliminated as a consequence of the radiation exposure (UNSCEAR 1996). Eisler (1994) reports that no changes in survival or species composition of game animals and birds were recorded, and because humans had been evacuated and hunting pressure was negligible, many game species including foxes, hares, deer, moose, wolves, and waterfowl have moved into the zone around the accident site. The Ukrainian government declared the irradiated area around the accident site as a wildlife refuge in 2000 (Schlesinger 2008). Some researchers report that animal populations appear to be thriving in the area, while others report significant adverse effects including increased genetic damage, reduced reproductive rates, mutations and deformities and dramatically higher mortality rates (Birch 2007). Both within the exclusion zone and beyond, chromosomal anomalies attributable to radiation continue to be reported from experimental studies, although the biological significance of those anomalies is unknown (Schlesinger 2008).

The Chernobyl accident has been described as the foremost nuclear catastrophe in human history (Schlesinger 2008). Despite the unprecedented level of radiological impact, the fish and wildlife populations that now inhabit the vicinity of the destroyed reactor appear to have developed adaptations for surviving in an area of continuing chronic radiation exposure.

E. Population Perspective

As previously stated, the existing principle of radiological protection is that if levels of exposure that are protective of individual humans are not exceeded, then populations of exposed biota will also be protected. What constitutes protection of an individual human is easy to define and measure. Protection of a population is a more difficult parameter to quantify.

The protection of populations should be clearly defined. A population is satisfactorily protected when it is self-sustaining in the presence of low levels of radiation, even if some fraction of the individual animals or plants that make up the population experience adverse impacts. At some point, however, the extent and severity of individual impacts must be considered as harmful, even if the population appears to be self-sustaining. Also, there are occasions when protection of a population is inadequate and individual animals must be protected. For example, rare, threatened, or endangered animals must be protected at the individual level. Other unique populations may also require protection at the level of the individual animal. The goal of radiological protection is maintenance of populations; however, as IAEA (1992) acknowledges: "Although it is perhaps obvious, it needs to be stated that there can be no effects at the population level if there are no detectable effects in individuals making up that population."

A population restored by immigration is not acceptable protection. Consider, for example, a population of organisms that experienced reduced reproduction rates because of continuous exposure to chronic, low levels of ionizing radiation, but the population was maintained by ongoing immigration from populations outside the contaminated area. That population should not be considered as "protected", and would indeed be experiencing an unacceptable level of impact; that is, the population would be significantly diminished or even extirpated if immigration was eliminated.

The discussion above attempts to provide examples of individual animals within a population experiencing adverse impacts from radiation, but the population being maintained without immigration. Of all the examples, the clearest is that of irradiated salmon, where despite measurable impacts to larval fish, greater numbers of irradiated fish survived and returned to spawn than fish not irradiated. Chernobyl is a complex example. Clearly, the acute impact of high doses of radiation was eradication of some local populations, but those populations were eventually restored by immigration. The significance of Chernobyl is not that animals returned once the acutely toxic dose rates had diminished. The significance is that populations are able to maintain themselves at present, despite continuous exposure to chronic, low levels of radiation.

9. Different Regulatory Approaches

A. IAEA/UNSCEAR

As stated previously, the nuclear industry in general has adopted the position that measures implemented to protect individual humans from harmful effects of ionizing radiation are also protective of populations of fish, wildlife, and terrestrial plants. Over time, studies have been undertaken to validate this position. IAEA (1992) conducted this type of assessment by first reviewing and summarizing the available literature for radiological effects, and identifying thresholds below which harmful effects were unlikely to occur. Then, the amount of radiation that would produce the threshold dose for human impacts (1 mSv/yr) (0.1 rem/yr) was modeled using various dosimetry techniques to determine the absorbed doses that similarly exposed fish and wildlife receptors would experience. Those exposure levels were then compared to the risk thresholds, and it was generally found that the exposure that fish and wildlife were likely to experience was below the risk threshold for population effects. This approach confirmed the concept that the levels of radiation protection for individual humans would also be protective of fish and wildlife resources. UNSCEAR (1996) completed a similar review and assessment and came to the same conclusions.

B. United States Department of Energy

The U.S. Department of Energy took the risk thresholds determined by IAEA (1992) and by others and refined them into the following biota dose rate limits as appropriate values for the protection of populations from effects of ionizing radiation (DoE, 2002):

- The absorbed dose to aquatic animals should not exceed 10 mGy/d (400 µGy/hr) (1 rad/d);

- The absorbed dose to terrestrial plants should not exceed 10 mGy/d (400 µGy/hr) (1 rad/d);

- The absorbed dose to terrestrial animals should not exceed 1 mGy/d (40 µGy/hr) (0.1 rad/d).

The proposed dose limit for aquatic organisms was also documented in a literature review and study (Blaylock and Templeton 1986) conducted in response to a request to the DoE Savannah River Laboratory Facility from the South Carolina Department of Health and Environmental Control for effluent limits for discharges of radioactivity to streams to protect aquatic biota (DoE 1986). These technical dose limits are consistent with the threshold limits observed/suggested by IAEA (1992) and UNSCEAR (1996). The DoE Technical standard will be discussed in detail in Section 10 below.

One of the problems with the process for deriving the dose limits proposed in (DoE 2002) is that they appear to be based only on a qualitative assessment of radiological effects data. A quantitative process was not followed to derive a value, such as is typically done with ambient water quality standards (Stephan, et al. 1984). While a qualitative review of the available data supports the position that few adverse impacts have been documented at absorbed doses or dose rates below the proposed DOE technical dose limits, there are also very few studies of effects of low levels of radiation. For example, Eisler (1994) summarizes more than 35 studies of radiological effects to aquatic organisms, but more than 70% were acute exposures. Of the remaining studies that could be described as chronic, only one study appeared to evaluate risks from a dose lower than the proposed DoE dose limit for aquatic life of 400 µGy/hr (1 rad/d). That study examined the effects of a 70 day exposure of salmon eggs to a dose rate of 200 µGy/hr[5] (0.5 rad/d), and found no adverse effects. Of the 69 animal studies documented in Appendix A that reported a dose rate (as opposed to total dose), only 12 studies evaluated effects of dose rates below the dose limits proposed by DoE (2002). Of those 12 studies with dose rates below the proposed limits, eight indicated no adverse effects noted. The remaining four noted adverse impacts (e.g., reduced egg capsule production, impacts to fecundity, greater number of dead or deformed embryos, increased frequency of chromosome aberrations), but three of the four specifically commented that populations were not affected.

