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# Radionuclide bioaccumulation and transfer in aquatic food webs

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## Abstract

With an increasing reliance on nuclear power as a low-carbon energy source, understanding the pathways and dynamics of planned or unplanned radionuclide releases to the environment and subsequent transfer within food webs is essential for assessing environmental risks. This essay explores the mechanisms by which radionuclides accumulate in aquatic organisms and transfer through food webs. Factors influencing bioaccumulation, including environmental parameters, radionuclide properties, and organismal characteristics, are reviewed. Additionally, this essay investigates the role of food web structure in mediating the transfer of radionuclides, highlighting the influence of climate-induced changes in food web structure. The findings underline the complexity of radionuclide contaminant pathways in aquatic ecosystems and emphasize the need for research on the impact of shifting food web structures due to climate change on radionuclide transfer. This understanding is crucial for assessing and mitigating future environmental risks associated with nuclear energy production.

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# Abbreviations

| | |
|---|---|
| NPP | Nuclear power plant |
| SIA | Stable isotope analysis |

# 1. Introduction

In light of the increasing global demand for electricity, which coincides with accelerated climate change, the importance of identifying and using energy sources capable of delivering substantial amounts of energy while maintaining a minimal carbon footprint has grown significantly (Ahmad & Zhang 2020). Nuclear power stands as the leading low-carbon emission, major energy system electricity producer in the world (IEA 2019). However, the use of nuclear power plants (NPPs) for energy is highly debated due to the potential release of ionising radiation, nuclear waste contamination, and the risks posed in the event of an accident (Ho & Kristiansen 2019). NPP accidents can release large quantities of radioactive isotopes (hereafter referred to as radionuclides) into the atmosphere and surrounding areas, resulting in detrimental effects on the environment and human populations (Strupczewski 2003; Christodouleas et al. 2011). Radionuclides with long half-lives, such as Caesium-137 ($^{137}$Cs) and Strontium-90 ($^{90}$Sr), with physical half-lives of 30 and 28 years respectively, can remain in the environment for decades after a radionuclide-releasing event and may potentially enter and bioaccumulate in organisms in affected areas (Pröhl et al. 2006; Zalewska & Suplińska 2013).

In high concentrations, radionuclides can have detrimental health effects on aquatic organisms (Kryshev & Sazykina 1998; Sazykina & Kryshev 2003; Lerebours et al. 2018). However, not all organisms in an affected area demonstrate equal concentrations of radionuclides, even if the environmental concentration (i.e., that in water or sediments) is the same (Forseth et al. 1998; Lerebours et al. 2018). The difference in radionuclide concentrations in tissues between organisms may depend on their differences in prey choice, trophic level, and habitat (Bezhenar et al. 2016). Food web structure, the arrangement of how organisms in an ecosystem are connected through their feeding interactions and flow of energy and matter, plays a significant role in the transfer of contaminants between organisms at different trophic levels (so-called trophic transfer; Alava et al. 2017), such as radionuclides. However, the stability of food webs is vulnerable to climate perturbations, with species' responses resulting in shifts in, e.g., biomass (du Pontavice et al. 2021). Climate-induced changes in biomass, driven by factors like warming, can trigger cascading effects on species interactions and, subsequently, food web structure (Bartley et al. 2019; Gibert 2019). Consequently, this can affect

the transfer of contaminants within these ecosystems (Alava et al. 2017). This highlights the importance of considering climate change when studying the transfer of radionuclides within food webs.

To better understand the potential environmental effects of nuclear accidents, and regular nuclear power plant operations under environmental change, there is a growing need to deepen our understanding of how radionuclides bioaccumulate in aquatic food webs (see Textbox 1 and Figure 1). Specifically, we must understand the mechanisms by which bioaccumulation might be affected by how food webs respond to climate change. In this essay, I aim to provide both an overview of radionuclides in the environment and discuss the current literature on how changes in food web structure resulting from disturbance and environmental change, may affect the transfer of these hazardous substances within food webs.

# 2. Definitions

Radionuclides and other contaminants can enter and bioaccumulate in food webs through various pathways (see Textbox 1 and Figure 1). Besides being taken up directly from water, also known as bioconcentration (i.e., uptake via respiration of dissolved contaminants), contaminants can enter the food web through ingestion of contaminated food items. The transfer of a substance from one organism (the resource) to another (the consumer) through consumption is called trophic transfer, or sometimes, biotransfer (Ali & Khan 2019). When a substance is accumulated in an organism over time at a faster rate than the substance is eliminated, it is referred to as bioaccumulation (Streit 1998; Ali & Khan 2019). Bioaccumulation covers the uptake into an organism from all environmental sources (e.g., food, water, sediment). If the concentration of a substance increases (decreases) as it moves up t.he food web, it is called biomagnification (biodilution; Ali & Khan 2019).

