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Dose assessment for marine biota and humans from discharge of ^{131}I to the marine environment and uptake by algae in Sydney, Australia

Jessica Veliscek Carolan*, Catherine E. Hughes, Emmy L. Hoffmann

Australian Nuclear Science and Technology Organisation, PMB 1, Menai, NSW 2234, Australia

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ABSTRACT

Iodine-131 reaches the marine environment through its excretion to the sewer by nuclear medicine patients followed by discharge through coastal and deepwater outfalls. ^{131}I has been detected in macroalgae, which bio-accumulate iodine, growing near the coastal outfall of Cronulla sewage treatment plant (STP) since 1995. During this study, ^{131}I levels in liquid effluent and sludge from three Sydney STPs as well as in macroalgae (*Ulva* sp. and *Ecklonia radiata*) growing near their shoreline outfalls were measured. Concentration factors of 176 for *Ulva* sp. and 526 for *E. radiata* were derived. Radiation dose rates to marine biota from ^{131}I discharged to coastal waters calculated using the ERICA dose assessment tool were below the ERICA screening level of 10 μGy/hr. Radiation dose rates to humans from immersion in seawater or consumption of *Ulva* sp. containing ^{131}I were three and two orders of magnitude below the IAEA screening level of 10 μSv/year, respectively.

1. Introduction

In Australia and globally, radioisotopes are routinely discharged to the environment following their uses in medicine, veterinary medicine, research and industry. Within nuclear medicine, which is an important industry in terms of its discharge of radioactivity to the marine environment, there are a wide range of radioisotopes that are used to diagnose and treat a variety of disorders. In general radioisotopes are chosen for use in medicine for their ease of detection, low radiotoxicity, short half-lives and appropriate chemical or biological behaviour (Titley et al., 2000). These radioisotopes will decay while in the body, but some of their activity will also be discharged to sewer as the patient excretes the radioisotope. The major radiopharmaceuticals used in the Sydney area include ^{51}Cr, ^{18}F, ^{125}I, ^{131}I, ^{89}Sr, ^{99m}Tc and ^{90}Y.

Iodine-131 is considered by the International Commission on Radiological Protection (ICRP) to be the most critical medical radionuclide in terms of the potential radiological dose it provides to medical staff, the public and their relatives following procedures involving therapeutic administration of unsealed radionuclides (ICRP, 2004). Iodine-131 is critical in part because dosages of ^{131}I used for treatment of thyroid cancers are typically the largest dosages administered in nuclear medicine (OSPAR Commission, 2002). It is also the most commonly used therapeutic radioisotope in Sydney (Davis, 2006). With a half-life (8.04 days) longer than the time it takes to be excreted from the body, transported to and treated at a sewage treatment plant (STP), ^{131}I is often still present when it reaches the marine environment and because the gamma radiation it emits is relatively high energy (364 keV) it is easy to detect and measure.

Iodine-131 is used for both diagnostic and therapeutic procedures. Diagnostic dosages range from 0.2 to 400 MBq and treatment dosages range from 50 MBq to 10 GBq, with a typical ablation dosage being approximately 7 GBq (ARPANSA, 2007; Davis, 2006; Titley et al., 2000). Not all of the ^{131}I administered to patients is excreted as some radioactive decay occurs while it is in the body. The percentage of, and rate at which, ^{131}I is excreted from the body is dependent upon the nature of the treatment. The levels of ^{131}I excreted decrease over time after treatment.

In Sydney, all STPs discharge their treated effluent to the marine environment either directly through coastal or deep ocean outfalls, or indirectly via outfalls into rivers. Some STPs also recycle some of their treated effluent as water for industrial, residential or municipal reuse (ARPANSA, 2007). Release of effluent containing radionuclides to the marine environment provides a potential route by which the public can be exposed to radionuclides. An investigation into the sources and fate of radioactive discharges to public sewers in the UK by Titley et al. (2000) found that the discharge of effluent

* Corresponding author. Tel.: +61 2 9717 7251.
E-mail address: jessica.veliscek-carolan@ansto.gov.au (J. Veliscek Carolan).

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containing radionuclides to the marine environment provides a higher radiological dose to members of the public than sludge containing radionuclides being re-used on agricultural land or disposed to landfill.

Marine algae are well known for their ability to concentrate iodine from seawater. One way to characterise the extent to which iodine is concentrated above ambient (seawater) concentrations is through use of a concentration factor (CF), which is the ratio of iodine in the algae over the concentration in the surrounding seawater. The recommended CF for iodine in macroalgae is 10 000 L/kg fresh weight (IAEA, 2004). Measuring the ^{131}I content of marine algae is a good way of monitoring ^{131}I levels in the local environment as it is concentrated to above ambient levels and continuously integrates changes in exposure. However, the use of CFs assumes the system in question to be at equilibrium and, since ^{131}I levels in algae are a net value from uptake, loss and radioactive decay of ^{131}I, this is not always the case. In addition, CFs for marine algae are affected by exposure time and algal condition. Thus, it can be difficult to infer ^{131}I concentrations in the environment surrounding algae by measuring its ^{131}I content (Druehl et al., 1988).

In this study we sought to gain an understanding of the relative contributions of direct sewage discharges from the Australian Nuclear Science and Technology Organisation (“ANSTO”) and discharges from nuclear medicine facilities and outpatients, to the levels of ^{131}I found in effluent from Cronulla STP. Also, in order to determine whether there is any radiological risk to marine biota or humans from the current levels of ^{131}I discharged to the coastal marine environment in Sydney, we measured the levels of ^{131}I in effluent from three STPs in the Sydney region (including Cronulla STP) and in algae from their corresponding shoreline outfalls. Using these data, we performed an environmental dose assessment for marine biota living near the sewage outfalls, as well as a human dose assessment for people swimming or eating algae from the sewage outfalls.

2. Study area

The population of Sydney is approximately 4.3 million. Sales of ^{131}I to nuclear medicine facilities in the Sydney region are approximately 2500 GBq per year. On average, approximately 70% of this activity would be expected to be excreted based on biokinetic studies on people undergoing various ^{131}I medical procedures (Barrington et al., 1996; Hilditch et al., 1991; ICRP, 2004), contributing about 1750 GBq of ^{131}I to the sewer system each year. During 2007–2008, 427 GL of wastewater was discharged to the marine environment through ocean outfalls and approximately 42 600 dry tonnes of biosolids were produced, corresponding to a biosolids removal rate of about 10%. However, these statistics are dominated by the three large high rate primary treatment STPs with offshore deep ocean outfalls, not the smaller secondary and tertiary treatment STPs focussed on in this study (Sydney Water, 2008b).

