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EXECUTIVE SUMMARY

The accident at the Fukushima Dai-ichi Nuclear Power Plant (NPP) raised concerns about the possible impact to Australia from the release of radionuclides, including caesium-137 (^{137}Cs) and caesium-134 (^{134}Cs), to the ocean. Numerical ocean modelling indicates that these anthropogenic radionuclides will reach the Australian coastline around 2016 and therefore could potentially impact marine organisms.

This study was conducted to determine the background activity concentrations of ^{137}Cs and ^{134}Cs in commonly consumed seafood from northern Australia. In addition, the background activity concentrations of naturally occurring radionuclide polonium-210 (^{210}Po) were also determined in order to provide a comparison with the anthropogenic radionuclides. The results were used to estimate the committed effective dose to the public from the consumption of seafood.

Seafood samples, sourced from northern Australian waters, were analysed in this study to establish typical radionuclide levels in seafood prior to the arrival of any contamination from radionuclides released during the 2011 Fukushima Dai-ichi NPP accident.

The key findings of this study are:

- ^{137}Cs activity concentrations are consistent with the anthropogenic background radioactivity in the region due to global fallout, mainly from historical nuclear weapons testing.
- The absence of any measurable ^{134}Cs activity in all of the seafood samples analysed indicates that the anthropogenic activity detected is not from the Fukushima Dai-ichi NPP accident, which is consistent with the modelling predictions for contamination transport by Nakano and Povinec (2011).
- Based on a daily consumption of the seafood over one year, the committed effective dose contribution from ^{137}Cs to the Australian public is considered negligible with estimations less than 0.001% of the average background (1.5 mSv/year). The committed effective dose from ^{210}Po ingestion is less than 2% of the average background.
- There are no health implications due to the ingestion of seafood containing the amounts of ^{210}Po and ^{137}Cs measured in this study.

This study has established a benchmark for ^{137}Cs activity concentrations in seafood sourced from the northern waters of Australia. Further seafood collections and analysis will be undertaken after 2016 to ascertain if seafood sourced from northern waters of Australia is affected by the predicted arrival of contaminated water from the Fukushima Dai-ichi NPP accident.

1. INTRODUCTION

1.1 Background radiation in the oceans

Radionuclides, of both natural and anthropogenic origins, are present in the marine environment throughout the world. The weathering and dissolution of rocks containing radionuclides from the naturally occurring uranium and thorium series, and the emanation and decay of radioactive radon in the atmosphere, results in the deposition of naturally occurring radionuclides in the marine environment. These radionuclides enter the water column where they are dispersed or adsorbed to sediment and can be taken up by marine organisms.

The naturally occurring radionuclides of polonium (Po) accumulate in the tissues of marine organisms and concentrates up the trophic levels of the food chain. Polonium is known to be present in higher activity concentrations in marine organisms due to its solubility in water and continued input into the marine environment through natural geochemical and physical processes. ^{210}Po has the longest half-life ($t_{1/2}$ = 138.4 days) of all the natural isotopes of polonium and is produced as part of the decay of uranium-238 (^{238}U). ^{210}Po has a tendency to bind to amino acids and proteins and therefore concentrates in the muscle tissues of marine organisms (Carvalho, 2010). ^{210}Po therefore provides a benchmark to enable comparison to levels of other radionuclides, either from natural or anthropogenic origins, to assess potential health impacts to humans who consume seafood.

Anthropogenic radionuclides have also been deposited into the oceans, predominantly as a consequence of the atmospheric nuclear weapons testing conducted between 1945 and 1980 (UNSCEAR, 2000), and nuclear accidents such as the Chernobyl nuclear power plant accident in 1986 and the Fukushima Dai-ichi nuclear power plant accident (the Fukushima Dai-ichi NPP accident) in 2011. Radionuclides released to the environment are then transported around the world by atmospheric and oceanic circulation.

1.2 Radioactive materials released during the Fukushima Dai-ichi nuclear power plant accident

The Fukushima Dai-ichi NPP accident resulted in significant releases of radionuclides including caesium-137 (^{137}Cs), caesium-134 (^{134}Cs) and iodine-131 (^{131}I) to the atmosphere and the ocean, which in turn led to significant public concern in Japan and internationally regarding exposure to the radionuclides that were released into the environment. Estimates of the total release of ^{131}I and ^{137}Cs to the environment are 100-500 PBq and 6-20 PBq respectively. Of these releases it was estimated that 3-6 PBq of ^{137}Cs entered the ocean directly by leakage and deliberate discharge, and 5-8 PBq entered the oceans as a result of atmospheric fallout (UNSCEAR, 2013). ^{131}I has a short half-life of 8 days and was only a risk to public health in the weeks following the accident. ^{137}Cs and ^{134} Cs remain in the environment for longer, with radioactive half-lives of approximately 30 years and 2 years respectively.

Following the Fukushima Dai-ichi NPP accident most northern hemisphere countries detected small amounts of released radioactivity in air samples within weeks, and in the Asia-Pacific region of the southern hemisphere detection occurred about a month later (Stohl et al. 2012, Orr et al 2013). In the deep oceans radionuclides and other contaminants can take many years to transport in water across the equator.

