ANSTO/E-764

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Radioecological Risk Analysis of ANSTO's Monthly Effluent Releases, 2006-07

by

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Sewage plume off Potter Point Photo: Tom Loosz

Prepared within the Institute for Environmental Research
Australian Nuclear Science and Technology Organisation

January 2008

ISBN: 1921268034

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1. Introduction

In line with its ISO 14000 initiatives, ANSTO endeavours to identify its environmental impacts. Part of this process includes the assessment of its releases of radioactivity into the wider environment. Liquid effluent with low levels of radioactivity is routinely released via Sydney Water's sewerage system, treated to tertiary standard in the Cronulla Sewage Treatment Plant (CSTP) and finally discharged into the marine environment at Potter Point (Figure 1). These releases occur approximately four times a week and, given the delay and mixing that occurs in transit before discharge into the ocean, can be considered to be a relatively constant release.

This raises the question as to whether the releases have the potential to cause unintended environmental impacts in the receiving environment. A comprehensive overview of the potential adverse effects of enhanced environmental radioactivity is given in Copplestone et al. (2001). In this study, the possible radiological dose-rates to biota arising from the radioactivity in the ANSTO effluent have been evaluated and compared to international criteria of acceptability to assess the potential ramifications of release.

2. Methodology

The activity concentrations of tritium (^{3}H, as tritiated water), ^{60}Co, ^{131}I and ^{137}Cs, reported in ANSTO's 2006/2007 Annual Report (Hoffmann et al. 2007, Tables 3 & 4), were used to evaluate potential environmental dose-rates to a range of organisms that may exist in the coastal environment of the discharge site at Potter Point. The other gamma-emitting isotopes detected; ^{51}Cr, ^{144}Ce and ^{226}Ra; were not included in this assessment because environmental dose conversion factors are not available for these radionuclides in the model we have applied. However, their exclusion should not substantially influence the overall conclusions reached as these radionuclides occur infrequently in the effluent. Also, ^{226}Ra is a member of the U-series and as such is an ubiquitous radionuclide in the environmental background. The ramifications of their exclusion from the analysis are discussed further below.

ANSTO effluent mixes with and is diluted by general sewage before it reaches the CSTP. Tertiary treatment, introduced in July 2001, has significantly increased the residence time and recirculation of effluent within the CSTP, resulting in a high degree of dispersion of ANSTO effluent plumes and a significant reduction in the peak tritium concentration and variability in the tertiary-treated effluent. A seven-day study of effluent dilution at Cronulla STP in 2003-04 (Hoffmann et al. 2004) found that the transit time of effluent from ANSTO to Cronulla STP was fairly constant at 5-6 hours. Under average flow conditions the detention time in the plant was approximately 22 hours, with 89% of this time in the secondary and tertiary stages of the plant. The study showed only gradual variation in the tritium concentration in the

final effluent stream, whereas prior to plant upgrade distinct pulses of tritium could be detected and measured offshore.

Tritiated water is a good, conservative tracer for dilution in these systems as it behaves identically to the non-radioactive water. Estimates of the dilution of peak tritium activity concentrations between ANSTO and the CSTP have been made since 2004 (mean ± standard error = 276 ± 377; median ± interquartile range = 179 ± 222; n = 43. However, the long retention time and high level of recirculation of effluent within the CSTP means that consecutive ANSTO releases cannot be clearly distinguished for accurate calculation of dilution. For this study we have therefore calculated monthly average dilution based on monthly flow data from the CSTP (J Smith, CSTP, pers comm.) and monthly ANSTO release volumes (Hoffman et al 2007). The estimates across the whole period are as follows: mean ± standard error = 246.9 ± 49.7; median ± interquartile range = 248.7 ± 52.3; n = 12.