There are many hospitals and clinics in Sydney that provide nuclear medicine services. Of these, there are 20 hospitals within Sydney Water’s area of operations that administer therapeutic nuclear medicine (Davis, 2006). Therapeutic procedures often involve administration of large activities. Therefore, some nuclear medicine patients are required to remain in hospital for up to one week after treatment in order to prevent exposure of family members, caregivers or members of the public to any sizeable radiological dose (ICRP, 2004). The majority of the activity excreted by these patients therefore travels to the receiving STP of the hospital, although some is also excreted into alternate sewerage systems after the patient returns home. Sydney Water’s current trade waste policy requires all nuclear medicine units to have holding tanks, in which the radioactive effluent from patients is stored and allowed to decay before it is discharged to the sewer. The required detention time of at least 16 days waste must be discharged to sewer at a controlled rate to achieve appropriate dilution.

Of the Sydney hospitals that provide nuclear medicine, 17 have their effluent treated by one of the three major Sydney STPs (Bondi, Malabar, North Head), all of which provide high flow rate primary treatment and have deepwater ocean outfalls between 2 and 4 km offshore (Sydney Water, 2008a). As this paper is investigating levels of ^{131}I in inter-tidal algae growing near shoreline ocean outfalls, these STPs are not discussed further here. Sutherland Hospital and a private nuclear medicine clinic (Southern Radiology) are the only providers of ^{131}I treatments within the Cronulla STP catchment. Neither of these nuclear medicine facilities provide inpatient dosages, therefore the activity they administer will not necessarily be received by Cronulla STP.

ANSTO discharges its treated liquid effluent to Cronulla STP and the Environmental Monitoring group at ANSTO has measured ^{131}I levels in algae from its ocean outfall at Potter Point since 1995 (Hoffmann et al., 1996, 1997, 1998, 1999, 2000, 2001; Hoffmann and Loosz, 2002; Hoffmann and Ferris, 2002; Hoffmann et al., 2003, 2004, 2005, 2006, 2008, see Section 5.3.1). For this reason, Cronulla STP and Potter Point ocean outfall were chosen as one of the study sites for this investigation. Two other STPs with shoreline outfalls of comparable size to Cronulla STP were also chosen for study and are shown in Table 1. Cronulla STP is the largest of the three STPs investigated and provides tertiary treatment, whereas STP A and STP B provide secondary treatment.

The location of ANSTO and its receiving STP at Cronulla are shown in Fig. 1.

3. Materials and methods

Treated sewage effluent and sludge from Cronulla STP, STP A and STP B, raw effluent from ANSTO and algae from Potter Point, outfall A and outfall B were collected and analysed for ^{131}I.

3.1. Sampling

3.1.1. Effluent and sludge

Iodine-131 treatments are administered irregularly and at most nuclear medicine clinics, infrequently. Hence, levels of ^{131}I in STP effluent vary widely. It was therefore necessary to collect samples over a period of time and calculate an average value to establish a representative ^{131}I concentration. The advantage of sludge sampling is that its longer residence time means a single sample provides a more long-term representation of ^{131}I levels in that particular STP. In addition, it can accumulate ^{131}I and thus demonstrate the presence of low levels of ^{131}I passing through an STP when they are too low to be detected in the effluent.

All effluent samples were collected in 1 L HDPE bottles from the final treated effluent stream so that levels of ^{131}I in the samples reflected what was discharged to the marine environment. Sludge samples were collected either in plastic or glass jars. Gloves were worn at all times when collecting and processing each sample type.

Six 1 L grab samples of liquid effluent from ANSTO holding tanks (HT, non-active trade waste) and three 1 L samples of liquid effluent decanted from ANSTO mixing tanks (MT, low level waste waters) were collected between 31/3/2008 and 17/4/2008.

From Cronulla STP, fourteen 1 L 24 h composite effluent samples, sampled hourly from 09:00 to 08:00, and two digested sludge samples were collected between 3/4/2008 and 17/4/2008. Eleven 1 L 24 h composite effluent samples, sampled hourly from 09:00 to 08:00 from Cronulla STP were also collected between

Table 1
Characteristics of Cronulla STP, STP A and STP B.

STP	Cronulla	A	B
Treatment level	Tertiary	Secondary	Secondary
Discharge volume (ML/day)	54	36	16
Discharge location	Potter Point (shoreline)	Outfall A (shoreline)	Outfall B (shoreline)
Effective population	200 000	140 000	59 000

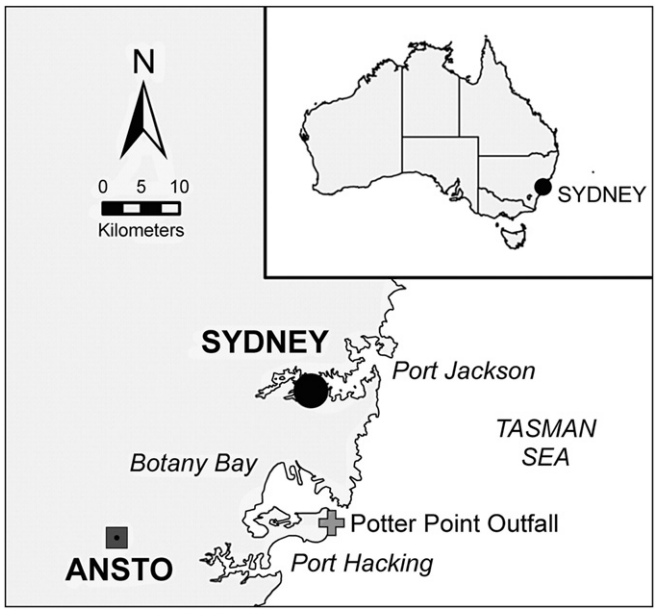

Fig. 1. Location of Cronulla STPs' outfall at Potter Point in relation to Sydney and ANSTO.

19/4/2007 and 29/4/2007. From STP A, nine 1 L 24 h composite effluent samples, sampled hourly from 08:00 to 07:00, and one liquid digested sludge sample were collected between 10/6/2008 and 19/6/2008. From STP B, ten 1 L 24 h composite effluent samples, sampled hourly from 00:00 to 23:00, and one digested sludge (biosolids) sample were collected between 23/6/2008 and 2/7/2008.

3.1.2. Algae

Ulva sp. (sea lettuce) is a common green macroalgae or seaweed that grows on inter-tidal rock platforms. *Ecklonia radiata* is a common brown kelp that grows below tidal level. Algae samples were collected from rock platforms at low tide near the sewage outfalls of the relevant STP at the end of the effluent sampling periods. Fresh algae samples were stored in plastic bags for transport back to the laboratory.