C. Canada

There have been several attempts to derive radiological limits quantitatively. Hinton, et al. (2004) described work done by the Canadian Nuclear Safety Commission in developing a method for assessing the risk from radionuclides in the environment by a method that is aligned with traditional ecological risk approaches. From the available data they derived estimated exposure values (EEVs) and estimated no-effect values (ENEVs) that are more restrictive than IAEA/DoE values for fish and terrestrial plants, but less restrictive than IAEA/DoE values for terrestrial animals (see Table 1). The Canadian method has been controversial because of their use of an RBE for α particle radiation of 40, which is twice the RBE for α particles typically used for human health protection. The use of a smaller RBE would undoubtedly result in less stringent limits.

Table 1. Comparison of radiological risk thresholds with those proposed by IAEA and DOE

	Canadian ENEV mGy/d (µGy/hr) {rad/d}	IAEA/DOE technical dose limit mGy/d (µGy/hr) {rad/d}
Aquatic organisms	0.5 (21) {0.05}	10 (400) {1}
Terrestrial plants	2.7 (113) {0.27}	10 (400) {1}
Terrestrial animals	2.7 (113) {0.27}	1 (40) {0.5}

D. European Union

The European Union (EU) has initiated several comprehensive, overlapping programs for developing limits for ecological protection from ionizing radiation. Their efforts appear to have been spearheaded by an English research and development project entitled “Impact Assessment of Ionizing Radiation to Wildlife” (Copplestone, et al. 2001). It describes an approach for

[5] Several studies reviewed in (Eisler 1994) reported on the exposure of eggs to concentrations of radionuclides in water as Bq/L. It is difficult to ascertain the dose or dose rate absorbed by the egg from such concentration data.

assessing total absorbed dose to selected reference organisms in freshwater, estuarine/marine, and terrestrial ecosystems. This effort was integrated into the FASSET (Framework for Assessment of Environmental Impact) program which ran from 2000 to 2003. FASSET was eventually superseded by the ERICA (Environmental Risk from Ionizing Contaminants: Assessment and Management) program which ran from 2004 to 2007.

An important component of the FASSET/ERICA project was the development of an online, publicly accessible database known as FRED (FASSET Radiation Effects Database). Initially, 1,033 references covering biological effects of ionizing radiation to a range of non-human species published between 1945 – 2001 were selected for review and entry into FRED, resulting in about 25,000 data records. With the inception of the ERICA project, additional data from the EU-funded EPIC (Environmental Protection from Ionizing Contaminants in the Arctic) project from 2002 – 2006 was integrated into FRED, adding about 1,400 data records from 435 references and books (Copplestone, et al. 2008). With the integration of EPIC data, the database was renamed FREDERICA. This database can be readily accessed by the public at: http://87.84.223.229/fred/mainpage.asp.

One of the deliverables of the ERICA project was the derivation of a predicted No-Effect-Dose-Rate for ecosystems. Studies from the FREDERICA database were screened and selected, and a systematic mathematical treatment was applied to reconstruct dose/dose rate-effect relationships and derive critical toxicity endpoints; the Effect Dose, or ED_{50} in Gy for acute exposure and the Effect Dose Rate or EDR_{10} in µGy/hr for chronic exposures. Once the critical toxicity endpoints were determined, they were used with standard EU risk assessment protocols such as the Safety Factor Method and the Species Sensitivity Distribution Method to derive a Predicted No-Effect-Dose (PNED) and Predicted No-Effect-Dose Rates (PNEDR). PNEDs and PNEDRs were derived for three composite ecosystems; terrestrial, freshwater, and marine. The final result was the derivation of Tier 1 acute total dose screening values for acute effects of 900 mGy (90 rads) for the protection of 95% of species in marine ecosystems and 300 mGy (30 rads) for the protection of 95% of species in terrestrial and freshwater systems, and chronic dose rate screening values of 10 µGy/hr (0.024 rad/d) for the protection of 95% of species in all three ecosystems (Garnier-Laplace and Gilbin 2006).

10. Detailed Overview of the U.S. DoE Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota

In 2002, the United States Department of Energy published the "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota," DOE Technical Standard DOE-STD-11523-2002 (DOE 2002). This technical standard integrates and utilizes the dose limits for the protection of aquatic animals, terrestrial plants, terrestrial animals listed in Section 9.B., above. It is a risk assessment procedure for evaluating the risks to aquatic animals or terrestrial animals or terrestrial plants in a given area of exposure to ionizing radiation. The Graded Approach uses a multiple-step process for evaluating the risks of ionizing radiation to fish and wildlife:

- Data Assembly
- General Screening
- Analysis
    - Site-Specific Screening
    - Site Specific Analysis
    - Site Specific Dose Assessment

To use the Graded Approach model, the Biota Dose Assessment Committee (BDAC) of DoE's Office of Health, Safety, and Security developed the RAD-BCG Calculator. This is a series of MS Excel spreadsheets that enable a user to work through both the general screening and data analysis steps of a risk assessment. To complete the General Screening step, the user only needs to enter the concentrations of various radionuclides in water and sediment or soil. The spreadsheets will estimate the internal and external dose rates for the cumulative load of radionuclides, and determine if the any of the technical dose limits are exceeded. If a dose limit is exceeded, the same series of spreadsheets allow the user to proceed to the Data Analysis step by changing default, conservative values in the spreadsheets to increasingly less conservative, site-specific, measured values. The RAD-BCG calculator can be downloaded from the BDAC website at: http://homer.ornl.gov/nuclearsafety/env/bdac/.

In the General Screening step, radionuclide concentrations in different media are compared to radionuclide-specific Biota Concentration Guides (BCGs). A BCG is the concentration of a specific radionuclide in a given media (water, sediment, soil) which would not result in an exceedance of the technical dose limits for protection of exposed plants or animals. If a site fails the general screening, the next step is the site-specific screening phase. In the site-specific screening phase, the risk assessor begins to replace conservative assumptions from the general screening step with site-specific measurements. For example, instead of using the *highest* radionuclide concentration measured anywhere in the study area, the *mean* radionuclide concentrations can be used. The size of the study area can refined, or broken down into several smaller sub-sections for individual analysis. The lumped parameter is a conservative estimate of the uptake of the different radionuclides from the surrounding media (soil, water, or sediment) derived from literature. In the site specific screening phase, the default lumped parameter uptake values can replaced with site-specific, empirically measured uptake values.