Additional dispersion of the ANSTO effluent peak is known to occur before the effluent is finally released into the sea at Potter Point, at which time the effluent is mixed with seawater and further diluted to an extent that is dependant upon the local currents and wave activity. Previous monitoring of tritium at Potter Point and numerical modelling of the effluent plume conducted by AWACS/WRL in 1995 and 1997 (AWACS, 1995; WRL, 1997) have shown that the freshwater effluent plume remains in the near-surface zone until it disperses at some distance from the outfall. The offshore dilution ratio has been estimated by comparing tritium concentrations at the CSTP outlet with that at the nearest off-shore sampling point (usually between 5 and 50m from the outfall). The estimates range from about 2 to 64, depending on sea conditions (mean ± standard error = 17.3 ± 18.1; median ± interquartile range = 14.4 ± 15.1; n = 10). The median offshore dilution of 14.4 represents a reasonable and conservative estimate of dilution to the near-shore environment for a median distance of 10 m from the outfall. The combined offshore and CSTP dilution values for each month, as used in this report, are given in Table 1.

It has also been assumed that the radioactivity remains in the dissolved phase, ignoring the probability that all of the radionuclides will bind to some degree with organic and inorganic particulates within the sewerage system and hence be less available for exposure to organisms in the receiving environment. Again, this is a precautionary measure to promote protection of the environment through this assessment by overestimating real dose-rates. Table 1 also contains the diluted radioactivity concentration values used for the dose-rate assessments.

Based on activity concentrations of specific radionuclides in the environment, the spreadsheets supplied with the UK Environment Agency publication 'Impact Assessment of Ionising Radiation on Wildlife' (Copplestone et al. 2001) use concentration factors (CFs, the ratio of the radioactivity in the organism over that in the water) to calculate both external and internal dose-rates to a range of ellipsoidal reference organisms. The model incorporates weighting factors for habitat occupancy (proportion of time spent in contact with water or proximity to sediment) and the primary radiation type (whether it is alpha, beta or gamma radiation, as this will affect penetration through tissues and the linear energy transfer (LET) of the radiation dose to the tissue along the track of the radiated particle). The CFs used in these spreadsheets for a range of organisms and isotopes were amended according to UK-EA (2003) and the revised values were applied to our analyses. Default radiation weighting factors for the radionuclides and habitat occupancy were used. Average, weighted dose-rates for each month (external plus internal in μGy hr^{-1}) for thirteen

groups of marine biota (*i.e.* bacteria, phytoplankton, zooplankton, macrophytes, fish eggs, benthic molluscs, small and large benthic crustacea, pelagic and benthic fish, seabirds, seals and whales) were calculated using 'Radiological Impact Assessment for Coastal Aquatic Ecosystems' (RIA model, version 1.15; Copplestone et al. 2001). These were calculated using the mean activity concentrations of all radionuclides for each month.

Ecological risk assessment is generally applied using a tiered approach (ANZECC/ARMCANZ 2002). The first tier comprises a comparison of the estimated or measured environmental exposures with generic regulatory or guideline criteria of acceptable exposure levels. Table 2, amended from Copplestone et al. (2001) and with Estimated No-Effect Values (ENEV) from Bird et al. (2003) and Garnier-LaPlace et al. (2006), gives guideline exposure criteria for radiological dose-rate assessment in aquatic ecosystems, both marine and freshwaters. The most restrictive aquatic criteria apply to freshwaters, an ENEV of 10 μGy hr^{-1}. This value was presented to the 2002 Symposium on the Protection of the Environment from Ionising Radiation (IAEA 2003a). It is the only criterion that has been developed using the species-sensitivity distribution approach (Posthuma et al. 2001). This is the now-favoured methodology in the biological risk assessment community for setting acceptability criteria (e.g. ANZECC/ARMCANZ 2000). The approach uses biological responses to radiation exposure across a range of different species to set effects levels for communities (e.g. dose-rates that protect 95% of species). Hence the freshwater value has been set using best-practice methodology. There is no good reason to assume that freshwater organisms are any more or less radiosensitive that marine species once the dose has been delivered (i.e. after environmental factors such as geochemistry and bioavailability that affect exposure have been taken into account). Hence, assuming that the criterion is realistically representative of adverse biological effects arising from radiological exposure, as well as being the most restrictive value and hence the most precautionary, then it is also the best criterion against which to make the assessment from a risk perspective. However, it should also be noted that the freshwater criterion assumes a chronic (i.e. long-term) exposure whereas the individual estimates in this report are from month to month.