*Table 1. Definitions. Adapted from Ali & Khan (2019).*

| **Expression** | **Definition** |
|---|---|
| Trophic transfer | (or biotransference) the transfer of a substance from organisms at one trophic level to organism(s) at the next. |
| Bioconcentration | the accumulation of a substance in an organism as a result of its uptake from the ambient abiotic environment. |
| Dietary accumulation | the accumulation of a substance in an organism as a result of its uptake from the organism's diet. |
| Bioaccumulation | the accumulation of a substance in an organism as a result of its uptake from both the ambient abiotic environment and the organism's diet. |
| Biomagnification | Biomagnification the increase of a substance along successive increasing trophic levels in food webs. |
| Biodilution | Biodilution the decrease of a substance along successive increasing trophic levels in food webs. |

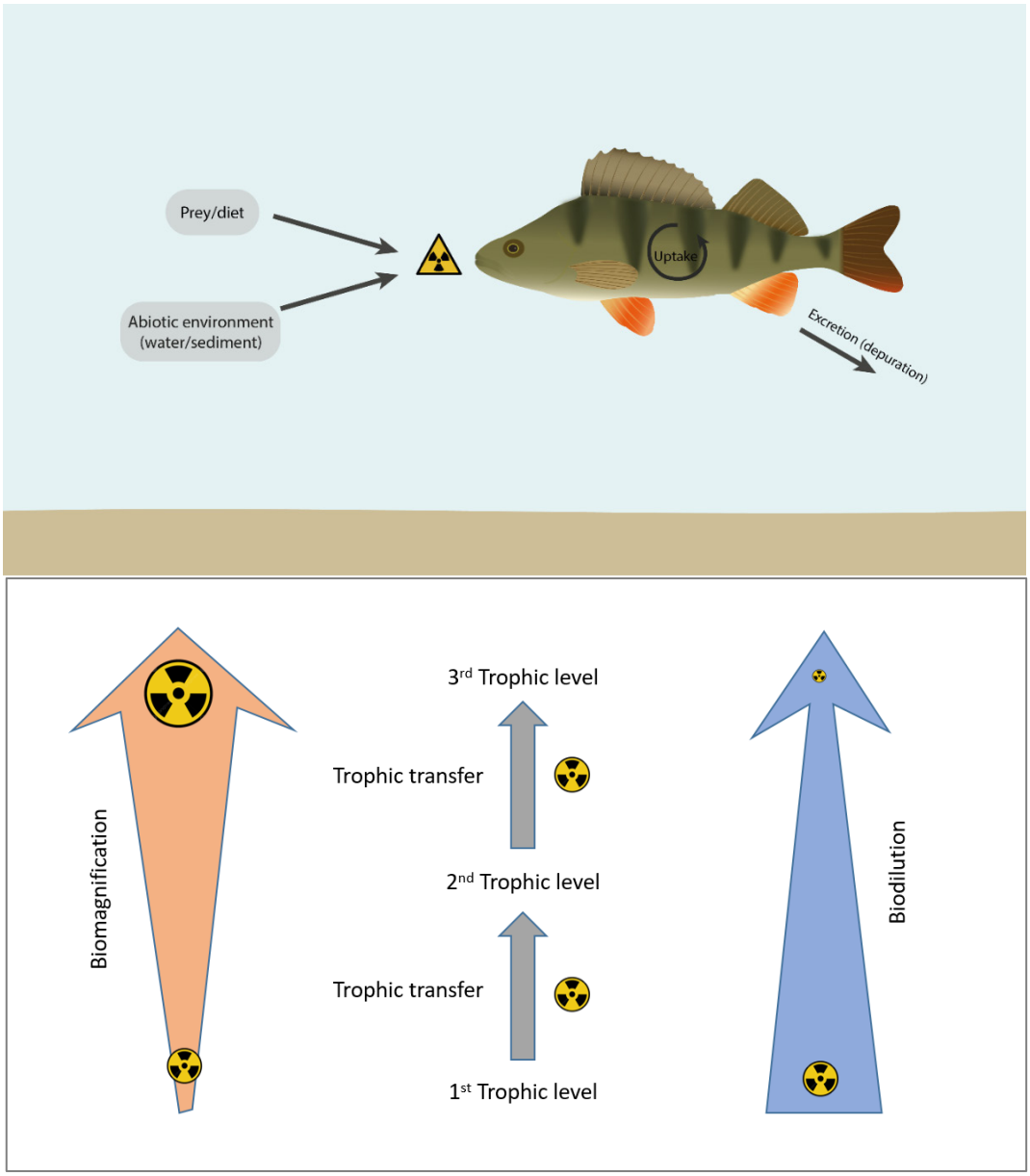


*Figure 1. Schematic summary of terms and concepts involved in the bioaccumulation and transfer of radionuclides in food webs. (a) Illustrates dietary based accumulation, bioconcentration, and bioaccumulation. (b) Illustrates trophic transfer, biomagnification,and biodilution.*