The site-specific analysis phase allows for the use of uptake information for specific representative animals within the study area. Default values for parameters such as body mass, ingestion and inhalation rates, and biological decay rates for radionuclides are replaced with specific values from representative species. Also, in the site-specific analysis phase, the use of a correction factor for the amount of time a representative animal is likely to spend within the contaminated area is considered. After making the changes allowed in both the site-specific screening phase and the site-specific analysis phase, the RAD-BCG calculator is re-run. If at the conclusion of either phase the RAD-BCG calculator shows that the standard of protection has been met, then further assessment is not needed. A risk assessor would only proceed to the site-specific analysis phase if the results of the site-specific screening phase still indicated that the technical dose limits were still exceeded.

If the dose rate determined from the site-specific screening analysis still exceeds the technical dose limits, the risk assessment proceeds to the site specific dose assessment step. In this step, a biota dose assessment team is assembled that designs appropriate studies and data collection efforts for a direct analysis of the contaminated environmental media and impacted biota.

As with any graded approach, the risk assessment is terminated at the lowest step in the process wherein acceptable risk levels are observed. For example, a study would stop at the general screening step and not proceed to the data analysis step if acceptable risk was indicated.

The same "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota" is included in DoE's RESRAD series of radiological models, and is known as RESRAD-Biota. The RESRAD-Biota model is a MS-Access database version; however, it uses the same parameters and algorithms and produces the same result as the RAD-BCG Calculator.

BCGs have been calculated for Americum-241, Cerium-144, Cesium-135, Cesium-137, Cobalt-60, Europium-154, Europium-155, Hydrogen-3, Iodine-129, Iodine-131, Plutonium-239, Radium-226, Radium-228, Antimony-125, Strontium-90, Technetium-99, Thorium-232, Uranium-233, Uranium-234, Uuranium-235, Uranium-238, Zinc-65, and Zirconium-95.

The EU ERICA project described in Section 9(D) above also incorporates a multi-tiered, graded approach similar to the "Graded Approach for Evaluating Radiation Doses to Aquatic and Terrestrial Biota." The screening values described in that section are used in the Tier 1 step analogous to the General Screening step for DoE's model. Higher tiers in ERICA similarly replace conservative default values with measured parameter values. ERICA also has downloadable, user-friendly assessment tools similar to the RAD-BCG Calculator (Beresford, et al. 2007).

11. Summary and Conclusions

Fish and wildlife can be exposed to ionizing radiation from intentional, regulated releases or accidental, unregulated releases. Regulated releases are limited by standards designed to protect individual members of the public from harm. International and Federal regulatory agencies have operated under the premise that standards for ionizing radiation that are protective of individual humans will also be protective of populations of fish and wildlife. While numerous data reviews appear to support that position, the principle has not been quantitatively tested.

The intentional release of radioactive materials into the environment is regulated by the NYSDEC Division of Environmental Remediation under the provisions of 6NYCRR Part 380. Such releases are regulated so as to meet standards of protection for individual members of the public. Assessments such as (IAEA 1992) qualitatively demonstrate that measures to protect individual humans appear to be protective of populations of fish and wildlife. Individual organisms in the direct vicinity of a regulated release of may experience adverse impacts, but populations as a whole are not likely to be harmed.

Ionizing radiation harms organisms directly at the molecular (chromosome, DNA) and cellular level. Damage to chromosomes and DNA can range from heritable mutations to outright cell death. If the damage is widespread and severe enough, tissues and organs can be damaged leading to a range of effects from impaired reproduction (probably the most sensitive impact of exposure to radiation) to the death of the organism. Tissues with rapid and frequent cell division such as bone marrow where red blood cells are produced, the lining of the gastro-intestinal system, and reproductive tissues are the most sensitive tissues to the effects of radiation. These types of effects are considered to be non-stochastic, or deterministic effects, for which there is a threshold dose for measurable harm. Stochastic effects are effects for which there is no threshold, and the likelihood of occurrence is proportional to the dose of radiation received; such as the initiation of cancer or heritable mutations. Stochastic effects are generally thought of as less important than non-stochastic effects when considering the risks to plant and animal populations. Laboratory studies and field investigations of radiation-related accidents has shown that exposure to the same levels of radiation capable of producing harmful effects to individual organisms does not necessarily impair or harm populations of organisms. Populations can show normal dynamics even though there is an increased frequency of recessive mutations, chromosomal aberrations, and non-viable or abnormal reproductive products (gametes, fetuses, embryos, larvae). Cells have the ability to repair damage from ionizing radiation, and as a result, the same dose given in lower increments over time does less damage.

From reviewing the available literature, the International Atomic Energy Agency and the United States Department of Energy have independently identified levels of absorbed dose rates of radiation that serve as technical dose limits for the protection of ecological resources. These absorbed dose rate risk thresholds are 400 µGy/hr (1 rad/d) for the protection of aquatic animals and terrestrial plants, and 40 µGy/hr (0.1 rad/d) for the protection of terrestrial animals. Below these levels, populations of fish, wildlife, and terrestrial plants are unlikely to be harmed, although individual organisms might be. These limits are derived from a qualitative assessment of available data.

The European Union constructed a comprehensive database of the effects of ionizing radiation to fish and wildlife, and used this database along with standard risk assessment protocols to derive quantitative risk thresholds for the protection of ecological resources. They determined that a dose rate of 10 µGy/hr (0.024 rad/d) to be the chronic predicted no effects dose rate for terrestrial, freshwater, and marine/estuarine ecosystems. The predicted no effect dose for acute effects found to be 300 mGy (30 rads) for species in terrestrial and freshwater ecosystems and 900 mGy (90 rads) for the protection of species in marine ecosystems.

These levels are *not* appropriate for situations when individual animals must be protected; for example, when rare, threatened, or endangered species are present. In those circumstances, site-specific values for the protection of individual organisms must be derived.

Both the U.S. Department of Energy dose rate limits and the European Union no-effects dose rate are general screening values that are applied in a graded approach. They are derived from very conservative assumptions regarding the exposure to sources of ionizing radiation. An exceedance of a screening value triggers a series of increasingly less conservative exposure modeling analyses that integrate site-specific measurements in place of default parameters for estimating the total absorbed dose.