3. Results & Discussion

In this section, the radioisotopes contributing most to the average annual and maximum monthly radiological dose-rate estimates over the assessment period will be discussed first. Significant factors that mitigate against the dose-rate estimates are then also considered. Those factors will cover exposure scenarios, bioavailability and geochemical conditions that tend to reduce the worst-case estimates applied in this assessment. The implications of omitting some of the minor radionuclides from the dose-rate assessments are also discussed. The estimated annual average and monthly dose-rates are then compared with international guidelines of acceptability and the background dose-rates that are continuously present in the receiving environment. Finally, conclusions are made of the significance of the estimated dose-rates arising from ANSTO's effluent to the biota in the receiving environment at Potter Point.

Mean annual dose-rate estimates – major contributions

Radionuclide activity concentrations for the radionuclides in liquid effluent from July 06 to June 07 were divided by calculated monthly dilution factors given in Table 1 and then averaged over the entire period before being entered into the 'Radiological Impact Assessment for Coastal Aquatic Ecosystems' (Version 1.15)

software to estimate the mean annual radiological exposure (dose-rate, µGy hr^{-1}) to various categories of organism as shown in Figure 2. From the figure it is apparent that the dose-rates derived by the various organisms are low and vary substantially but that, generally, most of the estimated radiological dose can be attributed to ^{137}Cs or ^{60}Co. Caesium is accumulated as an analogue of the monovalent cation, K^+, which remains soluble in marine systems and deposited mostly within muscle mass. Cobalt is an essential trace element for phytoplankton and higher organisms, acquired from bacterial vitamin B_{12} synthesis, and hence enters the food web at a low trophic level (Santschi 1988).

Monthly dose-rate estimates & mitigating factors

The estimated monthly radiological dose-rates to a range of different organisms are given in Table 3. The maximum combined dose-rate (*i.e.* the sum of internal and external exposures from the monthly average of radionuclide concentrations for all four radionuclides) was estimated to be 0.01 µGy hr^{-1} to the whale. There are no experimental studies assessing the impact of radiological dose on aquatic mammals (Copplestone et al. 2001 citing Woodhead 1998). The RIA model predictions show that this maximum was primarily due to internal dose from ^{60}Co in June 2007. In this study we have assumed that there is no loss of radioactivity due to adsorption on particles during transit through the sewerage system. Cobalt is known to be particle reactive and has a recommended sediment water partitioning coefficient (K_d) in coastal marine sediments of 3×10^5 (IAEA, 2004). As such, most of the ^{60}Co will not have been released to the near shore environment, having been retained on particles removed at the CSTP. The remainder of the particle-associated material that was released will have been non-bioavailable and the evaluated internal dose-rates will be high overestimates. Confirmation of the low exposures from ^{60}Co is also provided by Hoffmann et al. (2007, Tables 37 – 39). Their data show the most recent measured concentrations of various radionuclides in biota collected at Potter Point. Cobalt-60 was not detected in any of those samples.

Significance of missing dose-estimates

The non-availability of dose-conversion models for ^{51}Cr, ^{144}Ce and ^{226}Ra is a shortcoming in this analysis but there are a number of factors that make the omission of relatively low consequence. Firstly, as noted earlier, these radionuclides were detected only infrequently in the effluent released to the sewage system. Secondly, whilst the recommended CFs for chromium and cerium are similar to cobalt across the range of organisms for which values exist (ranging from 10^3 to 10^4), they are slightly less than that of cobalt for all types of organisms except phytoplankton (IAEA 2004). Radium is at least an order of magnitude less with CFs ranging from 10^2 to 10^3 for the same organisms. However, this advantage is negated by the fact that, as an alpha particle emitter, ^{226}Ra has a radiation weighting factor of 20 following incorporation into tissues, bringing it back into line with the more strongly accumulated radionuclides. Thirdly, the energy and relative abundance of gamma emissions from the omitted isotopes (^{51}Cr: 320 keV @ 9.8%; ^{144}Ce: 133 keV @ 10%; ^{226}Ra: 186 keV @ 3.2%) are also less than that for ^{60}Co (two emissions at 1170 & 1332 keV both at approximately 100% abundance). Lastly, the half-lives of ^{51}Cr (27 days) and ^{144}Ce (286 days) are also less than that of ^{60}Co (1925 days) which means that they will decay more quickly in transit and after entering the environment and hence will have less time to deliver any dose to the biota. All these factors contribute to reducing the dose-rate estimate from the omitted radioisotopes compared to ^{60}Co

via either internal exposure (due to reduced bioaccumulation) and external exposure (due to reduced half-life) and via both pathways from LET within tissues (due to reduced gamma energy and abundance).