# 3. Radionuclides in the environment

## 3.1. Radionuclides

Radionuclides, also known as radioisotopes or radioactive isotopes, are atoms that have an unstable nucleus and emit ionising radiation (IAEA 2016). In contrast to atoms with non-ionising radiation, such as visible light and microwaves, ionising radiation possesses sufficient energy to disrupt regular cellular processes, potentially leading to cell death or the transformation into cancerous cells (Blaylock et al. 1996). Not all radionuclides possess the same levels of radiotoxicity, meaning some radionuclides are more hazardous than others (IAEA 1963; see Table 1 for examples).

Radionuclides emit radiation in the form of alpha (α), beta (β), or gamma (γ) radiation. Alpha particles are the least penetrating of the three types of radiation and consist of two protons and two neutrons from the atom's nucleus and are positively charged. Alpha radiation can be stopped by a sheet of paper and poses a significant hazard only if the radioactive substance is ingested or inhaled, i.e., bioaccumulated. Beta particles consist of high-energy electrons and are more penetrating than alpha radiation but less penetrating than gamma radiation. Beta particles can penetrate through human skin and cause harm if a significant amount is ingested or inhaled. In contrast to alpha and beta particles, gamma rays are pure energy and do not have any mass. Gamma radiation can pass through several centimetres of lead and several metres of concrete (depending on the radionuclide and associated gamma energy released) and may cause harm if bioaccumulated in a significant amount (Poschl & Nollet 2006; Benitez-Nelson et al. 2018c). Aquatic organisms can be exposed to radiation both internally, through respiration of contaminated water or ingestion of contaminated food and sediment, and externally, known as irradiation, from radionuclides in the surrounding water or sediment (Benitez-Nelson et al. 2018

*Table 2. Examples of common radionuclides (Benitez-Nelson et al. 2018; Radiation - Accumulation, Critical Organs | Britannica 2024)*

| Radionuclide | Physical half-life | Effective half-life | Radiation type | Radio-toxicity | Radio-logical analogue | Body tissue accumulation (in mammals) | Hazard (for mammals) | Typical origin |
|---|---|---|---|---|---|---|---|---|
| Polonium-210 ($^{210}$Po) | 138.4 d | 42 d | Alpha | Very High | | Spleen | Cancer | Natural – decay of uranium |
| Radion-226 ($^{226}$Ra) | 1600 y | 44 y | Alpha | Very High | | Bone | Bone cancer, anaemia, cataracts | Natural – decay of uranium |
| Strontium-90 ($^{90}$Sr) | 28 y | | Beta | Very High | Calcium | Bones | Bone cancer | Nuclear weapons testing, Chernobyl NPP accident |
| Cesium-137 ($^{137}$Cs) | 30.17 y | 70 d | Beta and gamma | High | Potassium | Muscles | Reproductive and developmental effects, damaged immune system, cancer (leukaemia, thyroid cancer, tumours) | Chernobyl NPP accident |
| Cesium-134 ($^{134}$Cs) | 2 y | | Beta and gamma | High | Potassium | Muscles | As for $^{137}$Cs | Chernobyl NPP accident |
| Cobalt-60 ($^{60}$Co) | 5.27 y | 0.75 d | Beta and gamma | High | Iron | Gastro-intestinal Tract | Anaemia, immune system dysfunction, damage to tissues and organs | Baltic NPPs |
| Iodine-129 ($^{129}$I) | 1570 y | 140 d | | High | | Thyroid | Thyroid cancer | Nuclear weapons testing, NPPs |
| Iodine-131 ($^{131}$I) | 8.02 d | 7.6 d | Beta and gamma | High | Stable iodine | Thyroid | Thyroid cancer | Nuclear weapons testing, Chernobyl NPP accident, radiation treatment |
| Tritium ($^{3}$H) | 12.32 y | 12 d | Beta | Low | | Full body | Cancer | Natural and NPPs |
| Potassium-40 ($K^{40}$) | 1.25 by | 30 d | Gamma | Low | Potassium | Full body | No hazard | Primordial |

## 3.2. Radionuclides exposure to aquatic environments and organisms

Radionuclides are released into the aquatic environment from both natural and anthropogenic sources (see Table 2). Some naturally occurring radionuclides, such as $^{3}H$, $^{14}C$, and $^{7}Be$, originated either from the formation of Earth or are continuously being generated by cosmic rays interacting with materials on Earth and in the atmosphere (Poschl & Nollet 2006; Benitez-Nelson et al. 2018a). Other primordial and naturally occurring radionuclides include $^{40}K$ and the U-Th series radionuclides (e.g., $^{238}U$, $^{234}Th$, $^{210}Pb$, $^{223, 224, 226, 228}Ra$, and $^{222}Rn$) (Benitez-Nelson et al. 2018b).