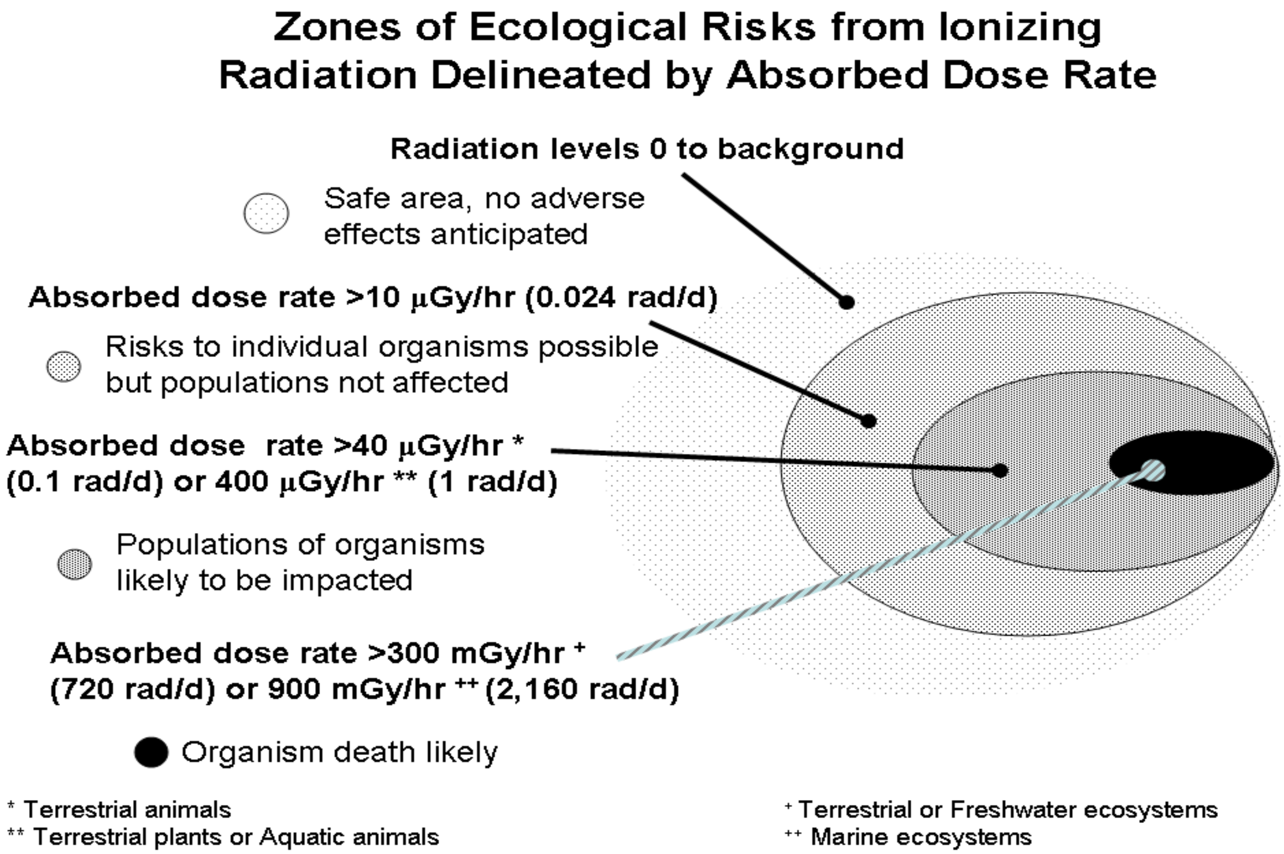
12. Recommendations

If the need arises, Division of Fish, Wildlife and Marine Resources (DFWMR) should make use of the technical dose limits proposed in DoE (2002) for the protection of aquatic animals, terrestrial plants, and terrestrial animals as screening-level threshold values for the protection of fish, wildlife, and habitat (i.e., terrestrial plant communities such as forests), when evaluating the risks from exposure to ionizing radiation. These risk thresholds should be used along with the graded approach described in DoE (2002) and the RAD-BCG calculator (or the RESRAD-Biota program) for evaluating and interpreting the risks to fish and wildlife from either accidental or heretofore unknown releases of radioactive materials into the environment. These tools/protocols are the same as those used by DoE for evaluating the ecological risks associated with the regulated releases of radioactivity from DoE-regulated sites.

DFWMR can also make use of the European Union-derived acute and chronic screening dose rates to identify the boundary of a contaminated site or exposure area wherein individual animals may be harmed but populations are not likely to be affected. The results of these two different risk assessment processes can be integrated to identify overlapping zones of risk for fish and wildlife. For example, consider an accidental spill or release of radioactive material. The EU screening levels and DOE dose limits can be integrated to identify zones of different degrees of potential risk to fish, wildlife, and terrestrial plants around an area of radiological contamination. Figure 1 is a graphical illustration of the different levels of ecological risk that can be associated with increasing levels of ionizing radiation.

A radiological accident or release of radioactive material that would produce the dose rates illustrated in Figure 1 would be an event of significant concern to human population within the area. It is unlikely that there are any actions that could be taken to protect exposed fish, wildlife, or plant communities from the effects of the exposure. This information is provided primarily to allow for the assessment of impacts that the exposed ecological communities are likely to experience, and possibly as the basis for future monitoring programs as the impacted area recovers.

Figure 1.



Zones of Ecological Risks from Ionizing Radiation Delineated by Absorbed Dose Rate
Radiation levels 0 to background
Safe area, no adverse effects anticipated
Absorbed dose rate >10 µGy/hr (0.024 rad/d)
Risks to individual organisms possible but populations not affected
Absorbed dose rate >40 µGy/hr * (0.1 rad/d) or 400 µGy/hr ** (1 rad/d)
Populations of organisms likely to be impacted
Absorbed dose rate >300 mGy/hr + (720 rad/d) or 900 mGy/hr ++ (2,160 rad/d)
Organism death likely
* Terrestrial animals
** Terrestrial plants or Aquatic animals
+ Terrestrial or Freshwater ecosystems
++ Marine ecosystems

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Appendix A. Summary of Representative Radiation Toxicity Data

For individual species, data are arranged first by total dose (acute) from lowest to highest, then by dose rate (chronic) from lowest to highest. Studies with both a total dose and dose rate are listed by total dose. For ease of comparison and ranking, dose rates described in the original literature source have all been converted to micro Grays per hour (µGy/hr) with historical units (rad/d) in parentheses. This listing is not intended to be comprehensive, but representative of the scope and range of radiological data available. The table includes data from both field and laboratory studies.