Exposure scenarios

It should also be noted that the estimated dose-rates assume chronic exposure under low dilution at the release point. However exposure will vary for different organisms because of occupancy and habitat factors. Whales and other large mammals are transitory in this area of the coast, and typically travel at some distance from the release point. Similarly, birds and other mobile and non-territorial biota will not remain permanently, nor usually for extended periods, within the mixing zone. Benthic organisms may have a reduced exposure to low K_d radionuclides because of surface trapping of the plume close to the outlet, but an increased exposure to particle reactive radionuclides such as ^{131}I or ^{60}Co due to settling out of particulates in the effluent stream as it mixes with the seawater.

Internal dose estimates within the RIA model (Copplestone et al. 2001) assume equilibrium concentration factors. The dimensions and bulk of mammals (particularly whales) and the biokinetics of metal accumulation are such that uptake equilibrium will not be reached, or even approached, within the likely exposure periods for these transitory species.

Hence, greatly increased oceanic dilution together with the lower exposure durations and non-equilibrium bioaccumulation (relevant for internal exposures) will inevitably lead to dose-rate estimates for these organisms being considerably reduced under more realistic exposure scenarios.

Comparison with international guideline values

Notwithstanding these various factors giving rise to substantial over-estimations of dose-rate in this analysis, the maximum estimate of 0.01 $\mu Gy\ hr^{-1}$ in one month is still much less than the most restrictive criterion of 10 $\mu Gy\ hr^{-1}$ (for freshwater organisms) based on chronic exposures and more than four orders of magnitude less than the IAEA (1992) recommendation of 1000 $\mu Gy\ hr^{-1}$ (for deep sea organisms) as listed in Table 2. It should also be noted that most of the estimated dose-rates were much less than the maximum. In standard environmental risk assessment procedures, if the scenario setting is precautionary (i.e. makes assumptions of what is not known well, such as the exposure of whales in this case, that tend to overestimate the risk) and the estimates come out to be less than any *a priori* concern level, then protection of the ecosystem is taken to be more assured.

Comparison with background radiological dose-rates

The dose-rates estimated here are in addition to those derived from the background radioactivity in the receiving environment and medical radioisotopes also present in the sewerage system. The background will comprise residues of man-made radioactivity, such as fallout from the atmospheric nuclear weapons tests of the last century, as well as natural radioactivity of primordial and cosmogenic origins. Fallout residues will include ^{3}H and ^{137}Cs, due to their long half-lives, but ^{131}I and ^{60}Co will have decayed to insignificance. ASPAMARD (Duran et al. 2004) gives a ^{137}Cs estimate for Pacific seawater at latitudes between 30-35º S ranging from 2.6-3.5 $Bq\ m^{-3}$. An average value for ^{3}H in seawater of approximately 1000 $Bq\ m^{-3}$ has been reported by Lua et al. (2007). Radium in the range of 1-2 $Bq\ m^{-3}$ has been measured for coastal waters (Godoy et al. 2006). The annual average values in Table 1 (last

column) and the maximum ^{226}Ra concentration given in Hoffmann et al. (2007, Table 4, 6.3 Bq L^{-1}), when diluted using the August 06 estimate (1.6 Bq m^{-3}), are all approximately equivalent to the values listed within an order of magnitude. That is, the estimated dose-rate from the effluent in the mixing zone at the end of pipe is similar to the typical background for these elements.

Medical radioisotopes in Sydney's sewerage system probably contribute the most significant amount of radioactivity in the nearfield zone. Davis (2006) collected twenty 24-hour composite effluent samples from four sewage treatment plants in Sydney (not including the CSTP). These samples had ^{131}I activities ranging from <MDA to 150 ± 11 Bq/L (mean ± standard error = 24.2 ± 47.5; median ± interquartile range = 6.85 ± 10.8 Bq/L; n = 16). Iodine-131 measured at Cronulla STP during April 2007 was consistently at the lower end of the scale seen in the Sydney area (mean ± standard error = 1.09 ± 0.33; median ± interquartile range = 1.05 ± 0.275 Bq/L; n = 8). This consistency is likely to be due to the lack of major nuclear medicine facilities in the Cronulla STP catchment.