The *Ulva* sp. samples collected had smooth, ~0.1 mm thick fronds approximately 10–50 mm in length and 5 mm in width that emerged from the holdfast in a roughly circular pattern. The *E. radiata* samples collected had ~0.5 mm thick fronds approximately 200–500 mm in length and 40 mm wide that grew vertically from the holdfast.

Ulva sp. (approx. 200 g fresh weight) was collected from 80 m south of the Potter Point ocean outfall at the end of effluent sampling. Approximately 2 weeks later a further two samples of *Ulva* sp. (approx. 500 g fresh weight each) from 40 and 80 m south of the outfall and one sample of the kelp *E. radiata* (approx. 1 kg fresh weight) from 100 m south of the outfall were collected from Potter Point. *Ulva* sp. from 30 m north of the outfall (approx. 200 g fresh weight) and 45 m south of the outfall (approx 100 g fresh weight) were collected from outfall A. Three samples of *Ulva* sp. (approx. 500 g fresh weight each) and three samples of *E. radiata* (approx. 1 kg fresh weight each) were collected from 10 m west, 50 m south-east and 80 m north-west of the outfall at outfall B. A seawater sample from directly above the outfall was also collected in a 1 L HDPE bottle at outfall B.

3.2. Preparation and analysis of samples

All samples were analysed for ^{131}I using gamma spectrometry and reported values are decay corrected to the time of sampling. Uncertainties are reported as one standard deviation (SD) unless indicated otherwise.

It should be noted that reported uncertainties for algae in this paper are from measurement only. Substantial uncertainty will also arise due to variability between samples, so that a single sample is not necessarily representative. Examples of variables that will affect concentrations of ^{131}I in algae include currents, tides, temperature and weather (such as storm events). Uncertainties due to sample variability are in most cases substantially greater than measurement uncertainties.

3.2.1. Effluent and seawater

The effluent or seawater (500 g) was weighed into a Marinelli beaker. The sample was then acidified to <pH 2 with nitric acid (15.8 M, 2 mL) to prevent adsorption of undissolved matter to the Marinelli beaker. The sample was then labelled and the Marinelli beaker wrapped in a plastic bag to prevent contamination of the gamma detector.

Liquid samples were counted using P-type and N-type High Purity Germanium (HPGe) coaxial detectors (EG&G, ORTEC®). Maestro (EG&G, ORTEC®) software was used to obtain the data and the data interpretation was completed using ORTEC® GammaVision®-32 Version 6 software. An efficiency calibration for the Marinelli beaker geometry was completed for each detector using a mixed gamma standard (Isotrak).

3.2.2. Sludge and algae

Algae were dried whole and unwashed in order to provide consistency and comparability with previous data collected by the Environmental Monitoring group at ANSTO. Algae are analysed whole as iodine concentration factors in algae vary depending on the part of the algae sampled (Frechou and Calmet, 2003; Marsh et al., 1988; Teas et al., 2004). It has also been shown that washing chopped algae with deionised water removes 34–44% of its iodide (Martinelango et al., 2006). Therefore, washing the algae samples prior to analysis is likely to result in underestimation of the concentration of ^{131}I in the algae and of the radiological dose to grazers.

Both the sludge and algae samples collected were dried in an oven at 50 °C for at least 20 h. Drying at high temperature is likely to cause loss of iodine through iodovolatilisation (Mairh et al., 1989). However, Druehl et al. (1988) have shown that oven drying algae at 60 °C for 24 h does not cause volatilisation of the iodine. Fresh and dry weights of the samples were recorded.

Dried sludge and algae samples were ground to a fine powder using a Rocklabs ring mill. The ring mill head was washed and dried between grinding of each sample to prevent cross-contamination. Samples were packed and sealed into 65 mm petri dishes, labelled and wrapped in a plastic bag to prevent contamination of the gamma detector. Samples were counted for 23 h on an N-type GMX High Purity Germanium (HPGe) coaxial detector (EG&G, ORTEC). Maestro (EG&G, ORTEC) software was used to obtain the data and the data interpretation was completed using an excel spreadsheet. An efficiency calibration for the 65 mm petri dish geometry was completed using a mixed gamma standard (ARPANSA).

4. Results

4.1. ANSTO, Cronulla STP and potter point results

All ANSTO waste effluent is received, processed and tested on site to ensure that it complies with the trade waste agreement with Sydney Water before release to the sewerage system. Mixing tank effluent containing moderate levels of ^{131}I is diluted with non-active effluent from holding tanks before release from ANSTO. No ^{131}I was detected in any of the ANSTO holding tank effluent samples. Levels of ^{131}I measured in the ANSTO mixing tank effluent samples that were released during the sampling periods are shown in Table 2. ANSTO effluent mixes with and is further diluted by general sewage as it travels to Cronulla STP. Thus the radioactivity levels of the effluent as it travels to and is treated at Cronulla STP are substantially lower than the levels shown in Table 2.

Iodine-131 levels in Cronulla STP effluent collected from 19/4/2007 to 29/4/2007 ranged from below the minimum detectable

Table 2
^{131}I in effluent released from ANSTO decay corrected to the typical time that effluent is discharged at Potter Point ocean outfall, 2007 and 2008 campaigns.

Sample description	Date discharged	^{131}I Activity at time of discharge (Bq/L)	Volume discharged (m^3)	^{131}I discharged (MBq)	^{131}I Activity at time of ocean release (MBq)
MT4 Decant	23/04/2007	76	70	5.3	4.8
MT4 Decant	24/04/2007	74	30	2.2	2
Total release of ^{131}I at Potter Point due to ANSTO 2007 (MBq) =					6.8
MT4 Decant	2/04/2008	73	100	7.3	6.6
MT2 Decant	10/04/2008	20	100	2	1.8
Total release of ^{131}I at Potter Point due to ANSTO 2008 (MBq) =					8.4

activity (MDA) to 1.6 Bq/L with an average of 0.90 ± 0.43 Bq/L, assuming that the ^{131}I concentration in samples below the MDA was half the MDA. The Cronulla STP flow rates over the sampling period ranged from 39 to 262 ML/day and there was a bypass of 94 ML of effluent due to high rainfall giving a total flow of 1165 ML. If we assume the concentration of ^{131}I in this bypassed effluent was the average of 0.90 Bq/L, the flow weighted average was 0.80 Bq/L and the total amount of ^{131}I released over the sampling period in 2007 was 931 MBq.