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
Mammals					
mice	0.044 (4.4)	130 (0.3)	14 days	50% reduction of immature oocytes, when exposed to tritiated water during days 19-33 after conception.	UNSCEAR 1996, para. 117
mice	0.08 (8)			50% reduction in the number of oocytes at their most sensitive stage in newborn mice.	UNSCEAR 1996, para. 116
mice	0.2 (20)			Reproduction impaired in females.	IAEA 1992, page 16
mice	1 (100)			Permanent sterility in females.	IAEA 1992, page 16
mice	3.2 (320)			Reproduction impaired in males.	IAEA 1992, page 16
3 to 5-day old mice	10 (1,000)			Can induce permanent sterility.	UNSCEAR 1996, para. 116
adult male mice	10 (1,000)			Temporarily impaired fertility.	UNSCEAR 1996, para. 116
mice	12 (1,200)			Approximate LD_{50} for gastro-intestinal syndrome.	UNSCEAR 1996, para. 106
male mice		38 (0.09)		Male mice produced 1.4 times weaned young as control; 1,000 times normal background radiation.	IAEA 1992, page 28
female mice		83 (0.2)		No. of offspring weaned was 74% of control; 1,000 times normal background radiation.	IAEA 1992, page 28
pocket mice		420 (1)		Marginally reduced survival.	UNSCEAR 1996, para. 114

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
mice		500-1,000 (1.2 – 2.4)	ten generations	Did not affect fertility as indicated by the average size of the first litter. Four different strains of mice undergoing continuous irradiation.	UNSCEAR 1996, para. 121
female mice		800 (1.9)		Became sterile at 25 weeks of age.	UNSCEAR 1996, para. 119
mice		833 – 8,333 (2 – 20)		Nearly all parameters were affected by all dose rates. In contrast to acute studies, males more adversely affected than females by chronic irradiation	IAEA 1992, page 20
mice		3,300 (7.92)	20 days	Complete sterility following irradiation during the period 20-40 days after conception.	UNSCEAR 1996, para. 117
female mice		4,000 (9.6)		Became sterile at 7 weeks of age.	UNSCEAR 1996, para. 119
rats	0.04 (4)			Significantly impaired learning ability. Four daily doses of 0.01 Gy at days 6-9 after conception.	UNSCEAR 1996, para. 127
rat	11 (1,100)			Approximate LD_{50} for gastro-intestinal syndrome.	UNSCEAR 1996, para. 106
albino rats		2,083 (5)	four generations	No effect on reproductive capacity.	IAEA 1992, page 20
desert rodents		417 (1)		Approximate chronic threshold.	IAEA 1992, page 20
redback voles		625 (1.5)		Expected effects totally masked by immigration from unirradiated areas.	IAEA 1992, page 20
rabbits		83 (0.2)		lymphocytes carried an increased number of unstable chromosome aberrations such as fragments and dicentrics	IAEA 1992, page 28
dogs	8 (800)			Approximate LD_{50} for gastro-intestinal syndrome.	UNSCEAR 1996, para. 106
beagle dog		36 (0.086)	whole life	No effect.	UNSCEAR 1996, para. 120
dogs		167 (0.4)		Measurable declines in number, motility, and viability of sperm.	IAEA 1992, page 20
sheep	0.1 (10)			Exposed every 28 days for 13 months; normal reproduction, healthy lambs.	IAEA 1992, page 16

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
Sheep	0.25 (25)			Exposed every 28 days for 13 months; normal reproduction, healthy lambs.	IAEA 1992, page 16
sheep	2.5 (250)	6,500,000 (15,600)		$LD_{50/60}$	IAEA 1992, page 16
sheep	> 10 (> 1,000)	< 10,000 (< 24)		$LD_{50/60}$	IAEA 1992, page 16
pigs and donkeys		4,167 (10)		Showed some deterioration in a few weeks and died after a few months of continuous exposure.	IAEA 1992, page 20
cattle, sheep, goats, pigs, burros, horses	1.2 - 3.9 (120 – 390)			Approximate LD_{50} for bone marrow failure (haematopoietic syndrome).	UNSCEAR 1996, para. 107
sheep, cattle, pigs, horses	4 – 7 (400 – 700)			LD50/60; acute, whole body dose γ irradiation	IAEA 1992, page 16
bull	0.5 (50)			Slight change in bull semen. Full recovery 30 weeks after irradiation.	IAEA 1992, page 16
rhesus monkey	9 (900)			Approximate LD_{50} for gastro-intestinal syndrome.	UNSCEAR 1996, para. 106
mixed animals	0.7 (70)	42 – 83 (0.1 – 0.2)		Large number of abnormalities noted, including: aberrant mitoses, decreased body fat, lower fertility, degeneration, necrotic processes, reduced fertility, and lower animal density. Area of high natural radiation, internal doses from incorporated radionuclides not considered, chemical toxicity of environment not considered.	IAEA 1992, page 28
numerous small mammals	>2 (>200)			Direct mortality from acute, whole body dose.	IAEA 1992, page 16
larger mammals	1.6 - 2.5 (160 – 250)			Approximate LD_{50} for bone marrow failure (haematopoietic syndrome).	UNSCEAR 1996, para. 107
small mammals	6 - 10 (600 – 1,000)			Approximate LD_{50} for bone marrow failure (haematopoietic syndrome).	UNSCEAR 1996, para. 107

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
numerous small mammals	5 – 11 (500 – 1,100)			$LD_{50/30}$	IAEA 1992, page 16
mammals in a hardwood forest		833 (2)		No effects on survival.	IAEA 1992, page 20
BIRDS					
white leghorn chickens	4 (400)			Egg production reduced for 10 days after exposure. At higher doses, effects more severe and longer lasting.	IAEA 1992, page 17
domestic fowl	6.7 (670)			Fowl irradiated at 2 days of age, growth rate over the next 30 days was reduced but only significantly at dose rates higher than 6.7 Gy	UNSCEAR 1996, para. 147
2 day-old broiler chicks	7 (700)	4,800,000 (11,520)		Growth rate decreased over 30 days at all exposures, but only significantly so at total dose of about 7 Gy; radiation from ^{60}Co γ source.	IAEA 1992, page 16
white leghorn chickens	8 (800)	14,000,000 (33,600)		Effects on egg hatchability, progeny unaffected.	IAEA 1992, page 17
white leghorn chickens	8 (800)	600,000 (1,440)		At this reduced dose rate (from above), effects on egg production from total does of 8 Gy were eliminated.	IAEA 1992, page 17
chickens	9 (900)			LD_{50} at hatching, irradiated on day 10 of development.	UNSCEAR 1996, para. 149
domestic poultry	9 (900)			$LD_{50/30}$	UNSCEAR 1996, para. 146
chickens	7 – 11 (700 – 1,100)		< 1 hour	$LD_{50/30}$	UNSCEAR 1996, para. 148
artificially incubated chicken embryo	12 – 13 (1,200 – 1,300)			LD_{50} irradiated at Day 10 of development.	IAEA 1992, page 17
chickens	12 – 20 (1,200 – 2,000)		24 hours	$LD_{50/30}$	UNSCEAR 1996, para. 148