However, the natural background overwhelms the background from anthropogenic sources. For example, naturally occurring U- and Th-series isotopes, particularly ^{210}Po, contribute significantly. In a major international study of radiological doses to humans from the consumption of marine foodstuffs, undertaken in the early 1990s and which included samples from the east coast of Australia (MARDOS; IAEA 1995), it was shown that the contribution from ^{137}Cs was only about 2% of that from ^{210}Po. Further, the radiological dose to biota in marine systems is dominated by natural ^{40}K. This strong gamma-emitter (1460 keV; $T_{1/2}$ $1.3x10^9$ y) has a typical surface seawater concentration of 12,000 Bq m^{-3} (Pentreath 1988) which is orders of magnitude greater than any of the other isotopes of interest in this report. Hence, the incremental dose to biota living in the vicinity of the outlet at Potter Point from ANSTO's effluent is negligible in comparison with the natural background to which they are continuously exposed.

4. Conclusions

Despite assuming realistic dilution factors, ignoring surface partitioning that would reduce exposure and making unrealistic assumptions about the continuous presence of transitory species, the estimates of radiological dose-rates to marine biota were, at all times, much less than even the most conservative of the internationally recognised criteria recommended for the protection of biota from radiological hazards. On this basis, the effluent released by ANSTO during the 2006-07 monitoring period can be considered to be of negligible radiological risk to biota in the receiving environment at Potter Point.

The major contributors to radiological dose-rate estimates were ^{137}Cs and ^{60}Co, although it is likely that the calculated contribution from ^{60}Co is a substantial over-estimate. Possibilities for further reducing the release of these radionuclides should be kept under review, taking into account the desirability of on-going improvement and the ALARA principle.

Acknowledgements

R. Barton, R. Cameron, N. Creighton, E. Hoffmann, P. Holden, S. McIntosh, D. Waters and D. Woods are thanked for their constructive criticisms of earlier drafts.

References

Aarkrog A, Baxter MS, Bettencourt AO, Bojanowski R, Bologa A, Charmasson S, Cunha I, Delfanti R, Duran E, Holm E, Jeffree R, Livingston HD, Mahapanyawong S, Nies H, Osvath I, Pingyu Li, Povinec PP, Sanchez A, Smith JN, Swift D, 1997. A comparison of doses from ^{137}Cs and ^{210}Po in marine food: A major international study. Journal of Environmental Radioactivity, 34 (1) 69-90.

ANZECC/ARMCANZ, 2002. Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council, and the Agriculture and resource Management Council of Australia and New Zealand, Canberra, Australia.

Australian Water and Coastal Studies, 1995. Potter Point Outfall Modelling 11 October 1995. AWACS Report 95/46. December 1995.

Bird GA, Thompson PA, Macdonald CR and Shepherd SC, 2003. Assessment of the impact of radionuclide releases from Canadian nuclear facilities on non-human biota. In Proceedings of the third international symposium on the protection of the environment from ionising radiation. Darwin, Australia, July 2002 (unedited papers, IAEA-CSP-17). International Atomic Energy Agency, Vienna, Austria. 241-247.

Copplestone D, Bielby S, Jones SR, Patton D, Daniel P and Gize I, 2001. Impact assessment of ionising radiation on wildlife, R&D Publication 128 (updated July 2002), Environment Agency, Bristol, UK. 222 pp.

Davis E, 2006. The Impact of Radionuclide Discharge from Hospitals Upon Wastewater Operations. UNSW School of Chemical Engineering and Industrial Chemistry Honours Thesis, 136 pp.

Duran E, Povinec PP, Fowler SW, Airey PL, Hong GH, 2004. ^{137}Cs and $^{239+240}$Pu levels in the Asia-Pacific regional seas. Journal of Environmental Radioactivity 76 (2004) 139–160.

Garnier-Laplace J, Della-Vedova C, Gilbin R, Copplestone D, Hingston J and Ciffroy P, 2006. First derivation of predicted-no-effect values for freshwater and terrestrial ecosystems exposed to radioactive substances. Environmental Science and Technology (40) 6498-6505.