Animals and plants (wildlife) living in the marine environment were also exposed to radionuclides released from the Fukushima Dai-ichi NPP accident, and in the most impacted regions this has resulted in uptake and incorporation of ^{137}Cs and ^{134}Cs into their tissues (MAFF, 2014). Some marine wildlife exposed to radionuclides can also travel long distances. For example, migratory tuna exposed to radionuclides from the Fukushima Dai-ichi NPP accident have been caught off the coast of the United States of America. These were found to contain low activity concentrations of ^{137}Cs and ^{134}Cs in their tissues (Madigan et al. 2012). In this case, the presence of the short lived ^{134}Cs was the indicator that this radionuclide was from the Fukushima Dai-ichi NPP accident, as any ^{134}Cs from the 1986 Chernobyl nuclear power plant accident would have already decayed to levels unable to be detected.

Figure 1.1 shows the results of long-term global dispersion modelling of the ^{137}Cs released to the marine environment following the Fukushima Dai-ichi NPP accident. It has been estimated that radioactive materials from the Fukushima Dai-ichi NPP could reach the northern waters of Australia via Malaysia and Indonesia between 2016 and 2021 (Nakano and Povinec, 2012). By the time the discharge from Fukushima reaches the waters off northern Australia, it is predicted that the continued dilution of these materials will result in ^{137}Cs activity concentrations of 100 to 300 mBq/m^3. At these levels the ^{137}Cs signature may be indistinguishable from the background ^{137}Cs currently present as a result of global fallout from historical nuclear weapons testing (Nakano and Povinec, 2012). ^{134}Cs was released in approximately a 1:1 activity ratio with ^{137}Cs but is less likely to be detectable by the time it is predicted to reach Australian waters due to its shorter half-life.

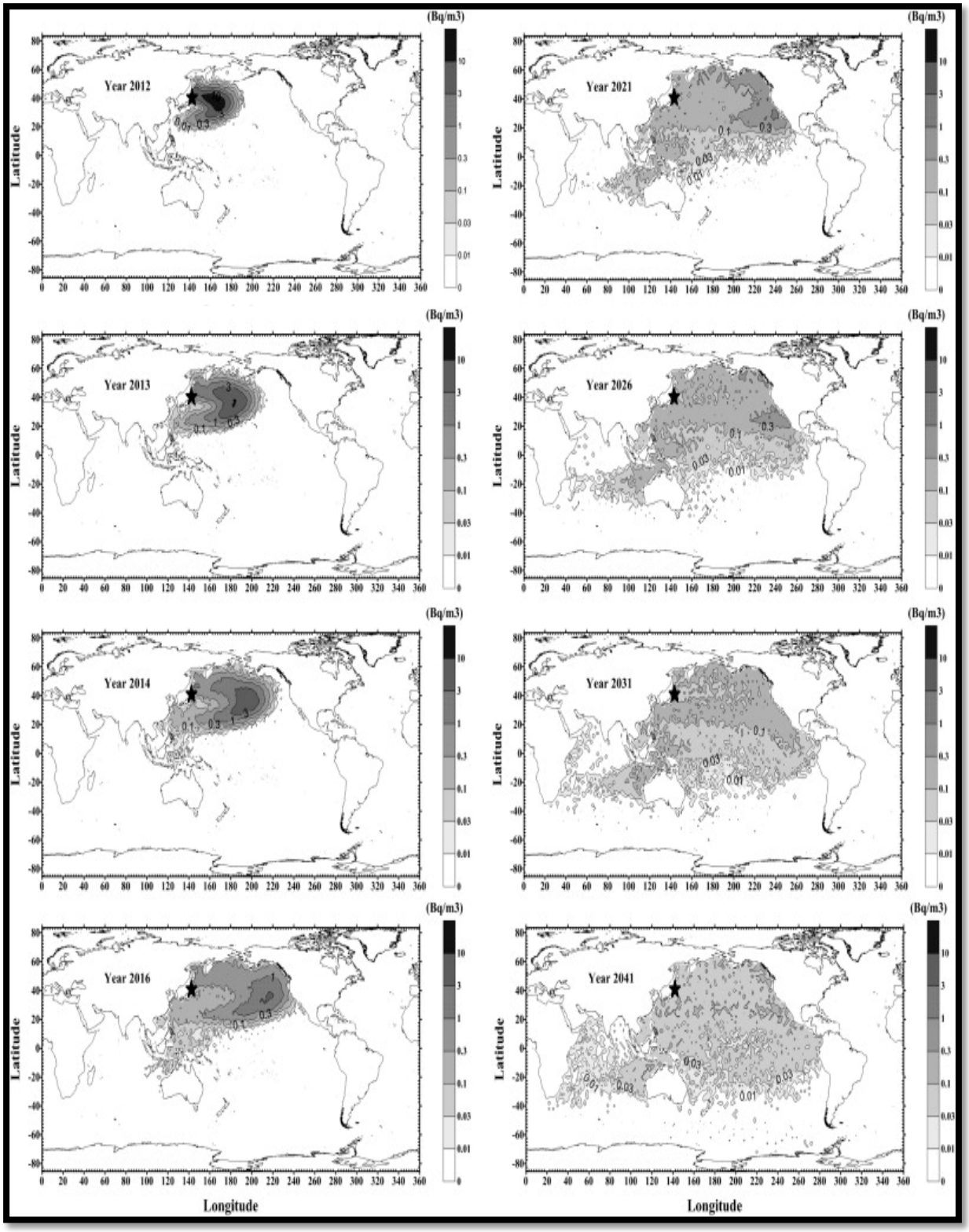

Figure 1.1: **^{137}Cs distribution in the world's oceans over time following radionuclide releases from the Fukushima Dai-ichi NPP accident in 2011. Reproduced from Nakano and Povinec (2012).**

1.3 Objectives

The aim of this investigation was to determine the background levels of the anthropogenic radionuclides, ^{137}Cs and ^{134}Cs, in edible portions of seafood sourced from northern Australian oceans, before the predicted arrival of radioactive caesium from the Fukushima Dai-ichi NPP accident to Australian waters. The levels of the naturally occurring radionuclide, ^{210}Po, were also determined for comparative purposes.