Four major known sources of anthropogenic releases of radionuclides, such as $^{3}H$, $^{14}C$, $^{90}Sr$, $^{134, 137}Cs$, $^{60}Co$, $^{129}I$, and $^{238, 239, 240}Pu$, to aquatic environments include: (1) nuclear weapons atmospheric testing in the 1950s to 1960s in the Pacific Ocean, by USA and former Soviet Union, (2) nuclear power plant (NPP) accidents, (3) planned releases from nuclear fuel reprocessing plants, such as Sellafield in the UK and Cap de la Hague in France, and (4) regular NPP operations (IAEA 1999; IAEA 2001; Hu et al. 2010). Both the Chernobyl and Fukushima Dai-ichi NPP accidents released around 15-20 Peta Becquerel (PBq) each of the long-lasting $^{137}Cs$ to oceans (IAEA 1999; IAEA 2001; Hu et al. 2010).

Anthropogenic radionuclides enter aquatic systems primarily through atmospheric fallout and from land, e.g. riverine discharge (Zalewska & Saniewski 2011). Once in the aquatic environment, radionuclides may be transported and diluted by horizontal (e.g., with currents or biota; Madigan et al. 2012; Periáñez et al. 2019b) and vertical (e.g., with sinking detritus or inorganic matter) processes (Kansanen et al. 1991). In the water column, radionuclides can be absorbed by suspended particulate matter (including organic detritus) or bioaccumulated by organisms, such as plankton (Periáñez et al. 2019a). Radionuclides can then enter the pelagic food web, through dietary consumption and trophic transfer (Heldal et al. 2003), or the benthic food web (Holmerin et al. 2021), following sedimentation processes causing the radionuclides to accumulate in bottom sediments, mainly via adsorption to sinking sediment particles and detritus (Bezhenar et al. 2016).

## 3.3. Radionuclide exposure to the Baltic Sea

The Baltic Sea is a unique brackish and semi-enclosed sea surrounded by nine countries and over 85 million people within its catchment area (*Helcom : Baltic facts and figures* 2023). Due to the long retention time of the water in the Baltic Sea, it exhibits some of the highest levels of radionuclides in a waterbody in the

world (Ikäheimonen et al. 2009; Zalewska & Suplińska 2013; *Radioactivity – HELCOM* 2023).

$^{137}$Cs and $^{90}$Sr are two of the most prevalent anthropogenic radionuclides in the Baltic Sea (Zalewska & Suplińska 2013), occurring as a result of uranium fission. Most of the $^{137}$Cs in the Baltic Sea originate from the Chernobyl NPP accident in 1986, whereas, $^{90}$Sr mainly derives from nuclear weapons testing from the 1950s and 1960s (Zalewska & Suplińska 2013; Figure 2). The long half-lives of $^{137}$Cs and $^{90}$Sr are one of the explanations for why these radionuclides still are detected in the Baltic Sea. Regular NPP operation may release aerosols, such as $^{60}$Co, into the air, although these amounts are negligible (Lüning 2005).