Godoy JM, de Carvalho ZL, Fernandes FdC, Danelon OM, Godoy MLDP, Ferreira ACM, Roldão LF, 2006. ^{228}Ra and ^{226}Ra in coastal seawater samples from the Ubatuba Region - Brazilian southeastern coastal region. Journal of the Brazilian Chemical Society, 17 (4) 730-736.

Hoffmann EL, Ferris JM, Harrison JJ, Loosz T, 2004. Environmental and effluent monitoring at ANSTO sites, 2003-2004. ANSTO Report E-755, Australian Nuclear Science and Technology Organisation, Sydney, Australia.

Hoffmann EL, Ferris JM, Loosz T, 2007. Environmental and effluent monitoring at ANSTO sites, 2006-2007. ANSTO Report E-762, Australian Nuclear Science and Technology Organisation, Sydney, Australia.

IAEA, 1988. Assessing the impact of deep sea disposal of low level radioactive waste on living resources. International Atomic Energy Agency, Technical Report Series No 288.

IAEA, 1992. Effects of ionising radiation on plants and animals at levels implied by current radiation protection standards. Technical Report Series No 332, International Atomic Energy Agency, Vienna.

IAEA, 1995. Sources of radioactivity in the marine environment and their relative contributions to overall dose assessment from marine radioactivity (MARDOS). IAEA TECDOC No 838, International Atomic Energy Agency, Vienna.

IAEA, 2003. Protection of the environment from ionising radiation. Proceedings of the third international symposium on the protection of the environment from ionising radiation. Darwin, Australia, July 2002 (unedited papers, IAEA-CSP-17). International Atomic Energy Agency, Vienna, Austria. 432 pp.

IAEA, 2004. Sediment Distribution Coefficients and Concentration Factors for Biota in the Marine Environment. Technical Reports Series No. 422. International Atomic Energy Agency, Vienna, Austria. 95 pp.

Lua J-G, Huang Y-J, Wang L-H, Li F, Li S-Z, Hsia Y-F, 2007. ^{3}H and ^{90}Sr background in water around Tianwan NPP, China. Radiation Measurements 42 (2007) 74 – 79.

NCRP, 1991. Effects of ionising radiation on aquatic organisms. National Council on Radiation Protection and Measurement Report No. 109. 115pp.

Pentreath RJ, 1998. Radionuclides in the aquatic environment. Ch 9 in 'Radionuclides in the Food Chain', JH Harley, GD Schmidt, G Silini. ILSI monographs, MW Carter, Editor-in Chief. Springer-Verlag, Berlin.

Posthuma L, Suter GW, Traas TP, 2001. Species Sensitivity Distributions in Ecotoxicology. Lewis Publishers, Boca Raton

Santschi PH, 1998. Factors controlling the biogeochemical cycles of trace elements in fresh and coastal marine waters as revealed by artificial radioisotopes. Limnology and Oceanography 33 (4; 2) 848-866

Thompson PM, 1999. Assessment programme to fulfil the commitments of the Nuclear Safety Commission – Proceedings of the Second International Symposium on Ionising Radiation and the Environment, Ottawa, May 1999.

UK-EA, 2003. Habitats regulations for Stage 3 assessments: radioactive substances authorisations. R&D Technical Report P3-101/SP1a. Environment Agency, Bristol. 100pp.

Water Research Laboratory, 1997. Potter Point Outfall Modelling 27 May 1997. WRL Technical Report 97/17. June 1997.

Woodhead DS, 1998. The Impact of Radioactive Discharges on Native British Wildlife and the Implications for Environmental Protection. R&D Technical Report P135, Environment Agency, Bristol, UK, 80pp.

Table 1 Average monthly activity concentrations (Bq m^{-3}) in ANSTO effluent 2006-07, diluted according to measured flow rates from ANSTO and the CSTP, as indicated. These data were used as input to the RIA model (Copplestone et al., 2001). Blanks indicate that the average activity concentrations were less than the minimum detection level.