1.4 Scope

The scope of the Background Radioactivity in Northern Australian Seafood study was to:

- Collect representative samples of seafood from the waters of northern Queensland, the Northern Territory and the north of Western Australia.
- Ensure that samples were collected well before 2016, the earliest predicted arrival date of the radionuclides released from the Fukushima Dai-ichi NPP accident based on oceanographic modelling undertaken by Nakano and Povinec (2012).
- Determine the activity concentrations of ^{137}Cs, ^{134}Cs and ^{210}Po in the seafood samples to determine background levels of these radionuclides in Australian seafood.
- Assess the radiation dose and potential health impact of the target radionuclides based on the average annual dietary intakes of seafood in the Australian diet.
- Provide data that will inform future decisions for further monitoring of seafood in Australia beyond 2016.

2. METHOD

2.1 Selecting Sampling Locations

In Australia, seafood for human consumption is sourced from a variety of marine species occupying a range of trophic levels. Based on the predicted dispersion (see Figure 1.1) of contaminants from the Fukushima Dai-ichi NPP accident, the seafood for this study was sourced from the marine waters of northern Queensland, Northern Territory and northern Western Australia.

The Food and Agriculture Organisation of the United Nations (FAO) has established nineteen major marine fishing areas (see Figure 2.1) covering the waters of the Atlantic, Indian, Pacific and Southern Oceans, with their adjacent seas (FAO, 2014). In this study seafood was sourced from the major marine fishing areas 57 and 71, as these cover the Australian geographical regions of interest.

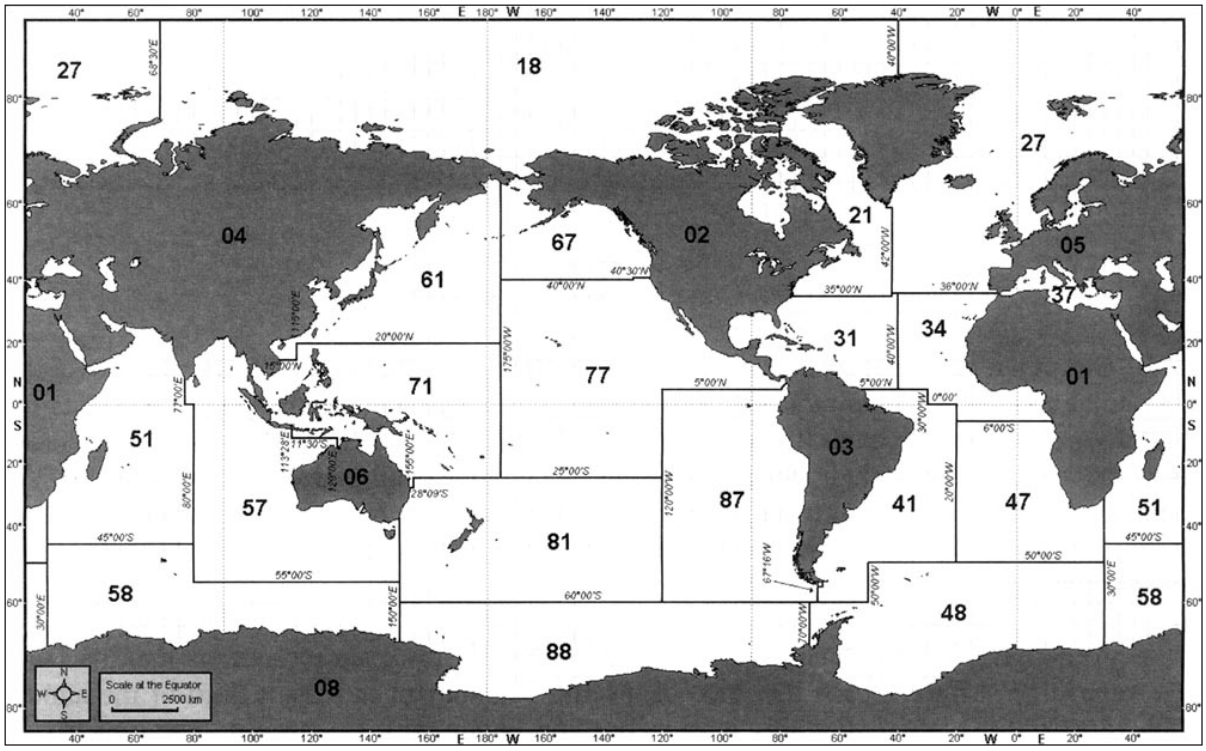

Figure 2.1: The nineteen Food and Agriculture Organisation (FAO) major fishing areas. Samples of seafood were sourced from the marine waters of Queensland and the Northern Territory (FAO Area 71) and the north of Western Australia (FAO Area 57). Reproduced from FAO (2014).