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
chickens		8,000 - 10,000 (19.2 – 24)		Complete sterility resulted.	UNSCEAR 1996, para. 130
tree swallow eggs	1.6 (160)			Growth depressed when irradiated on days 7-8 of development.	UNSCEAR 1996, para. 149
tree swallow eggs	3.4 (340)			Total dose up to 3.4 Gy did not affect hatching or fledging success, but the time to hatching increased. Irradiated on days 7-8 of development.	UNSCEAR 1996, para. 149
swallow embryos	>1.6 (>160)			Incubation time slightly increased, growth slightly depressed irradiated on days 7-8 of development.	IAEA 1992, page 17
swallow embryos	3.2 (320)			No effect on hatching or fledgling success, irradiated on days 7-8 of development.	IAEA 1992, page 17
tree swallow and house wren nestlings	0.9 (90)			No growth effects.	IAEA 1992, page 17
tree swallow and house wren nestlings	2.6 (260)			growth slightly depressed.	IAEA 1992, page 17
tree swallows, eastern bluebirds, house wrens	0.9 – 6 (90 – 600)			Progressively reduced growth from increasing acute exposure.	UNSCEAR 1996, para. 147
tree swallows, eastern bluebirds, house wrens	4 and 6 400 and 600			Reduced body mass, 10% and 13% respectively. Also affected feather length and foot length.	UNSCEAR 1996, para. 147
swallows and wrens		30 – 250 (0.072 – 0.6)		Appeared essentially normal, 18-160 uCi/kg/day, radionuclide unnamed.	IAEA 1992, page 20
swallows and wrens		42 (1)		No effects observed.	IAEA 1992, page 20
swallows and wrens		<50 (< 0.12)		No observed differences in sperm count.	IAEA 1992, page 20
eastern bluebird nestling	30 (3,000)			Lethal dose.	UNSCEAR 1996, para. 148

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
male weaver finch	0.5 - 2.1 (50 – 210)			No testicular damage.	IAEA 1992, page 17
male weaver finch	4 (400)			Apparent abnormalities induced.	IAEA 1992, page 17
black headed gull egg	12 – 13 (1,200 – 1,300)			LD_{50} at hatching, irradiated on day 10 of development.	UNSCEAR 1996, para. 149
black-headed gull embryo	9 (900)			LD_{50} at hatching, irradiated at Day 10 of development.	IAEA 1992, page 17
passerine birds		8,333 (20)		Embryonic mortality.	IAEA 1992, page 21
any bird	≤ 0.5 (≤50)			Most effects studied show thresholds at higher doses	IAEA 1992, page 17
wild birds	5 – 12 (500 – 1,200)			LD_{50} in the same range as small mammals.	UNSCEAR 1996, para. 146
various bird species	4.6 – 30 (460 – 3,000)			$LD_{50/30}$	IAEA 1992, page 16
birds		30 – 260 (0.072 – 0.624)		No apparent effect study of nesting success, irradiated with large γ ray source	UNSCEAR 1996, para. 150
birds		8,400 - 42,000 (20 – 100)		Embryonic mortality study of nesting success, irradiated with large γ ray source.	UNSCEAR 1996, para. 150
Amphibians and Reptiles					
juvenile toads	0.1 (10)			$LD_{50/50}$ values. Irradiated tadpoles failed to metamorph.	UNSCEAR 1996, para. 152
toad tadpoles	0.1 (10)			$LD_{50/50}$ values. Irradiated tadpoles failed to metamorph.	UNSCEAR 1996, para. 152

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
toads	3 – 20 (300 – 2,000)			Induction of abnormalities in and reduced survival of offspring after paternal exposure.	UNSCEAR 1996, para. 152
adult toads	18 (1,800)			$LD_{50/50}$ values. Irradiated tadpoles failed to metamorph.	UNSCEAR 1996, para. 152
frogs, salamanders, turtles, snakes, lizards	2 – 22 (200 - 2,200)			LD_{50} with mean survival times following irradiation ranging up to 190 days. Cause of death usually damage to the haematopoietic system.	UNSCEAR 1996, para. 151
four species of amphibians	0.8 – 7 (80 – 700)			LD_{50} values, after allowing for up to 200 days for post radiation mortality	UNSCEAR 1996, para. 152
four species of amphibians	10 (1,000)			100% lethal	UNSCEAR 1996, para. 152
common side-blotched lizard	4.5 (45)			Irradiation of gonads lead to substantial decrease in production of offspring, by the next year, natality and population density was recovering.	UNSCEAR 1996, para. 151
common side-blotched lizard	50 (5,000)			Temporary sterility but recovery to normal spermatogenesis well advanced by day 48 after irradiation.	UNSCEAR 1996, para. 151
common side-blotched lizard		830 (2)	5 years	No apparent effects to sex ratio, maximal life span, and age distribution.	UNSCEAR 1996, para. 153
western whiptail		228 – 285 (0.55 – 0.68)	5.5 years	Lack of reproduction.	UNSCEAR 1996, para. 154
western whiptail		228 – 285 (0.55 – 0.68)	7.5 years	Female ovaries regressed completely, one of three males sterile.	UNSCEAR 1996, para. 154
long-nosed leopard lizard		456 – 570 (1.1 – 1.4)	3.5 years	Lack of reproduction.	UNSCEAR 1996, para. 154

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
long-nosed leopard lizard		456 – 570 (1.1 – 1.4)	5.5 years	Female ovaries regressed completely, one of three males sterile.	UNSCEAR 1996, para. 154
Iguanid lizard spp		833 (2)	5 years	No significant differences in sex ratios, age distributions, or life spans. Iguanids mature earlier, produce more egg clutches, died earlier than other species tested,	IAEA 1992, page 20
other lizard spp		833 (2)	1 or 2 years	females became sterile, test population drifted towards extinction. Longer-lived species than Iguanids.	IAEA 1992, page 20
Fish, Freshwater					
medaka	3.9 - 17.3 390 – 1,730)	18,000 - 79,000 (43 – 190)	9.1 days	Exposed to either tritiated water or external Cesium-137 γ radiation from 3 hrs after fertilization to hatching. Little effect on hatching rate.	UNSCEAR 1996, para. 169
medaka	5 and 10 (500 – 1,000)			Temporary sterility with recovery apparent at 60 days after irradiation.	UNSCEAR 1996, para. 166
mosquito fish	12 – 54 (1,200 – 5,400)	14,000 - 54,000 (34 – 130)	40 days	Irradiated with Cobalt-60 γ rays; no increased mortality relative to controls.	UNSCEAR 1996, para. 169
medaka		>18,000 (>43.2)	9.1 days	Occurrence of vertebral anomalies resulted from irradiation with Cesium-137 γ rays	UNSCEAR 1996, para. 169
medaka		>35,000 (>84)	9.1 days	After exposure to β particles from tritium, larval survival to one month of age significantly decreased, and occurrence of vertebral anomalies.	UNSCEAR 1996, para. 169
medaka		79,000 (190)	9.1 days	Larval survival to one month of age significantly decreased after irradiation with Cesium-137 γ rays	UNSCEAR 1996, para. 169