	Jul-06	Aug-06	Sep-06	Oct-06	Nov-06	Dec-06	Jan-07	Feb-07	Mar-07	Apr-07	May-07	Jun-07	Annual average
Dilution factor	5268	3839	3436	3514	3635	3076	2628	4135	2780	3675	2847	3747	3548
^{3}H	4.75E+02	4.79E+02	8.47E+02	1.16E+03	9.13E+02	1.21E+03	2.32E+03	7.40E+03	8.49E+03	1.57E+03	2.11E+03	1.33E+03	2.36E+03
^{60}Co												7.47E-02	6.23E-03
^{131}I		1.54E-01	1.14E-01		9.08E-02	3.80E-01	9.36E-01	2.18E-01	1.33E-01	1.50E-01		1.95E-01	1.97E-01
^{137}Cs		3.57E+00	4.92E+00		5.50E-01	7.25E-01	1.04E+00	1.29E+00	2.28E+00	3.36E+00	6.50E+00	4.16E+00	2.37E+00

Table 2 Guideline dose-rate limit criteria (µGy hr^{-1}) to biota.

Units µGy hr^{-1}	**NCRP (1991)**	**IAEA (1988, 1992)**	**Canada (Thompson 1999)**	**US DOE**	**Canada (Bird *et al*. 2003)**	**France – IRSN (Garnier-Laplace *et al.* 2006)**
Freshwater organisms	400	400		400		10
Benthic invertebrates			100		200	
Fish			50		20	
Deep ocean organisms		1000				

Table 3 Estimated monthly and maximum radiological dose-rates (μGy hr^{-1}) to a range of marine biota arising from diluted ANSTO effluent in 2006-07. Values derived using the input data in Table 1 and the RIA model (Copplestone et al., 2001).

	Bacteria	Phytoplankton	Zooplankton	Macrophyte	Fish egg	Benthic mollusc	Small b. crust.	Large b. crust.	Pelagic fish	Benthic fish	Seabird	Seal	Whale
Jul-06	4.70E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06	4.67E-06
Aug-06	2.00E-03	6.50E-06	5.70E-05	1.60E-03	9.90E-04	7.40E-04	8.10E-04	7.30E-04	1.30E-04	6.40E-04	2.40E-03	7.60E-04	3.30E-04
Sep-06	2.80E-03	1.10E-05	5.50E-05	2.30E-03	1.30E-03	1.00E-03	1.10E-03	1.00E-03	1.80E-04	8.80E-04	3.30E-03	1.00E-03	3.90E-04
Oct-06	1.10E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05	1.14E-05
Nov-06	3.20E-04	9.30E-06	3.50E-05	2.70E-04	1.80E-04	1.20E-04	1.30E-04	1.20E-04	2.90E-05	1.10E-04	4.10E-04	1.60E-04	1.10E-04
Dec-06	4.20E-04	1.20E-05	1.10E-04	3.70E-04	3.00E-04	1.60E-04	1.80E-04	1.60E-04	3.80E-05	1.40E-04	6.50E-04	3.70E-04	3.20E-04
Jan-07	6.10E-04	2.40E-05	2.70E-04	5.80E-04	5.20E-04	2.40E-04	2.60E-04	2.40E-04	6.10E-05	2.10E-04	1.10E-03	7.60E-04	7.40E-04
Feb-07	8.00E-04	7.40E-05	1.30E-04	6.80E-04	4.70E-04	3.40E-04	3.60E-04	3.40E-04	1.20E-04	3.00E-04	1.00E-03	4.40E-04	3.00E-04
Mar-07	1.40E-03	8.47E-05	1.26E-04	1.13E-03	7.18E-04	5.52E-04	5.98E-04	5.47E-04	1.64E-04	4.87E-04	1.65E-03	5.89E-04	3.15E-04
Apr-07	1.90E-03	1.70E-05	6.60E-05	1.60E-03	9.40E-04	7.10E-04	7.70E-04	7.00E-04	1.30E-04	6.10E-04	2.30E-03	7.30E-04	3.20E-04
May-07	3.70E-03	2.39E-05	4.24E-05	2.97E-03	1.74E-03	1.35E-03	1.49E-03	1.34E-03	2.51E-04	1.17E-03	4.34E-03	1.24E-03	4.14E-04
Jun-07	1.10E-02	1.50E-05	8.70E-05	1.10E-02	1.90E-03	5.20E-03	5.30E-03	5.10E-03	5.70E-04	4.30E-03	8.70E-03	9.50E-03	1.30E-02
Maximum	1.10E-02	8.47E-05	2.70E-04	1.10E-02	1.90E-03	5.20E-03	5.30E-03	5.10E-03	5.70E-04	4.30E-03	8.70E-03	9.50E-03	1.30E-02