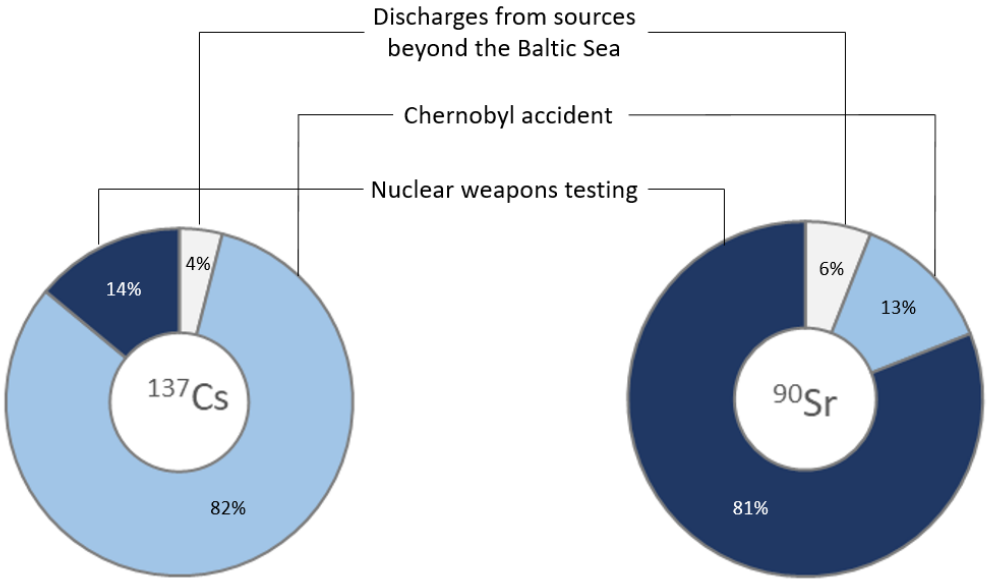


*Figure 2. Origin proportion for $^{137}$Cs and $^{90}$Sr in the Baltic Sea (Adapted and redrawn from Zalewska & Suplinska 2013)*

# 4. Radionuclides in organisms

## 4.1. Potential hazards to organisms

High or long-term exposure to ionising radiation can cause detrimental effects to humans and other organisms (Williams 2008). The effects can be either deterministic or stochastic. Deterministic effects mean that ionising radiation can, at a certain threshold dose, cause predictable cell injury which increases in severity with increased dose. Stochastic effects explain the probability of permanent DNA alterations, potentially leading to cancer (ICRP 2007). Examples of radiation effects in organisms surrounding the Chernobyl NPP after the accident in 1986 include reduced fertility (Møller et al. 2014; Lerebours et al. 2018), developmental errors (Hoffmann 2001), and damaged immune system (Morley 2012; Kesäniemi et al. 2019). Environments with significant background radiation can also lead to oxidative stress in organisms (Hoffmann 2001; Bonisoli-Alquati et al. 2010), and, consequently, reduced brain size, which has been found for birds inhabiting the Chernobyl NPP exclusion zone (Bonisoli-Alquati et al. 2010; Møller et al. 2011).

Not all species within an affected area are equally sensitive to radiation. Lerebours et al. (2018) studied the effects of $^{137}$Cs and $^{90}$Sr in perch (*Perca fluviatilis*) and roach (*Rutilus rutilus*) in lakes near Chernobyl 30 years after the accident. They found that perch were more sensitive to radiation than roach. The perch matured later than expected and many perch individuals exhibited undeveloped fish gonads. To understand why radiation affects species and phenotypes differently, one must consider the central factors on how radionuclides enter and accumulate in body tissues, and transfer between organisms.

## 4.2. Factors that can affect the bioaccumulation of radionuclides

The accumulation of radionuclides by aquatic organisms is influenced by several factors, including environmental conditions (e.g., temperature and salinity) and habitat (e.g. pelagic vs benthic), the nature of the radionuclide (e.g., its physical-

chemical forms), and the type of organism (e.g., species physiology) and its diet and trophic level.

### 4.2.1. The environmental conditions

Temperature increases metabolic rates (Gillooly et al. 2001; Clarke & Fraser 2004; Huey & Kingsolver 2019), which can affect the uptake and excretion of radionuclides by aquatic organisms (Nakahara et al. 1977). Salinity has also been found to affect the bioaccumulation of radionuclides in aquatic organisms (Topcuoğlu 2001; Pouil et al. 2018). For instance, Pouil et al. (2018) found juvenile turbot (*Scophthalmus maximus*) to exhibit lower assimilation efficiency (i.e., assimilation of radionuclides from diet) of $^{137}$Cs with higher salinity, likely due to osmoregulation.

Fallout radionuclides move downwards towards and into the sediment with time through the process of sedimentation (Bossew & Kirchner 2004; Ikäheimonen et al. 2009). Once in the sediment, the sediment pH can influence the chemical form (speciation) of radionuclides (Iurian et al. 2015), which might contribute to more or less efficient uptake by organisms. For instance, inorganic forms tend to have lower gut absorption than organic forms of the same radionuclide when ingested (Carvalho 2018). As radionuclides tend to undergo sedimentation, they accumulate in benthic food webs more than in pelagic food webs (Bezhenar et al. 2016; Holmerin et al. 2021). Organisms residing and feeding from the benthos would, therefore, likely bioaccumulate more radionuclides than pelagic-feeding organisms.

### 4.2.2. The radionuclide

The physical state, such as if the radionuclide is solid, liquid, or gas, and the chemical speciation of the radionuclide can influence how it is absorbed, distributed, and excreted in living organisms. For example, a radionuclide like iodine-131 ($^{131}$I) can exist in different chemical forms, such as $^{131}$I bound to sodium (NaI), which is commonly used in medicine (Nofal et al. 1966). Radionuclide accumulation in organisms can also be affected by other factors influencing bioavailability. For example, $^{137}$Cs, $^{90}$Sr, and $^{60}$Co are easily taken up by organisms as they are chemically similar to essential elements such as potassium (taken up in muscles), calcium (taken up in bones), and iron (taken up in blood), respectively (Reichle et al. 1970; Desmet et al. 1991; Table 1). In addition, iodine is an essential element for many organisms, hence, radioactive iodine isotopes may be easily taken up by those organisms (Yavuz & Puckett 2023). Once the radionuclide has entered the body, it varies in how long it takes for it to exit, partly depending on the biological half-life of the radionuclide (Richmond 1980), and the rate of metabolism of the organism (Streit 1998; Yamazaki 2020).