Iodine-131 levels in Cronulla STP effluent sampled from 3/4/2008 to 17/4/2008 ranged from below the MDA to 1.4 Bq/L with one sample being lost. The average ^{131}I concentration was 0.70 ± 0.31 Bq/L, assuming that the ^{131}I concentration in the sample below the MDA was equal to the half the MDA and in the lost sample was equal to the average (0.70 Bq/L). The Cronulla STP flow rates over the sampling period ranged from 49 to 89 ML/day for a total of 943 ML. The flow weighted average was 0.71 Bq/L and the total amount of ^{131}I released over the sampling period was 671 MBq. Two digested sludge samples collected on 9/4/2008 and 16/4/2008 contained 42 ± 3.4 and 45 ± 3.2 Bq ^{131}I per kg fresh weight, respectively.

Levels of ^{131}I measured in algae collected from Potter Point are shown in Table 3. One *Ulva* sp. sample was collected at the end of effluent sampling at Cronulla STP. Two *Ulva* sp. samples and one *E. radiata* sample were collected approximately two weeks later.

4.2. STP and outfall A results

No ^{131}I was detected in any of the treated effluent samples from STP A from 11/6/2008 to 19/6/2008 (all below the MDA). The liquid digested sludge sample collected at the end of effluent sampling contained 8.1 ± 0.89 Bq ^{131}I per kg fresh weight.

Levels of ^{131}I measured in algae collected from outfall A at the end of effluent sampling are shown in Table 3. Algae cover at the outfall site was very sparse so only small amounts of *Ulva* sp. could be collected. *E. radiata*, although present, was unattainable under the conditions of sampling as the waves were large and the tide too high.

4.3. STP and outfall B results

Iodine-131 levels in STP B effluent sampled from 23/6/2008 to 2/7/2008 ranged from below the MDA (for the first 4 days of sampling) to 4.7 Bq/L. There were perceptible increases on 27/6/2008 and 1/7/2008. The average ^{131}I concentration was 1.6 ± 1.5 Bq/L, assuming that the ^{131}I concentration in the first four samples (<MDA) were equal to half the MDA. Flow rates for STP B over the sampling period ranged from 15 to 19 ML/day for a total of 165 ML. The flow weighted average was 1.6 Bq/L and the total amount of ^{131}I released during the sampling period was 256 MBq. A digested sludge sample collected on 30/6/08 contained 18 ± 1.6 Bq ^{131}I per kg fresh weight.

Table 3 shows the ^{131}I content of *Ulva* sp. and *E. radiata* samples collected from outfall B at the end of effluent sampling. The *Ulva* sp. samples contained between 1.2 and 38 Bq ^{131}I per kg fresh weight. The *E. radiata* samples contained between 24 and 73 Bq ^{131}I per kg fresh weight. A seawater sample from directly above the outfall was also collected and measured to contain 0.84 ± 0.08 Bq ^{131}I per L.

Table 3
^{131}I in marine algae collected from Potter point, outfall A and outfall B.

Date sampled	Species	Distance from Outfall	^{131}I (Bq/kg FW)	% Uncertainty
Potter point				
18/04/2008	*Ulva* sp.	80 m S	6.5	8
5/05/2008	*Ulva* sp.	40 m S	11	7
5/05/2008	*Ulva* sp.	80 m S	5.8	8
5/05/2008	*E. radiata*	100 m S	20	7
Outfall A				
19/06/2008	*Ulva* sp.	30 m N	0.86	14
19/06/2008	*Ulva* sp.	45 m S	0.53	46
Outfall B				
2/07/2008	*Ulva* sp.	10 m W	38	7
2/07/2008	*Ulva* sp.	80 m NW	6.1	8
2/07/2008	*Ulva* sp.	50 m SE	1.2	27
2/07/2008	*E. radiata*	10 m W	63	7
2/07/2008	*E. radiata*	80 m NW	73	7
2/07/2008	*E. radiata*	50 m SE	24	7

5. Discussion

5.1. Direct contribution of ANSTO to ^{131}I levels in the marine environment at potter point

In order to determine ANSTO's direct contribution to the levels of ^{131}I discharged via Potter Point, the amount of ^{131}I in ANSTO effluent that would have reached Potter Point was calculated. A seven day study of effluent dilution at Cronulla STP in 2003–2004 found that under average flow conditions, liquid effluent discharged from ANSTO takes approximately 5–6 h to travel to Cronulla STP and the residence time of effluent in Cronulla STP is approximately 22 h (Hoffmann et al., 2004). Therefore, ^{131}I concentrations in the effluent at the time of release from ANSTO were decay corrected by 27 h to when they would have reached the outfall at Potter Point.

The calculations in Table 2 demonstrate that a total of 6.8 MBq ^{131}I in 2007 and 8.4 MBq ^{131}I in 2008 was released at Potter Point due to direct discharges from ANSTO. The total activity of ^{131}I discharged at Potter Point during the sampling period in 2007 was 931 MBq and in 2008 was 671 MBq. Therefore, ANSTO's contribution to the levels of ^{131}I reaching the marine environment during the 2007 sampling period was 0.7% and during the 2008 sampling period was 1.3%. From these results it is evident that ^{131}I from ANSTO effluent does not substantially contribute to the levels of ^{131}I discharged at Potter Point. Excretions from nuclear medicine patients treated with ^{131}I are the most likely source of the majority of the ^{131}I seen in Cronulla STP effluent.

5.2. Behaviour of ^{131}I in effluent and sludge

5.2.1. Cronulla STP

The ^{131}I levels measured in Cronulla STP effluent during this study are low compared to levels that have been measured in Sydney STPs that treat effluent from large hospitals; ^{131}I levels from below the MDA to 150 Bq/L with an average of 24 Bq/L were observed in effluent from Liverpool, Malabar and Bondi STPs (Davis, 2006). The lower levels of ^{131}I seen in Cronulla STP compared to Liverpool, Malabar and Bondi STPs is most likely because there are no nuclear medicine facilities that provide inpatient dosages within the Cronulla STP catchment area. However, ^{131}I is clearly and consistently present in the Cronulla STP effluent. The fact that ^{131}I appears to be more consistently present in Cronulla STP effluent than in effluent from STPs A and B may be because it is the largest of the three STPs and has the highest effective population. This makes the likelihood of at least one person within the catchment area of Cronulla STP excreting ^{131}I to the sewer system higher than at STPs A or B. In addition, at Cronulla STP the average return rate of activated sludge from the clarifiers to the bioreactor is equal to the average flow rate under normal conditions (Mark Ziogas, pers comm, 12/8/2005), resulting in a high degree of recycling (50%) and mixing of the incoming effluent stream which may result in an averaging effect over time of any incoming ^{131}I.

5.2.2. STP A

No ^{131}I was detected in any of the effluent samples from STP A. This suggests that either no ^{131}I passed through the STP during sampling or that the levels were too low to be detected. The latter scenario is more probable as ^{131}I was detected in the sludge sample from STP A, which has a residence time of approximately 10 days, suggesting that some ^{131}I did pass through the STP during the sampling period. In addition, effluent samples were counted between 2 and 8 days after collection so any small amounts of ^{131}I present may have decayed to below detectable levels before counting.