2.2 Selection of Seafood

Seafood was selected to represent a variety of marine biota. The following factors were considered:

- information on Australian seafood consumption
- geographical areas of capture (northern Australian waters)
- references containing species information including physical descriptions, distribution and feeding behaviour
- advice provided by wholesalers and retailers of seafood, which assisted in confirming the availability and sources of each seafood product and securing alternatives in the case where items of interest were not available.

Based on this information it was decided that the sample set should contain commonly consumed seafood sourced from the following groups:

- crustaceans (crabs, lobsters, bugs, prawns)
- pelagic predatory fish (Spanish mackerel, cobia)
- estuarine predatory fish (jewfish, threadfin salmon, barramundi, mangrove jack)
- reef predatory fish (coral trout, snapper)
- foraging fish (whiting)
- mollusc bivalves (scallops, oysters)
- mollusc cephalopods (squid, cuttlefish).

Although the sample set was small, it was considered to be adequate for the purpose of a study into the background radioactivity levels. The groups represented a mixture of trophic levels, habitats, behaviours and diets for the species identified.

2.3 Sourcing Seafood

Samples were obtained through direct purchase from seafood retailers within the three regions covered in this study. Sampling for Queensland was conducted in September 2012 when the seafood was purchased from a retailer in Townsville. The Northern Territory sampling was conducted in May 2013 when the seafood was purchased from a retailer in Darwin. The Western Australia sampling was conducted in February 2014 when the seafood was purchased from a seafood supplier in South Fremantle, who sourced their seafood from the Pilbara or Shark Bay area (northern Western Australia).

In May 2014, the laboratory analysed four samples of seafood, caught in the southern waters of Western Australia by a recreational fisherman. These samples represented only one group, the pelagic predators. The results for these samples have been included as information only and serve as a comparison against the samples obtained from northern waters.

Seafood was obtained either fresh or frozen. Since the study focussed on the edible portion of the species the samples were obtained as fillets and, wherever possible (in the case of crustaceans), peeled or shelled. The minimum mass required per sample was 2 kg of the edible portion. This enabled low limits of detection in analysis and ensured a greater probability that the sample contained more than one specimen from the catch. The samples were sent by airfreight from the supplier to ARPANSA in Melbourne. The shipments were collected on the same day of arrival and frozen until analysis commenced.

2.4 Sample Preparation

The frozen samples were thawed and diced or blended into smaller pieces. Approximately 2 kg of each sample was distributed evenly in foil trays. The sample trays were then placed in an oven at 90°C and dried to a constant mass. The dry weight was recorded to determine the wet to dry mass ratios. The dry samples were then homogenised by grinding to a fine powder using a knifemill.

2.5 ^{134}Cs and ^{137}Cs Analyses

Depending on the mass available for analysis, the dry samples were placed into 200 mL or 450 mL Marinelli beakers. These beakers, cylindrical in shape and incorporating an inverted cylindrical well, improve detection of gamma radiation by allowing them to fit over a larger surface area of the detector. The samples were then analysed, to determine the activity concentrations of ^{134}Cs and ^{137}Cs, by high resolution gamma-ray spectrometry with Genie 2000 software. High purity germanium detectors were used, designed to achieve a low minimum detectable activity concentration (MDC) for the sample geometries selected. The calibration sources for the sample geometries, traceable to NIST (National Institute of Standards and Technology), are multi-radionuclide standards that define the efficiency curve for all energies including ^{134}Cs and ^{137}Cs. Counting times of up to 100 hours were used to further improve the MDC.

2.6 ^{210}Po Analysis

^{210}Po activity concentrations were determined by high resolution alpha spectroscopy following radiochemical separation. The dry samples were weighed (5 - 20 g) into beakers and each was spiked with a known activity of ^{208}Po tracer solution obtained from QSA Amersham International, or a ^{209}Po tracer solution obtained from Eckert & Ziegler. The samples were then dissolved in 100 mL concentrated HNO_3 and refluxed overnight on a stirring hotplate. The solution was then allowed to cool and any excess solidified fats were skimmed from the surface. The solutions were evaporated until approximately 20 mL remained. Another 100 mL of concentrated HNO_3 was added and the reflux procedure was repeated. After cooling any remaining organic matter in the solutions was destroyed using a 1:1 mixture of concentrated HNO_3 and 30% H_2O_2. The solutions were evaporated until 10 mL of acid solution remained. This was diluted to approximately 250 mL with deionised water.

The solutions were adjusted to a pH of 9 using ammonia hydroxide solution. Solutions of $MnCl_2$ and $KMnO_4$ were added to co-precipitate the polonium with manganese dioxide. The precipitate was separated by centrifuging and dissolved in a solution of hydroxylamine hydrochloride and hydrochloric acid. The polonium in the resulting solution was auto deposited onto silver discs for approximately 1.5 hours. The discs were counted by alpha spectrometry for 24-72 hours depending on activity concentration.

For each batch of samples a tracer blank and a ^{210}Po in-house reference standard were incorporated into the analysis as quality control. Some samples were randomly analysed in duplicate and some matrix spikes were analysed where a duplicate sample was spiked before with a known amount of ^{210}Po to determine the ^{210}Po recoveries.