### 4.2.3. The organism and its diet

Differences in radionuclide accumulation are likely due to the differences in biological and ecological factors (Elliott et al. 1992; Doi et al. 2012). Biological factors include metabolism (Yamazaki 2020; Doi et al. 2012; Kesäniemi et al. 2019), body size (Sundbom et al. 2003; Thomas & Fisher 2019), age (Pyle & Clulow 1998), and biological sinks, the body tissues in which they accumulate (Reichle et al. 1970). Fish with higher metabolic rates exhibit increased respiration and food intake, leading to greater exposure to contaminants (Brown et al. 2004; Doi et al. 2012). Enhanced metabolic rate also promotes faster uptake and distribution of contaminants throughout the body (Douben 1989). In contrast, fish with higher metabolic rates may also have an increased excretion rate, potentially leading to faster elimination of contaminants (Forseth et al. 1998). The metabolic rate might also vary depending on temperature, where, until a certain point, higher temperatures are correlated with higher metabolism which may affect the body size of the organism (Douben 1989; Ugedal et al. 1992; Gillooly et al. 2001; Brown et al. 2004; Lindmark et al. 2022). Several studies have found a relationship between radionuclide levels and body size (Elliott et al. 1992; Koulikov & Ryabov 1992; Kasamatsu & Ishikawa 1997; Thomas & Fisher 2019; Ishii et al. 2023). For instance, Ishii et al. (2023) found that radiocesium concentrations in fish increased with body weight, although the strength of the correlation varied between species. Growth rate might also affect contaminant concentrations in fish. Rapid growth might reduce contaminant concentrations in fish, a phenomenon also known as growth dilution (Karimi et al. 2007; Niizeki et al. 2020). However, growth dilution only happens if the fish is not exposed to the contaminant as it grows.

Ecological factors, including diet, are significant drivers of radionuclide bioaccumulation in organisms. For instance, benthivores might bioaccumulate radionuclides more than pelagic feeders as their diet might include organisms with higher concentrations of radionuclides due to radionuclide accumulation in the sediment (Holmerin et al. 2022). Examples of organisms with high uptake of radionuclides include filter-feeders, such as mussels, as they filter large volumes of water and retain particles, including radionuclides (Zuykov et al. 2013). Benthic gastropods might also take up radionuclides more readily than other organisms as they consume algae and bacteria in the sediment, which can have high radionuclide concentrations (Rothmeier et al. 2022). Organisms that can take up radionuclides easier than others can accumulate higher levels of these substances, making them more likely to, in turn, transfer high concentrations of radionuclides to other organisms higher up in the food web that predates on them. For instance, species with an exoskeleton or endoskeleton take up and bioaccumulate radionuclides, such as $^{90}$Sr, which is a calcium analogue (Reichle et al. 1970). This means that species without an exo- or endo-skeleton, as many soft-bodied invertebrates, might not bioaccumulate these radionuclides as much as species with an exo- or endo-

skeleton. The differences in radionuclide uptake and excretion between organisms, and between different types of radionuclides, may affect the transfer of radionuclides within food webs (Kryshev & Ryabov 2000; Topcuoğlu 2001; Pan & Wang 2016).

# 5. Trophic transfer in food webs

## 5.1. Food web structure and the transfer of substances

The interactions between species, and thus the structure of aquatic food webs, vary over space and time depending on the environment and the pressures the food webs are exposed to (Briand 1983; McMeans et al. 2015; Bartley et al. 2019) and within species across their ontogeny (Werner & Gilliam 1984). Changes in food web structure from pressures can occur due to either indirect effects, such as bottom-up or top-down processes (Lynam et al. 2017), or direct effects, such as species extinctions or invasions (Byrnes et al. 2007). For example, increased nutrient supply to an area might instigate an increase in the biomass of some primary producers, resulting in increased food availability for consumers who feed on these (i.e. a bottom-up effect; Rosenblatt & Schmitz 2016). Top-down effects can also have a strong impact on how the food web is structured. If a piscivorous species declines, from, for example, overfishing, this can result in an increased biomass of its prey fish population, which may lead to subsequent biomass reductions of the prey's resources, a process also known as trophic cascades (Paine 1980; Carpenter et al. 1985; Polis et al. 2000; Casini et al. 2008). Casini et al. (2008) reported a trophic cascade in the Baltic Sea, where overfishing of cod (*Gadus morhua*) directly increased the population of its main prey, the zooplanktivorous sprat (*Sprattus sprattus*), and indirectly the biomass of zooplankton, as well as chlorophyll-*a*, an indicator of phytoplankton biomass. Food web stability to such perturbations may depend on how biodiverse the system is (Paine 1966; Ives & Carpenter 2007). More diverse ecosystems tend to be more stable over time, resisting disturbances and maintaining their functioning (Ives & Carpenter 2007).