There are two nuclear medicine facilities within the catchment area of STP A that utilise ^{131}I. Data from ANSTO Radiopharmaceuticals and Industrials (ARI) show that the most recent sale of ^{131}I to either of these nuclear medicine facilities before the end of the sampling period was 600 MBq posted on 26/5/08. This is a typical hyperthyroidism dosage. According to the ICRP (2004) approximately 54% of the activity administered (320 MBq) to hyperthyroidism patients is excreted and approximately 5% of the excreted activity (16 MBq) would have been excreted during the sampling period. Flow data from STP A indicates that approximately 360 ML of effluent passed through the STP during sampling, giving an average concentration of ^{131}I in the effluent of approximately 0.045 Bq/L. This is below the MDA (~0.1 Bq/L) and may explain why no ^{131}I was detected in any of the effluent samples.

Secondary treatment, as is provided by STP A, removes approximately 10% of the ^{131}I in untreated effluent to the sludge phase (ARPANSA, 2007; Titley et al., 2000). It has been shown above that approximately 16 MBq of ^{131}I passed through these STPs in the 10 days prior to when the sludge sample was collected. Therefore, approximately 1.6 MBq of ^{131}I would have been removed to the sludge during the sampling period. Removal of suspended solids during treatment at STP A is approximately 330 mg/L (Barbara McRae, pers comm, 23/01/2009) so approximately 120 tonnes of sludge were removed during the sampling period, giving an average ^{131}I concentration in the sludge of approximately 14 Bq/kg. This is in reasonable agreement with the measured ^{131}I concentration in the sludge of 8.1 Bq/kg, considering the large number of approximations and assumptions made in this estimate. Therefore, the assumption of 10% removal efficiency for ^{131}I during secondary treatment at an STP appears to be valid.

5.2.3. STP B

The average ^{131}I concentration in effluent from STP B is higher than in Cronulla STP effluent despite the fact that no ^{131}I was measured in the STP B effluent during the first 4 days of sampling. This is most likely because Cronulla STP is larger, hence there is less effluent flow and less dilution at STP B than Cronulla STP. It is not because more ^{131}I passes through STP B. In fact, flow data from STP B and Cronulla STP and the measured concentrations of ^{131}I in their effluent show that 27 and 45 MBq ^{131}I per day passed through STP B and Cronulla STP during their respective sampling periods.

Iodine-131 levels in STP B effluent were highly variable compared to the levels in Cronulla STP effluent. This variability may be explained by the lower recirculation and mixing at STP B. At STP B, only approximately 0.5% of the final effluent stream is returned to the aeration tank (Matthew O'Sullivan, pers comm 27/2/2009) compared with 50% at Cronulla STP. In addition, as explained in Section 5.2.1, Cronulla STP's larger effective population could also explain the higher variability in ^{131}I levels in effluent from STP B.

There were sudden increases in ^{131}I levels in effluent from STP B on both 27/06/2008 and 1/07/2008. As there are no hospitals or nuclear medicine clinics within the catchment area of STP B, it is likely that two people living in the catchment area returned home after being treated with ^{131}I on these two dates. The observed activity levels in STP B are consistent with this hypothesis.

The digested sludge sample from STP B contained noticeably less ^{131}I than the sludge samples from Cronulla STP. There are three possible reasons for this; 1) the average residence time of sludge at STP B is approximately 23 days, almost twice that at Cronulla (12 days), therefore decay could account for the majority of the difference; 2) Cronulla STP provides tertiary treatment while STP B provides only secondary treatment, so the removal of suspended solids is higher at Cronulla STP. Since ^{131}I is adsorbed onto suspended solids in effluent, removal of more suspended solids corresponds to removal of more ^{131}I to the sludge phase; and 3) more ^{131}I passed through Cronulla STP than STP B during the sampling period.

The effluent sample collected on 2/7/08 contained 2.6 Bq/L ^{131}I and the seawater sample contained 0.84 Bq/L ^{131}I, giving a dilution factor of 2.9 between the STP sampling site and directly above the outfall.

5.3. Uptake of ^{131}I by algae

The levels of ^{131}I detected in algae samples collected during this study are, in general, higher than levels measured in algae from other parts of Australia and internationally as shown in Table 4. This may be because during this study samples were collected from close to sewage outfalls where ^{131}I is discharged.

The biological half-life of radioiodine in several brown alga have been reported between 5 and 23 days (Druehl et al., 1988). The biological half-life of radioiodine in *Ulva pertusa* is reported in Nozaki and Saito (1995) as varying between 1.5 and 4.1 days depending on the part of the algae frond. On this basis 131I levels in *Ulva* sp. would be predicted to exhibit a higher level of temporal variability than *E. radiata*, particularly near outfalls such as outfall B where effluent concentrations are more variable. In future studies, performing laboratory tests to determine the biological half-live(s) for ^{131}I of the algae species being investigated may be of interest.

5.3.1. Potter point

The data in Fig. 2 shows the ^{131}I content of marine algae (mean ± SD, standard deviation due to counting uncertainty) from Potter Point measured by the Environmental Monitoring group at ANSTO since 1995 to range between 1 and 375 Bq/kg. From 1995 to March 2001 *Enteromorpha intestinalis* and *Cladophora* sp. were

Table 4
Concentrations of ^{131}I in algae collected in Australia and Internationally.

Location	Species	^{131}I concentration (Bq/kg FW)	Reference
Potter Point	*Ulva* sp.	1.4–55	Hoffmann et al., 2000, Hoffmann and Loosz, 2002, Hoffmann and Ferris 2002, Hoffmann et al., 2003, 2004, 2005, 2006, 2008
	E. radiata	20	This study
Outfall A	*Ulva* sp.	0.53–0.86	This study
Outfall B	*Ulva* sp.	1.2–38	This study
	E. radiata	24–73	This study
Western Australia	*E. radiata*	0.38–3.0	Marsh et al., 1988
Hong Kong	*Ulva lactuca*	<0.31[a]	Royal Observatory Hong Kong 1995
Sweden, Germany, UK, Iceland, Canada, USA, Japan	*Fucus* sp.	<MDA – 4800[a]	Druehl et al., 1988[b]
Northeast Pacific	*Fucus* sp.	<MDA - 735[a]	Druehl et al., 1988[b]
Canada, Norway, Japan (commercially available)	*Alaria marginata*	1.1[a]	van Netten et al., 2000

[a] Calculated assuming algae to contain 82% (*Ulva.*) or 85% (*Alaria, Fucus*) water.
[b] This paper includes ^{131}I concentrations in *Fucus* sp. subsequent to Chernobyl, therefore some ^{131}I concentrations are uncharacteristically high.