2.7 Calculation of Committed Effective Dose

The radionuclides present in seafood enter the human body via the ingestion pathway. The committed effective dose was calculated taking into account the varying sensitivities of organs and tissues in the body, the type of radiation emitted, and the biological half-life of the particular radionuclide (the time it takes for half of the radionuclide to be excreted).

The committed effective dose to the public from consumption of seafood was estimated for ^{137}Cs and ^{210}Po using equation (1):

Committed Effective Dose (mSv/year) = D*R*C*1000 (1)

Where: D is the age-related dose coefficient for ingestion (Sv/Bq) (ICRP, 2012).
R is the rate of consumption (kg/year).
C is the average activity concentration in fish or shellfish (Bq/kg).

For ^{137}Cs and ^{210}Po, the minimum detectable activity concentration was applied as the activity concentration for samples where 'no detection' was reported. Due to its short half-life, ^{134}Cs from atmospheric nuclear weapons testing or the Chernobyl accident in 1986 would have decayed to levels unable to be detected. Therefore the contribution to overall dose from ingestion was assumed to be insignificant. As ^{134}Cs was not detected in any of the samples it has been excluded from the dose calculations. The dose coefficients used for the calculation are listed in Table 2.1.

Table 2.1: Age dependant ingestion dose coefficients. Sourced from ICRP (2012).

Age Group	Dose Coefficient (Sv/Bq)	
	^{137}Cs	^{210}Po
Adult	1.3E-08	1.2E-06
Child (10 years)	1.0E-08	2.6E-06

The most recent available consumption data was from the 1995 National Nutrition Survey (McLennan and Podger, 1995). This quoted the following consumption rates for adults and 10 year old children respectively:

- fin fish (excluding canned fish) at 6.4 g/day (2.3 kg/year) and 2.1 g/day (0.8 kg/year),
- crustacean and mollusc at 2.7 g/day (1 kg/year) and 1.1 g/day (0.4 kg/year),
- total seafood (including canned) at 25.7 g/day (9.4 kg/year) and 13.7 g/day (5 kg/year).

The consumption data of Australians will vary depending on geographical location, sex and nationality of the survey participants. For this study the calculation of committed effective dose was based on an approximate consumption value of 10 kg/year for adults and 5 kg/year for a child.

3. RESULTS

3.1 Activity Concentrations

The measured activity concentrations of ^{134}Cs, ^{137}Cs and ^{210}Po for the seafood samples are shown in Tables 3.1 – 3.4. All activity concentrations are reported on a wet weight basis. The activity concentration values for seafood from southern Australian waters (Table 3.4) are presented for comparison with the northern sample set.

The mean ^{137}Cs activity concentration in those samples where ^{137}Cs was detected was 126 mBq/kg and 87 mBq/kg for fish and crustaceans respectively. ^{137}Cs was not detected in any of the mollusc samples analysed. ^{134}Cs was not detected in any of the seafood samples analysed. ^{210}Po was detected in every sample except for the barramundi sourced from Queensland. The mean ^{210}Po activity concentration was 1280 mBq/kg and 3560 mBq/kg for fish and crustaceans respectively. The activity concentrations of ^{210}Po were generally higher than those determined for ^{137}Cs. For some groups of seafood the activity concentration of ^{210}Po was more than an order of magnitude higher than the activity concentration of ^{137}Cs.

Table 3.1: Activity concentrations of ^{134}Cs, ^{137}Cs and ^{210}Po in seafood purchased from Queensland.

Seafood Type	Species	Activity Concentration (mBq/kg Fresh Weight)		
		^{134}Cs	^{137}Cs	^{210}Po
Crustaceans	**Spanner Crab** *Ranina ranina*	<33	<28	1220 ± 210
	Tiger Prawn *Penaeus sp.*	<36	<23	2120 ± 360
	Morten Bay Bug *Thenus sp.*	<33	<43	2680 ± 480
Pelagic Predators	**Spanish Mackerel** *Scomberomorus commerson*	<60	279 ± 60	7300 ± 1100
Estuarine Predators	**Barramundi** *Lates calcarifer*	<33	<28	<20
Reef Predators	**Saddletail Snapper** *Lutjanus malabaricus*	<42	112 ± 32	229 ± 60
	Coral Trout *Plectropomus sp.*	<47	195 ± 40	264 ± 70
Bivalves	**Scallops** *Pecten fumatus*	<35	<28	3020 ± 470
Cephalopods	**Squid** *Loligo sp.*	<35	<35	3140 ± 590

Table 3.2: Activity concentrations of ^{134}Cs, ^{137}Cs and ^{210}Po in seafood purchased from the Northern Territory.

Seafood Type	Species	Activity Concentration (mBq/kg Fresh Weight)		
		^{134}Cs	^{137}Cs	^{210}Po
Crustaceans	**Mud Crab** *Scylla serrata*	<17	12.7 ± 4.4	4930 ± 600
	Tiger Prawn *Penaeus sp.*	<43	73 ± 16	1390 ± 190
Pelagic Predators	**Spanish Mackerel** *Scomberomorus commerson*	<37	273 ± 18	3170 ± 390
Estuarine Predators	**Barramundi** *Lates calcarifer*	<34	56 ± 12	311 ± 56
	Mangrove Jack *Lutjanus argentimaculatus*	<32	85 ± 14	198 ± 42
	King Threadfin Salmon *Polydactylus macrochir*	<28	100 ± 15	178 ± 37
	Black Jewfish *Protonibea diacanthus*	<27	87 ± 13	361 ± 63
Reef Predators	**Saddletail Snapper** *Lutjanus malabaricus*	<39	118 ± 19	106 ± 29
Foraging Fish	**Sand Whiting** *Sillago sp.*	<51	42 ± 13	2600± 340
Bivalves	**Pearl Oyster** *Pteriidae sp.*	<37	<42	442 ± 89
Cephalopods	**Squid** *Loligo sp.*	<21	<14	6280 ± 720

Table 3.3: Activity concentrations of ^{134}Cs, ^{137}Cs and ^{210}Po in seafood purchased from northern Western Australia.