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
butterfly splitfin *Ameca splendens*	0.95 (95)	7,300 (17.5)	5.4 days	Disruption of spermatogenesis from irradiation Cesium-137 γ rays.	UNSCEAR 1996, para. 171
butterfly splitfin *Ameca splendens*	9.7 (970)	7,300 (17.5)	est. 8 weeks	Sterility from irradiation with Cesium-137 γ rays. Recovery to 60-70% of controls occurred after 236 days of recovery.	UNSCEAR 1996, para. 171
Gambusia affinis (Mosquitofish) in White Oak Lake		167 µGy/hr (0.4 rad/d) in 1965 declining to 83 µGy/hr (0.2 rad/d) in 1971 and 25 µGy/hr (0.06 rad/d) in 1975		Significantly more dead and abnormal embryos in irradiated pop compared to control pop, but larger brood size in irradiated pop. Increased frequency of deleterious genes in irradiated pop, but no overall effect on viability of pop as a whole.	IAEA 1992, page 26
adult female loach	2.5 (250), 5 (500), 10 (1,000), 20 (2,000)			Exposure to 2.5, 5, and 10 Gy produced slight effects in mature oocytes. At 20 Gy, clear response in both mature and developing oocytes	UNSCEAR 1996, para. 166
Roach Rutilus lacustris		83 – 192 µGy/hr (0.2 - 0.46 rad/d) internal and 125 – 417 µGy/hr (0.3 – 1 rad/d) from bottom sediments		Detrimental effects to fecundity.	IAEA 1992, page 26
silver salmon embryos	0.3 (30)			LD_{50} at hatching, irradiated at single cell stage.	UNSCEAR 1996, para. 165

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
silver salmon embryos	0.16 (16)			LD_{50} at 90 days post hatching, irradiated at single cell stage.	UNSCEAR 1996, para. 165.40
rainbow trout	6 and 8 (600and 800)			> 50% incidence of sterility when irradiated late in embryonic development	UNSCEAR 1996, para. 167
Fish, Saltwater					
plaice	0.09 (9)			LD_{50} following irradiation at blastula stage and assessed at metamorphosis.	UNSCEAR 1996, para. 165
Plaice		3 – 25 (0.007 - 0.06)		No apparent effects on the population from radiation.	IAEA 1992, page 25
Plaice		11.4 (0.027)		Normal frequency of chromosome aberrations.	IAEA 1992, page 25
6 species of marine fish	9 – 23 (900 – 2,300)			$LD_{50/50}$	UNSCEAR 1996, para. 165
Freshwater Aquatic Invertebrates					
Daphnia pulex	120 (12,000)	220,000 (528)	est. 23 days	Irradiation with Cobalt-60 γ rays, lifetime median dose, fixed per capita food supply. Little effect on age-specific survival noted.	UNSCEAR 1996, para. 168
Daphnia pulex		167,000 – 542,000 (400 – 1,300)		The maximum dose rates from Cobalt-60 γ source that were compatible with survival of population.	IAEA 1992, page 26
Daphnia pulex		35,000 - 45,000 (84 – 108)	est. 23 days	Increase in total death rate in food-limited populations.	UNSCEAR 1996, para. 168
Physa snail	22 (2,200)	11,000 (26.4)	12 weeks	Radiation from Cobalt-60 γ rays significantly reduced egg production.	UNSCEAR 1996, para. 170
Physa snail	26 (2,600)	11,000 (26.4)	14 weeks	Radiation from Cobalt-60 γ rays significantly reduced egg production.	UNSCEAR 1996, para. 170

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
Physa snail	45 (4,500)	10,000 (24)	24 weeks	No significant effect on reproduction, mortality, or size.	IAEA 1992, page 26
snail		100,000 (240)	lifespan	Radiation from Cobalt-60 γ source had significant effect on reproduction, mortality, or size	IAEA 1992, page 26
snail pop in White Oak Lake		250 (0.6)		Frequency of egg capsule production was reduced. However, egg production between irradiated and non-irradiated populations was similar, because irradiated population produced an increased number of eggs per capsule.	IAEA 1992, page 25
snail		10,000 (24)	lifespan	Radiation from Cobalt-60 γ source had no significant effect on reproduction, mortality, or size of snail	IAEA 1992, page 26
midge and snail populations		263 (0.63)		Increased frequency of chromosome aberration, but no apparent impact to populations	IAEA 1992, page 25
Saltwater Invertebrates					
Blue crabs	38.4 (3,840)	32,000 (77)	50 days	No effect.	UNSCEAR 1996, para. 168
blue crabs	330 (33,000)	290,000 (696)	50 days	Radiation from Cobalt-60 γ rays resulted in mortality greater than controls.	UNSCEAR 1996, para. 168
Blue crabs		32,000 (77)		No effect to growth rate or survival.	IAEA 1992, page 26
blue crabs		73,000 (175)		No effect to growth rate or survival.	IAEA 1992, page 26
Blue crabs		290,000 (696)		Significant reduction in growth rate and survival.	IAEA 1992, page 26
juvenile marine scallops and clams	80 – 88 (8,000 – 8,800)	9,000 - 10,000 (22 – 24)	est. 370 days	No effect on survival.	UNSCEAR 1996, para. 168
Polychaete, *Neanthes arenaceodontata*		2,100 (5)		No significant effects.	UNSCEAR 1996, para. 170

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
Polychaete, *Ophryotrocha diadema*		3,200 (7.7)	Seven generations	Lowest dose rate to produce a significant effect, which was a reduction in larvae in generation 2.	UNSCEAR 1996, para. 170
polychaete *Neanthes arenaceodontata*		17,000 (41)		Significant effects in generation one.	UNSCEAR 1996, para. 170
10 marine invertebrates	2 – 680 200 – 68,000)			$LD_{50/60}$	UNSCEAR 1996, para. 165
Terrestrial Invertebrates					
earthworms	20 (2,000)			Significantly reduced hatching success.	UNSCEAR 1996, para. 156
lumbricidae		1000 (2.4)		Unidentified effects, but lumbricidae was considered most sensitive soil invertebrate tested.	IAEA 1992, page 30
invertebrate populations in soil	100 (10,000)			LD_{50}; Survival of the majority of the identifiable taxonomic groups was less than 50% of the control values at acute doses < 100 Gy	UNSCEAR 1996, para. 156
6 insect species	20 – 40 (2,000 – 4,000)			50% reduction in mean life span.	UNSCEAR 1996, para. 156
11 insects & 1 isopod	80 (8,000)			Significant reduction of life expectancy.	UNSCEAR 1996, para. 156
grasshoppers	4, 13, 7, 8, 8 (400; 1,300; 700; 800; 800			$LD_{50/20}$ for eggs, 1st & 3rd instar; 2nd instar; 4th instar, and adult, respectively.	UNSCEAR 1996, para. 160