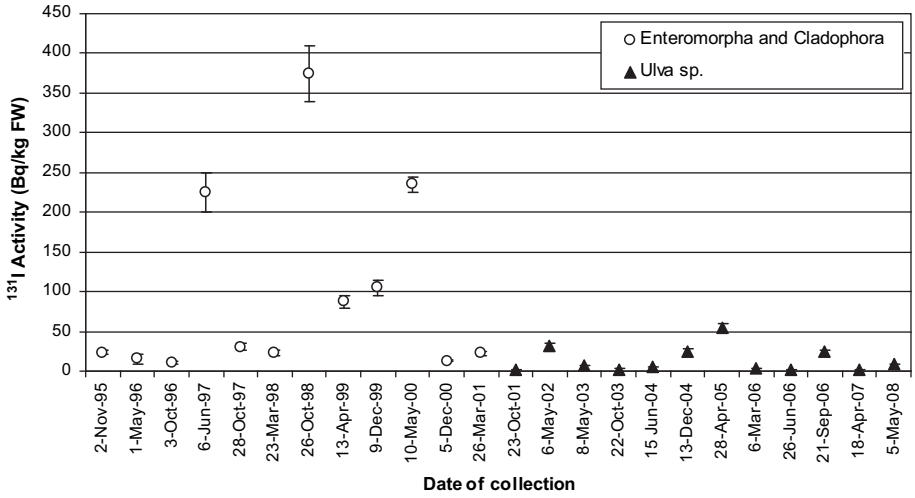

Fig. 2. ^{131}I levels (mean ± SD) in marine algae collected from Potter Point, November 1995 to May 2008.

collected. In October 2001 these species were unavailable, so the non-filamentous green algae *Ulva* sp. was collected and has been used since. In July 2001, Cronulla STP was upgraded to tertiary treatment. Iodine-131 concentrations in Cronulla STP effluent discharged would have become less variable after the upgrade and it can be seen that the ^{131}I content of the algae also became less variable. The observed reductions in the ^{131}I content of the algae after July 2001 may reflect a lower CF for *Ulva* sp. than the algae species collected previously or a reduction in the average ^{131}I concentration in the seawater.

The ^{131}I concentration in the *Ulva* sp. sample collected at the end of the effluent sampling period at Cronulla STP was 6.5 Bq/kg fresh weight, consistent with levels measured prior to this study (Fig. 2). The flow weighted average concentration of ^{131}I in the Cronulla STP effluent over the preceding two weeks was 0.71 Bq/L and the ^{131}I concentration in the seawater would have been lower than this due to dilution and dispersion of the effluent as it was discharged. Thus, it is clear that *Ulva* sp. concentrates ^{131}I from the seawater.

Two *Ulva* sp. and one *E. radiata* sample were collected from Potter Point on 5/5/08. Of the two *Ulva* sp. samples collected, the sample growing closer to the outfall contained more ^{131}I. This is to be expected as the concentration of ^{131}I in the seawater nearest the outfall will be highest (less dispersion and dilution). The *E. radiata* sample contained higher levels of ^{131}I than either of the *Ulva* sp. samples although it was growing further from the outfall. This suggests that *E. radiata* has a larger CF for iodine than *Ulva* sp. This is not surprising as in general brown algae have higher iodine CFs than green or red algae (Bowen, 1979). *E. radiata*'s higher CF means using it as an environmental monitor would allow detection of lower levels of ^{131}I. In fact, this is the main reason *E. radiata* was collected and analysed during this study in addition to *Ulva* sp. However, *E. radiata* is more difficult and hazardous to collect than *Ulva* sp. as it grows below low tidal level and can therefore only be collected at very low tide and when seas are calm.

5.3.2. Outfall A

The lack of algae cover at outfall A may have been due to the rough seas (6–7 m waves) in the days preceding sampling. Alternatively, algae growth may simply be sparse in this area. As at Cronulla STP, levels of ^{131}I were higher in the *Ulva* sp. sample from closer to the outfall. ^{131}I concentrations in *Ulva* sp. from outfall A were approximately one order of magnitude lower than at Potter Point. This is to be expected as the ^{131}I in effluent from STP A was below detectable levels and therefore must have been at least one order of magnitude lower than at Cronulla STP.

5.3.3. Outfall B

The *Ulva* sp. sample collected from closest to the outfall at site B again contained the highest concentration of ^{131}I. This *Ulva* sp. sample appeared smaller and younger than the other samples collected, which may explain the particularly high concentration of ^{131}I in this sample as it is known that juvenile algae concentrate iodine more effectively than mature specimens (Frechou and Calmet, 2003; Holmes-Farley, 2003; Teas et al., 2004). The *Ulva* sp. sample collected 80 m north-west of the outfall contained more ^{131}I than the sample collected 50 m south-east of the outfall, suggesting that dominant currents and wave action at outfall B prior to sampling were to the north-west, as was observed during sampling.

The *E. radiata* sample from closest to the outfall did not contain the highest concentration of ^{131}I. This may be because this sample was not collected down current of the outfall and was growing in quite a protected area. Alternatively, the *E. radiata* collected from this site may have been older and less effective at concentrating ^{131}I than the other samples collected. As with the *Ulva* sp., the *E. radiata* sample collected 50 m south-east of the outfall contained the lowest levels of ^{131}I, again suggesting that effluent from the outfall at site B moves predominantly north-west upon discharge. The *E.*

Table 5
Literature Iodine content and concentration factors for iodine in marine algae.

Algae species	*Ulva lactuca*	*Alaria esculenta*	*Alaria marginata*
Frechou and Calmet, 2003 (stable iodine conc, CF)	86.6, 223 (October) 43.3, 93 (March)		
Hou and Yan, 1998 (mg I/kg DW, CF[a,c], CF[b,c])	54, 225, 164		
Martinelango et al., 2006 (mg I/kg DW, CF[a,d], CF[b,d])	25, 208, 151	111, 925, 569 (annual zone) 151, 1258, 774 (perennial zone)	
van Netten et al., 2000 (mg I/kg DW, CF[a,c], CF[b,c])			151, 629, 387
Saenko et al., 1978 (mg I/kg DW, CF[a,c], CF[b,c])			1500, 6250, 3846
Teas et al., 2004 (mg I/kg DW, CF[a,c], CF[b,c])		110, 458, 282	

[a] Calculated assuming algae to contain 75% water.
[b] Calculated assuming algae to contain 82% (*Ulva* sp.) or 85% (*Alaria*) water.
[c] Calculated assuming the concentration of iodine in seawater to be 0.06 ppm (Bowen, 1979).
[d] Calculated assuming the concentration of iodide in seawater to be 30 μg/L (Martinelango et al., 2006).