Seafood Type	Species	Activity Concentration (mBq/kg Fresh Weight)		
		^{134}Cs	^{137}Cs	^{210}Po
Crustaceans	**Morten Bay Bug** *Thenus sp.*	<42	176 ± 96	9040 ± 930
Pelagic Predators	**Cobia** *Rachycentron canadum*	<25	128 ± 17	984 ± 68
Estuarine Predators	**Barramundi** *Lates calcarifer*	<42	75 ± 16	69 ± 16
Reef Predators	**Saddletail Snapper** *Lutjanus malabaricus*	<29	87 ± 14	812 ± 70
Cephalopods	**Cuttlefish** *Sepiidae sp.*	<32	<33	1174 ± 79

Table 3.4: Activity concentrations of ^{134}Cs, ^{137}Cs and ^{210}Po in seafood obtained from southern Western Australia.

Seafood Type	Species	Activity Concentration (mBq/kg Fresh Weight)		
		^{134}Cs	^{137}Cs	^{210}Po
Pelagic Predators	**Australian Salmon** *Arripis trutta*	<42	246 ± 32	10900 ± 1300
	Bonito *Sarda australis*	<52	271 ± 35	8270 ± 970
	Skipjack Tuna *Katsuwonus pelamis*	<43	167 ± 24	9700 ± 1100
	Samson Fish *Seriola hippos*	<40	387 ± 44	1440 ± 200

Some species were sampled across multiple regions. Spanish mackerel was obtained from Queensland and the Northern Territory. Saddletail snapper was obtained from all three regions. This allowed for a comparison of activity concentrations across regions as shown in Figures 3.1 and 3.2.

The comparison of ^{137}Cs in seafood from each region shows that there was no difference within the experimental uncertainty in the activity concentrations between identical species however the ^{210}Po results did show differences in activity concentrations between sampling areas, with Northern Territory samples showing the lowest activity concentrations.

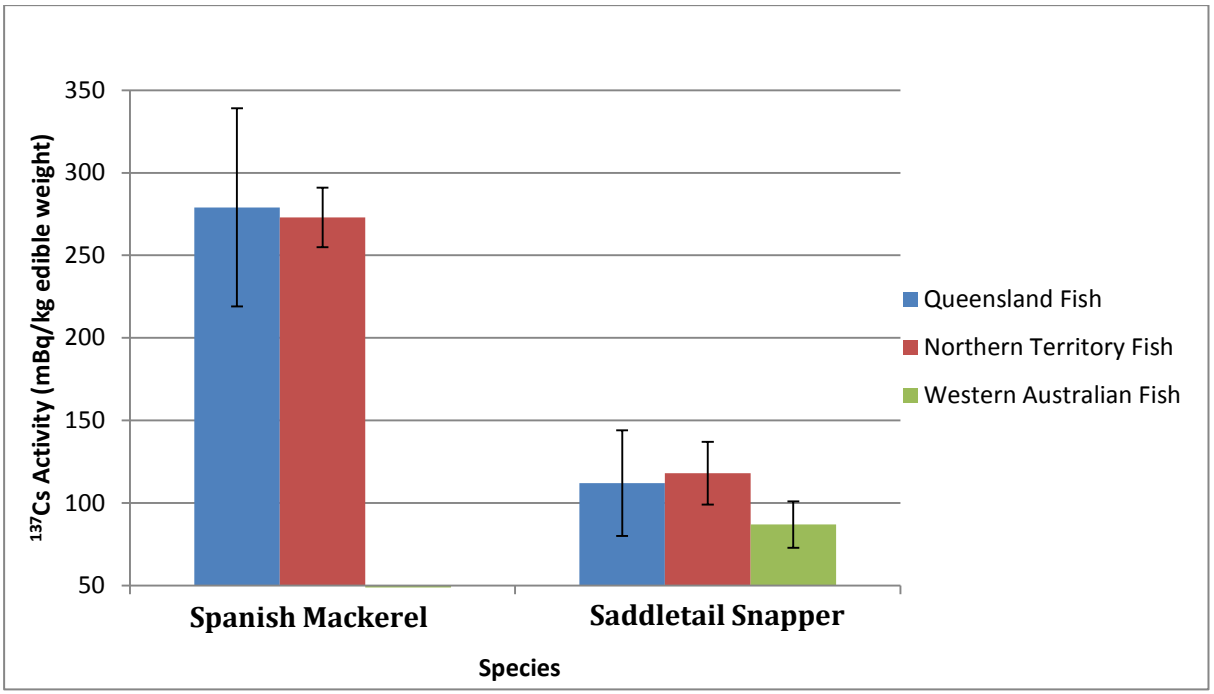

Figure 3.1: Variation in ^{137}Cs activity concentrations in the same species from different geographical locations.