So far, I have discussed how alterations in biomass may influence food web structure. However, food web structure can also shift due to changes in the species feeding ecology. For instance, generalist feeders tend to cope better than specialist feeders in changing food webs as they can exploit a wide range of resources, allowing them to persist in various conditions (McPeek 1996). On the contrary, specialists rely on specific resources and may face challenges if those resources become scarce or change (McPeek 1996).

One useful method for examining food web structure is stable isotope analysis of the organisms interacting in the food web (SIA; Peterson & Fry 1987; Layman et al. 2012). SIA ultimately follows the principle of "*you are what you eat*" and can assist in establishing trophic position by analysing the ratio of $^{15}N$ to $^{14}N$ (expressed as $\delta^{15}N$) and carbon source by $^{13}C$ to $^{12}C$ (expressed as $\delta^{13}C$) in organisms (Rounick & Winterbourn 1986; Post 2002). $\delta^{15}N$ becomes enriched with trophic position and $\delta^{13}C$ can explain the source of dietary carbon where a low $\delta^{13}C$ indicates pelagic origin whereas a high $\delta^{13}C$ indicates a benthic/littoral origin (Solomon et al. 2011; Layman et al. 2012). Other stable isotopes used in aquatic systems include oxygen ($\delta^{18}O$), which is used to determine migratory history in organisms, as it reflects the temperature and salinity of ambient waters (Killingley 1980). Strontium ($\delta^{87}Sr$), is used to trace the origin and migratory history of anadromous fish (Hobson 1999). Additionally, sulphur ($\delta^{34}S$) is frequently used to distinguish marine- versus terrestrially-based diets (Lott et al. 2003; Connolly et al. 2004). SIA might also be conducted as a compound-specific analysis where the isotopic composition of individual organic molecules within an organism is isolated and analysed. This enables the identification of subtle variations in the diet and trophic position of organisms (Chikaraishi et al. 2009; Troch et al. 2012; McMahon et al. 2016). Although SIA is an important tool when studying food webs it has some limitations. For instance, SIA can only be conducted on the organisms sampled and therefore inferences about the rest of the population must be made. Furthermore, organism collection might be biased (e.g. using gillnets might not accurately sample the benthic fish food web) and might, therefore, not give a representative indication of the whole food web in an area. Another shortcoming is that it does not elucidate exactly what is consumed in the food web. However, visual stomach content analyses could supplement SIA to enhance this understanding. Although, stomach content analyses only provide information about the organism's diet as a snapshot, whereas SIA provides information about their general long-term diet. SIA has become an important tool used to assess trophic transfer of contaminants in food webs, in addition to evaluating the importance of carbon source on contaminant bioaccumulation (as seen with radionuclides by Ishii et al. 2023).

Food web structure plays a vital role in the way that contaminants move and accumulate in food webs (Alava et al. 2017; Figure 3). For instance, if a prey population which accumulates contaminants begins to increase in biomass relative to a less contaminated prey population, a generalist consumer fish might progressively begin to predominantly consume these more abundant, contaminated, organisms. To my knowledge, however, this has not yet been studied for radionuclides. Food chain length might also affect the trophic transfer of contaminants in food webs (Kidd et al. 1998). If the contaminant biodilutes, its concentrations at the top trophic levels might be negligible in a long food chain. On the contrary, top trophic levels might have high concentrations if the contaminant

biomagnifies in food webs (Ali & Khan 2019), but less so in a shorter compared to a longer food chain. The trophic transfer of contaminants in food webs might also be affected by the complexity of the food web. For instance, simpler food webs, characterized by low species diversity, might lead to greater biomagnification than complex food webs due to more limited prey choice for a given consumer in a simpler food web (Lavoie et al. 2013).