Table 6
Comparison of derived and literature CFs for *Ulva sp.* and *E. radiata* growing at Potter Point and Outfall B.

Species	Location	Average [^{131}I] in effluent (Bq/L)	[^{131}I] in seawater (Bq/L)	[^{131}I] in algae (Bq/kg FW)	CF (derived)		CF (literature)
					Mean	Standard deviation	
Ulva sp.	Potter Point	0.71	0.041	7.8[a]	189	69	158
E. radiata	Potter Point	0.71	0.041	20[b]	475	n/a	1172
Ulva sp.	Outfall B	1.6	0.092	15[a]	163	216	158
E. radiata	Outfall B	1.6	0.092	53[a]	577	280	1172

[a] $n = 3$.
[b] $n = 1$.

radiata sampled from the site 50 m south-east of the outfall was also growing in quite a sheltered area and this probably contributed to the low levels of ^{131}I detected in this sample. As at Potter Point, *E. radiata* samples from outfall B consistently contained significantly higher levels of ^{131}I than *Ulva* sp. samples, suggesting they have a higher CF for iodine.

5.3.4. Derived concentration factors (CFs)

It can be seen from Table 5 that CFs for marine algae vary widely in the literature, most likely because they are influenced by many factors including the species and age of the algae as well as the season, water temperature and location of sampling. The literature CFs shown in Table 5 were averaged to give the values in Table 6, which compares the derived CFs calculated for *Ulva* sp. and *E. radiata* from data obtained during this study with literature values. A literature CF for *E. radiata* could not be found, so the average of literature CFs for the related species *Alaria esculenta* and *Alaria marginata* were used.

The derived concentration factors in Table 6 were calculated by dividing the average of the measured concentrations of ^{131}I in the algae by the concentration in the surrounding seawater. The ^{131}I concentration in the seawater was calculated by dividing the flow weighted average ^{131}I concentration in the effluent by the average near shore (between 5 and 50 m from the outfall) dilution factor of 17, considered to be a reasonable estimate of dilution to the near shore environment at Potter Point (Twining and Hughes, 2008). Near shore dilution factors between 1.8 and 64 have been estimated by dividing measured tritium in Cronulla STP effluent by tritium measured at sampling points between 5 and 50 m from the Potter Point outfall (sampling location and dilution depended on sea conditions). The dilution factor of 17 was also used for outfall B. It was assumed that dilution is similar at Potter Point and outfall B as the environment and conditions at both outfalls are similar. This is a conservative estimate of dilution for a number of reasons: firstly background was neglected and the maximum effluent concentration was used in dilution estimates, both of which would lead to an underestimate of dilution; secondly most algae samples were collected further from the outfall than the average distance of dilution data.

When possible, the values in Table 6 of the ^{131}I concentrations in the algae are the average of several measured ^{131}I concentrations of algae sampled at the end of the effluent sampling period. At Potter Point however, there was only one *Ulva* sp. sample and no *E. radiata* sample collected at the end of effluent sampling. Therefore, the ^{131}I concentrations measured in *Ulva* sp. and *E. radiata* approximately two weeks later were included. It should be noted that only one ^{131}I concentration was available for *E. radiata*.

The results in Table 6 demonstrate that both *Ulva* sp. and *E. radiata* concentrate ^{131}I to above ambient levels and that *E. radiata* has the higher CF. The derived CFs for *Ulva* sp. are consistent between Potter Point and outfall B and also agree well with the literature value. This suggests that this method of calculation of CFs is effective and reliable. There is also reasonable agreement between the two derived *E. radiata* CFs. However, the derived CFs do not agree with the average literature CF, which is an order of magnitude greater. This is most likely because the literature CF is for the related genus Alaria. The results in Table 6 suggest that the CFs of related algae species are not necessarily comparable.

Although no ^{131}I was detected in the effluent from STP A, ^{131}I was detected in the sludge from STP A and the algae from outfall A, suggesting that ^{131}I was present in the effluent below detectable levels. The average derived CF for *Ulva* sp. (176) can be used to calculate the concentration of ^{131}I in effluent from STP A in order to determine if this is the case. The *Ulva* sp. samples from outfall A contained an average of 0.70 Bq/kg fresh weight, so the concentration of ^{131}I in the seawater would have been 0.0040 Bq/L. If we assume near shore dilution of 17 then the average concentration of ^{131}I in the effluent would have been 0.069 Bq/L, which is below the MDA ($\sim$0.1 Bq/L) and may explain why no ^{131}I was detected in any of the effluent samples from STP A.

6. Dose assessment

Using measured ^{131}I levels in STP effluent and near shore dilution factors allows determination of the ^{131}I concentration in the seawater at STP ocean outfalls. The ^{131}I concentrations in the algae and the seawater can then be used to perform dose assessments for local marine biota and for people swimming in water or consuming algae at these shoreline outfalls.

6.1. Dose rates to marine biota

The environmental dose rates to marine biota due to discharge of ^{131}I to the marine environment were determined using the Tier 2 ERICA environmental dose assessment tool 1.0 (Brown et al., 2008). This model calculates internal dose rates to the biota from uptake of radionuclides, in this case ^{131}I, and external dose rates from irradiation by seawater and sediment.

Table 7
Average and maximum ^{131}I concentrations in seawater and algae from Potter Point and Outfall B.

	Average			Maximum		
	[^{131}I] in seawater (Bq/L)	[^{131}I] in *Ulva* sp. (Bq/kg FW)	[^{131}I] in *E. radiata* (Bq/kg FW)	[^{131}I] in seawater (Bq/L)	[^{131}I] in *Ulva* sp. (Bq/kg FW)	[^{131}I] in *E. radiata* (Bq/kg FW)
Potter Point	0.044	13	20	0.89	55	20
Outfall B	0.092	15	53	2.6	38	73

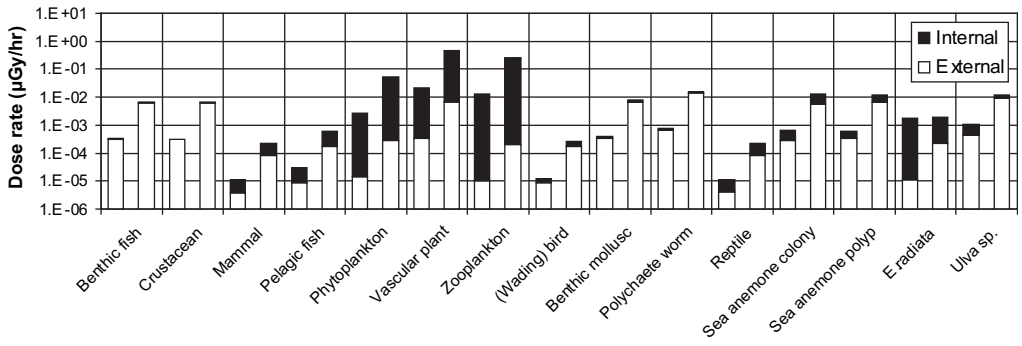

Fig. 3. Average and maximum dose rates to marine biota at Potter Point.