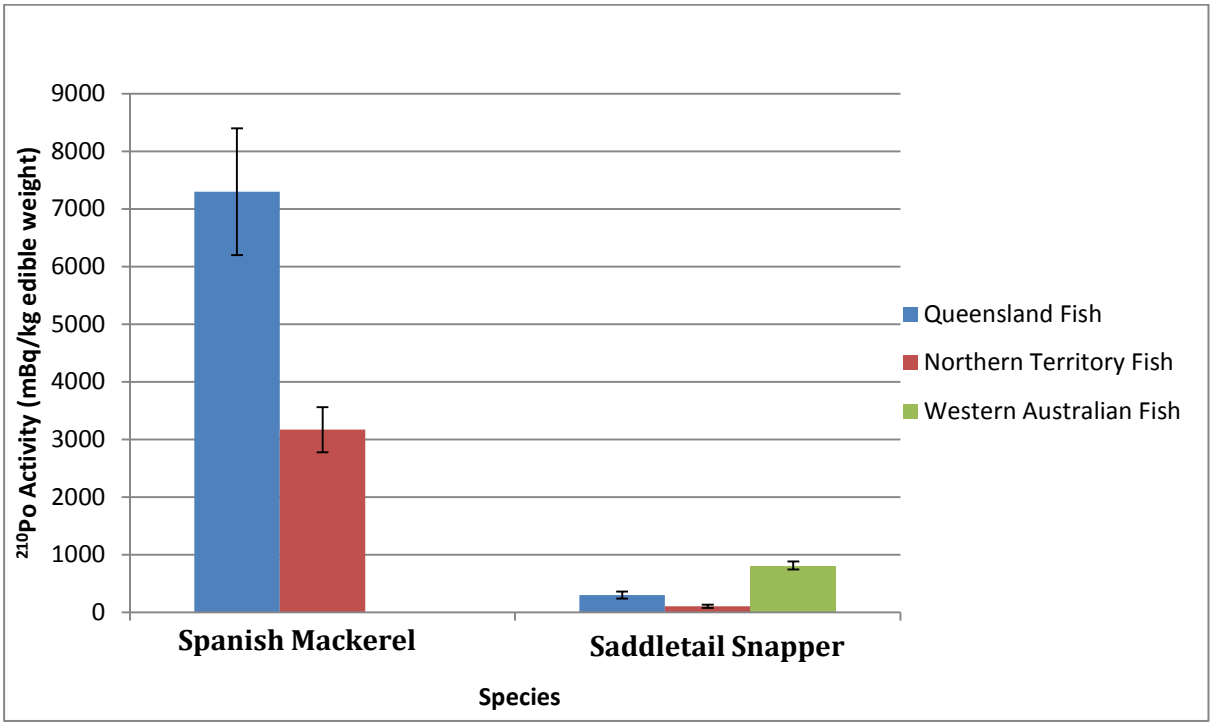

Figure 3.2: Variation in ^{210}Po activity concentrations in the same species from different geographical locations.

3.2 Dose to the Public from Ingestion of Seafood

The committed effective dose (mSv/year) has been calculated for the consumption of:

- migratory fish (pelagic predators)
- local fish (estuarine predators, reef predators and foraging fish)
- crustaceans and molluscs (bivalves and cephalopods).

Table 3.5 shows the estimate of annual committed effective dose (mSv/year) for the three different seafood categories and for all samples (based on the average activity concentration and consumption of all the seafood types).

The contribution to the ingestion dose from ^{210}Po is significantly higher than that for ^{137}Cs for both adults and children. This is due to the higher ^{210}Po activity concentrations, the type of ionising radiation being emitted (alpha) and the behavior of radionuclides in the body.

The ingestion doses for children from ^{210}Po are higher than for adults since children have higher dose coefficients. This is despite the consumption rate for a child being half that of an adult. In contrast the doses to children from the beta and gamma emissions from the decay of ^{137}Cs in seafood are lower than that for adults. This calculation is dominated by the consumption rates, as the ingestion dose coefficients for a child and an adult are similar.

Table 3.5: Committed effective doses from consumption of seafood in the average Australian diet based on ingestion of ^{137}Cs and ^{210}Po.

Seafood Type	Age Group	Committed Effective Dose (mSv/year)	
		^{137}Cs	^{210}Po
Migratory Fish	**Adult**	2.95E-05	2.72E-03
	Child	1.13E-05	4.96E-02
Local Fish	**Adult**	1.16E-05	5.62E-03
	Child	4.48E-06	6.08E-03
Crustacean/Mollusc	**Adult**	6.00E-06	3.87E-02
	Child	2.31E-06	4.19E-02
All samples	**Adult**	1.13E-05	2.50E-02
	Child	4.35E-06	2.71E-02

4. DISCUSSION

The activity concentration of ^{137}Cs in marine organisms in this report were compared to the Fisheries and Agriculture Organisation (FAO) fishing areas regional data published by the IAEA in 1995 (MARDOS). Consistent with the approach in the IAEA MARDOS report, the results were compared across three categories; fish, molluscs and crustaceans. Table 4.1 shows the 1995 MARDOS data, with the MARDOS data decay corrected to the 1st June 2013 compared to the results from this study.

Table 4.1: Mean ^{137}Cs activity concentration results compared with those documented in the 1995 MARDOS report.