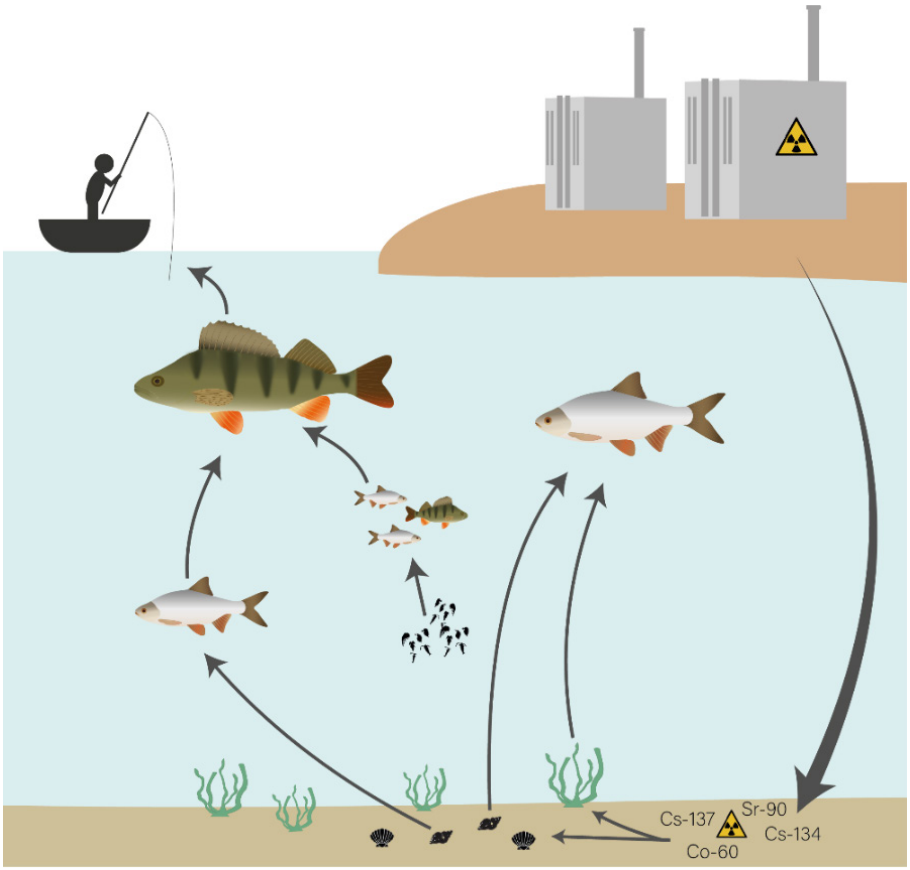


*Figure 3. An example of trophic transfer, indicated by arrows, of radionuclides in an aquatic food web.*

## 5.2. How warming affects the dynamics of aquatic food webs and potentially trophic transfer

Climate change and pollution are two of the major anthropogenic threats to aquatic systems (Alava et al. 2017). Climate change is predicted to increase sea water temperatures in European basins by 2-6 °C by 2100 under the high emissions scenario (Calvin et al. 2023), with the Baltic Sea being one of the fastest warming seas in the world (Dutheil et al. 2023). Warmer waters are predicted to change food

web structures in aquatic systems (Petchey et al. 1999; Bartley et al. 2019) and can strengthen trophic cascades (Svensson et al. 2017), cause changes in primary production (Häder et al. 2014), species compositions (Dijkstra et al. 2011; Gruner et al. 2017), and interspecific interactions (Milazzo et al. 2013; Durant et al. 2020). For instance, Svensson et al. (2017) found that warming compressed a food chain from four to three trophic levels when comparing a heated with a non-heated shallow coastal food web. A change in food web structure affects the contaminants pathways, and thus the rate of trophic transfer, in food webs (Noyes et al. 2009; Alava et al. 2017). Fish body growth might also be affected by warming due to changes in metabolism and prey availability (Gillooly et al. 2001; Desai & Singh 2009; Baudron et al. 2014; O'Gorman et al. 2016; Huss et al. 2019; van Dorst et al. 2019). For example, Huss et al. (2019) found that warming substantially increased the body growth of juvenile perch. An increase in body growth might allow juvenile perch to switch prey types earlier in life, potentially affecting radionuclide bioaccumulation. In addition, van Dorst et al. (2024) found that warming increased female perch growth, whereas it decreased growth of large males. Differences in metabolism due to varying growth rates are likely to affect the bioaccumulation of contaminants. An increase in metabolism due to warming (Gillooly et al. 2001) may also increase the excretion rate of contaminants (Yamazaki 2020), such as radionuclides.

Nuclear accidents and the projected increase in nuclear power plant operations may release high levels of radionuclides into surrounding aquatic ecosystems (Ho & Kristiansen 2019), potentially causing harmful effects on fish and other organisms (Kryshev & Sazykina 1998; Lerebours et al. 2018). Additionally, global warming is predicted to alter food web structures and feeding interactions (Bartley et al. 2019). Observational/field studies have the potential to study natural systems with