Appendix A. Summary of Representative Radiation Toxicity Data

Species	Total Dose, Grays (Rads)	Dose Rate, µGy/hr (rad/d)	Exposure time	Effect/Comment	Source
cricket	50 (5,000)	300,000 (720)		10% mortality in 20 days.	UNSCEAR 1996, para. 159
cricket	50 (5,000)	>2,000,000 (>4,800)		$LD_{50/20}$	UNSCEAR 1996, para. 159
Terrestrial Plants					
Pine forests	1 – 5 (100 – 500)		short term	Minor changes in productivity and reproduction, from which rapid recovery is expected following removal of radiation source.	UNSCEAR 1996, para. 222
Pine forests	20 (2,000)		short term	Mortality of almost all higher plants; recovery would take decades to centuries.	UNSCEAR 1996, para. 222
Pine forests		400 - 4,000 (1 – 10)	long term	Minor changes in productivity and reproduction, from which rapid recovery is expected following removal of radiation source.	UNSCEAR 1996, para. 222
Pine forests		>40,000 (>100)	long term	Mortality of almost all higher plants; recovery would take decades to centuries.	UNSCEAR 1996, para. 222

Appendix B. Radiation Units and Conversions

VARIABLE	OLD UNIT	NEW UNIT	CONVERSIONS
Activity	Curie (Ci) = $3.7x10^{10}$ dps 1 mCi = 0.001 Ci (milli) 1 µCi = 0.000001 Ci (micro) 1nCi = 0.000000001 Ci (nano) 1pCi=0.000000000001 Ci (pico)	Becquerel (Bq) = 1 dps kBq = 1,000 Bq (kilo) MBq = 1,000,000 Bq (mega) GBq = $1x10^9$ Bq (giga) TBq = $1x10^{12}$ Bq (tera) PBq = $1x10^{15}$ Bq (peta) EBq = $1x10^{18}$ Bq (exa)	1 Ci = $3.7x10^{10}$ Bq 1 Ci = 37 GBq I pCi = 0.037 Bq 1 Bq = 27 pCi
Exposure, field intensity in free air	Roentgen (R) = $2.58x10^{-4}$ coulombs/kg	Coulomb/kg (C/kg)	1 R = $2.58x10^{-4}$ C/kg
Absorbed dose	Rad = 100 erg/g	Gray (Gy) = 1 J/kg	1 Rad = 0.01 Gy 1 Gy = 100 Rad
Dose equivalent	Rem = damage effects of 1 R	Sievert (Sv) = 1 J/kg	1 Rem = 0.01 Sv

Coulomb: The amount of electric charge transported in 1 second by a steady current of 1 ampere (1C = 1A·1s) (6.241 509 629 152 65×10^{18} electrons (i.e., elementary charges).

Curie: The number of disintegrations per second in 1 gram of radium

dps = disintegrations/second

Dose equivalent: Absorbed dose in Gy times the Relative Biological Effectiveness (RBE) weighting factor for the type of radiation and the target organism.

Erg: A unit of energy. An erg is equal to the force of one dyne exerted for a distance of one centimeter; it is equal to one gram centimeter squared per second squared ($g{\cdot}cm^2/s^2$). It is equal to $1x10^{-7}$ Joules or 100 nanojoules. 1 erg = $6.2415x10^{11}$ eV. 1 dyne cm = 1 erg

J = joule: The work done, or energy spent, by a force of one Newton moving one meter along the direction of the force, aka Newton-meter or N·m. Work done to move an electric charge of one coulomb through a potential difference of one volt; one coulomb volt (C·V); the work done to produce one watt continuously for one second (W·s).

Appendix C: Characteristics of Important Radionuclides

The following table provides detailed information about the 23 radionuclides for which the (USDOE 2002) developed Biota Concentration Guidelines for. The information from this table was taken from (USDOE 2002 and/or Eisler 1994).

Radionuclide	**Symbol**	**Half-life**	**Emission type**	**Decay Energy MeV**[a]	**Specific activity GBq/g**
Americium-241	^{241}Am	458 years	α	0.0575	118.4
Cerium-144	^{144}Ce	284 days	β	1.3517 (D)	118,400
Cesium-135	^{135}Cs	3,000,000 years	β	0.0563	0.03
Cesium-137	^{137}Cs	30.2 years	β	0.7966 (D)	3626
Cobalt-60	^{60}Co	5.3 years	β	2.6016	40,700
Europium-154	^{154}Eu	8.593 years	β	1.5269	5,550
Europium-155	^{155}Eu	15.2 days	β	0.1224	51,800
Hydrogen-3	^{3}H	12.26 years	β	0.0057	358,900
Iodine-129	^{129}I	16,000,000 years	β	0.0789	0.01
Iodine-131	^{131}I	8 days	β	0.5715	4,440,000
Plutonium-239	^{239}Pu	24,110 years	α	0.0056	2.29
Radium-226	^{226}Ra	1,620 years	α	2.7023 (D)	37
Radium-228	^{228}Ra	5.75 years	β	1.3677 (D)	8,510
Antimony-125	^{125}Sb	2.7 years	β	0.5670 (D)	51,800
Strontium-90	^{90}Sr	29 years	β	1.1305 (D)	5,550
Technetium-99	^{99}Tc	213,000 years	β	0.0846	0.48
Thorium-232	^{232}Th	14,000,000,000 years	α	0.0121	0.000004
Uranium-233	^{233}U	160,000 years	α	0.0037	0.35
Uranium-234	^{234}U	245,000 years	α	0.0128	0.23
Uranium-235	^{235}U	710,000,000 years	α	0.3729 (D)	0.000078
Uranium-238	^{238}U	4,470,000,000 years	α	0.9154 (D)	0.000012
Zinc-65	^{65}Zn	244 days	*	0.5904	296,000
Zirconium-95	^{95}Zr	65 days	β	1.6614 (D)	777,000

[a] Total energy of all photons and electrons emitted per decay of radionuclide. A (D) indicates that the estimate of decay energy assumes the radionuclide is in equilibrium short-lived decay products.
* Zinc-65 emits a positron and X-rays from electron capture.