Seafood Type	Mean ^{137}Cs activity concentration (mBq/kg)		
	MARDOS (1995)	MARDOS Decay corrected to 2013	Experimental Results from this study
FAO Area 57:			
Fish	140	92	97
Molluscs	140	92	Not detected
Crustaceans	90	59	176
FAO Area 71:			
Fish	300	196	130
Molluscs	140	92	Not detected
Crustaceans	110	72	43

The results obtained in this current study are generally consistent with the published data for the two areas. Values for molluscs could not be compared because ^{137}Cs was not detected in any of the samples within this category. In this study only one crustacean sample was analysed for FAO Area 57. Observed differences may be due to several reasons.

- Distribution of radionuclides in the marine environment is not uniform. Distribution is far more complex as sediments and biota scavenge and retain different radionuclides at different rates. Therefore the activity concentrations will vary spatially and temporally.
- The IAEA sample set was larger and included results from specimens caught in a much larger area covering both the northern and southern hemispheres as well as several major ocean currents as shown in Figure 2.1.
- The ages of the specimens examined and the food sources they exploit play a significant part in the accumulation of radionuclides in their tissues.

The agreement of the ^{137}Cs activity concentrations with the results reported in the 1995 MARDOS IAEA report supports the likelihood that the ^{137}Cs detected in this study, is due to fallout from historical releases of ^{137}Cs into the environment.

The Queensland samples were collected about 18 months after the Fukushima Dai-ichi NPP accident. Any ^{134}Cs released to the ocean following the accident would be at levels approximately two thirds that of ^{137}Cs. For the Northern Territory sample period, about 24 months after the accident, any ^{134}Cs from the accident would be about half that of ^{137}Cs. The samples from northern Western Australia were obtained almost three years after the accident; therefore any ^{134}Cs from the accident would be at levels approximately one third of ^{137}Cs. The ^{134}Cs activity concentration results can be used as an indicator as to whether contaminated water has reached Australian waters. This current study did not detect any ^{134}Cs in the samples collected.

In general, higher activity concentrations were observed in large pelagic predators, crustaceans and molluscs. However, though the general trend is maintained when comparing similar species in the three regions, there was more variability in ^{210}Po activity concentrations than in ^{137}Cs activity concentrations. These differences may be attributed to differing ages of the specimens caught, different food sources, and the non-uniform distribution of ^{210}Po in the environment (Sirelkhatim et al., 2008).

4.1 Health Impacts

The results from the study were used to determine a committed effective dose to children and adults for both ^{137}Cs and ^{210}Po from the ingestion of seafood. The dose to children from ^{137}Cs is less than half that for adults. For both age groups, the dose estimates from the consumption of seafood containing ^{137}Cs based on this study are very small. For adults the calculated dose from ^{137}Cs was 1.1E-05 mSv/year, over 100,000 times lower than the average background dose of 1.5 mSv/year to the Australian public from natural sources of radiation (Webb at al., 1999).

The ^{210}Po dose contribution was similar for adults (2.5E-02 mSv/year) and children (2.7E-02 mSv/year) regardless of the consumption rate. For both age groups, the dose is over 50 times lower than the average background dose to the Australian public from natural sources of radiation (Webb at al., 1999). This represents less than 2 % of the overall average background dose.

Overall, based on the assumed consumption value of 10 kg/year for adults and 5 kg/year for a child, the radiation dose from the ingestion of seafood containing the amounts of ^{210}Po and ^{137}Cs measured in this study have no health implications.

5. CONCLUSION

Activity concentrations for ^{134}Cs, ^{137}Cs and ^{210}Po were determined in all of the seafood samples obtained from northern Australian oceans. When compared to data published by the IAEA (MARDOS) report, it was found that ^{137}Cs activity concentrations, although slightly lower than expected, were consistent with the levels of anthropogenic background radioactivity due to global fallout, mainly from historical nuclear weapons testing. The absence of any measurable ^{134}Cs activity in all of the seafood samples analysed indicates that the anthropogenic activity detected is not from the Fukushima Dai-ichi NPP accident, which is consistent with the modelling predictions for contamination transport by Nakano and Povinec (2012). The results of the analysis of samples for ^{137}Cs from southern waters of Western Australia are consistent with the results obtained for the same group of seafood (pelagic predators) collected from the other regions. This further supports the conclusion that releases resulting from the Fukushima Dai-ichi NPP accident have not impacted Australian seafood.

This study has enabled the establishment of baseline information for anthropogenic radionuclides found in Australian marine wildlife that are commonly consumed in Australia. This baseline provides useful data for comparison in the case of radiological accidents such as the Fukushima Dai-ichi NPP accident, where radioactive material has the potential to reach Australian coastlines and be detected in marine organisms.

The results from the ^{210}Po activity concentrations detected provide a useful means of comparing the annual dose contributions from this naturally occurring radionuclide to the anthropogenic radionuclides of concern. It was found that the dose from ^{210}Po was many orders of magnitude higher than that from ^{137}Cs.

Based on a daily consumption of the seafood over a year (10 kg/year for an adult and 5 kg/year for a child), the committed effective dose from ^{210}Po ingestion would contribute less than 2 % of the average background dose to the Australian public from natural sources of radiation (1.5 mSv/year). The contribution from ^{137}Cs from ingestion of seafood was considered negligible, less than 0.001% of the average background dose to the Australian public. At the levels of ^{210}Po and ^{137}Cs measured in this study, there are no health implications from the ingestion of seafood.

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