Figure 1 Location map showing the Sydney Coast, ANSTO at the Lucas Heights Science and Technology Centre and the final liquid effluent discharge site at Potter Point

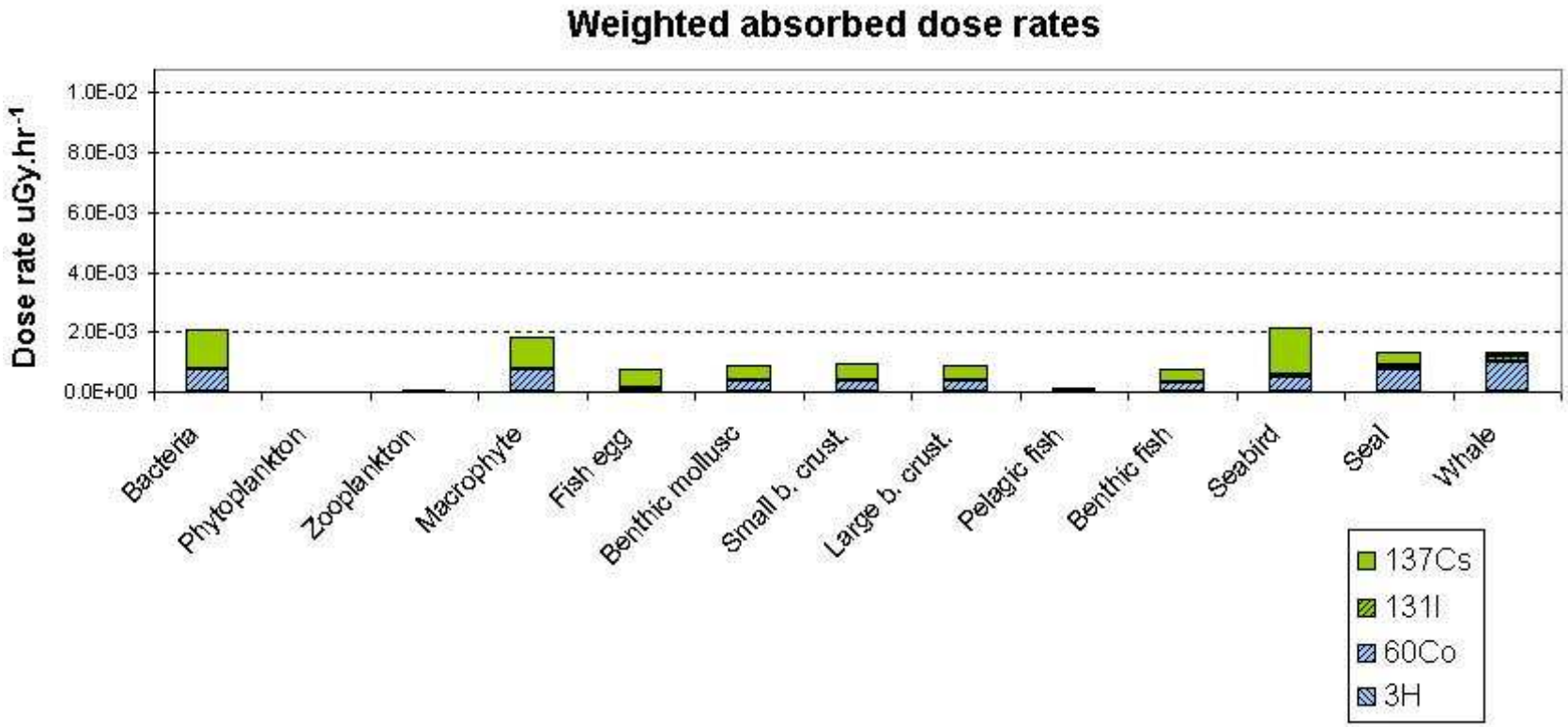

Figure 2 Cumulative bar charts of dose-rate (μGy hr^{-1}) estimated for thirteen types of marine biota exposed to the averaged radionuclide concentrations over the entire 2006-07 monitoring period.