Performing the dose assessment required assumptions to be made. Inputs to the model include concentration factors, occupancy factors, dose conversion factors, partition coefficients and the dimensions of the biota. Default values for these inputs were used for all biota except *Ulva* sp. and *E. radiata*.

The average of the two derived CFs for *Ulva* sp. (176) and *E. radiata* (526) in Table 6 were used in this dose assessment. *Ulva* sp. was assumed to have 100% sediment surface occupancy as it grows on rock platforms close to the rock surface and is always kept wet by spray, although it spends 1–4 h per day not immersed in water. The partition coefficient of ^{131}I was assumed to be the same for rock as for coastal sediment, which is 70 (Bq/kg DW)/(Bq/L) according to the IAEA (2004). *E. radiata* is immersed in water all of the time with a holdfast connecting it to the rock surface. Therefore, irradiation from the rock surface was assumed to be negligible and 100% occupancy in the water column was assumed.

The typical dimensions of an *Ulva* sp. frond entered into the ERICA tool were 30 mm height, 7 mm length and 0.1 mm width and of an *E. radiata* frond were 400 mm height, 40 mm length and 1 mm width. Each frond was assumed to be elliptical. The densities of both algae were assumed to be 1 g/mL as algae consist mostly of water.

Dose assessments were performed for biota from Potter Point and outfall B for both an average and a worst case scenario. The average and maximum concentrations of ^{131}I measured in the algae were used for these two scenarios, respectively, and are shown in Table 7. It should be noted that at Potter Point only one *E. radiata* sample was collected so the average and maximum ^{131}I concentrations are the same. Thus the average and maximum internal dose rates calculated for *E. radiata* will also be the same. The average and maximum *Ulva* sp. ^{131}I concentrations at Potter Point in Table 7 are from data collected by Environmental Monitoring at ANSTO since 2001 (Hoffmann et al., 2000, Hoffmann and Loosz, 2002; Hoffmann and Ferris, 2002; Hoffmann et al., 2003, 2004, 2005, 2006, 2008). The flow weighted average or maximum ^{131}I concentration measured in the effluent was divided by the average (17) or minimum (1.8, Twining and Hughes, 2008) dilution factor to calculate the concentration of ^{131}I in the seawater for the average and worst case scenarios, respectively.

Figs. 3 and 4 show the ERICA calculated dose rates for various marine biota, including *E. radiata* and *Ulva* sp., from internal and external sources of ^{131}I at Potter Point and outfall B, respectively. In each figure there are two columns for each organism, which represent the dose rates in the average (first column) and worst case (second column) scenarios, respectively.

The modelled dose rates in Figs. 3 and 4 show that for all marine biota, including *Ulva* sp. and *E. radiata*, almost all dose rates were in the same order of magnitude for Potter Point and outfall B. However, dose rates for marine biota from outfall B were all slightly higher. All dose rates, including those calculated for the worst case scenario, were less than the ERICA dose rate screening value for marine biota of 10 µGy/h (Garnier-Laplace et al., 2006). This is the most restrictive dose rate guideline for aquatic ecosystems in use and, as a screening value, assumes chronic exposure, whereas the worst case exposure would be transient.

The data in Figs. 3 and 4 show that *E. radiata* has a larger internal dose rate than Ulva sp, which is due to its larger concentration factor for iodine. *Ulva* sp. on the other hand can be seen to have a larger external dose rate, due to the assumption that it is constantly exposed to sediment which has adsorbed ^{131}I particles.

The dose rate for *E. radiata* at outfall B (Fig. 4) is more representative as it comes from the average or maximum of three samples rather than one. This also makes it more comparable with the *Ulva* sp. dose rates which also come from the average or maximum of three samples.

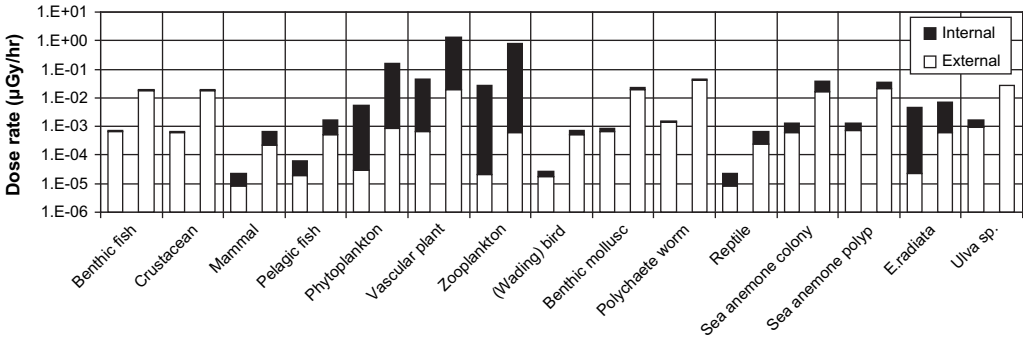

Fig. 4. Average and maximum dose rates to marine biota at Outfall B.

Table 8
Human Dose Assessment from Scenarios 1 and 2.

Scenario 1: Swimming/ Surfing	[^{131}I] in seawater (Bq/m^3)	Exposure time (s)	Annual dose from immersion (μSv)	^{131}I Activity in 170 mL seawater (Bq)	Swims/surfs per year	Annual dose from ingestion (μSv)	Total annual dose (μSv)
Potter Point	44	2 246 400	3.9E-03	7.5E-03	312	3.4E-02	3.8E-02
Outfall B	92	2 246 400	8.2E-03	1.6E-02	312	7.0E-02	7.8E-02
Scenario 2: Consumption of Ulva sp.	[^{131}I] in Ulva sp. (Bq/kg FW)	Exposure time (s)	Annual dose from immersion (μSv)	^{131}I Activity of one serve, 16 g FW (Bq)	Serves per year	Annual dose from ingestion (μSv)	Total annual dose (μSv)
Potter Point	13	n/a	nil	0.21	52	1.6E-01	1.6E-01
Outfall B	15	n/a	nil	0.24	52	1.8E-01	1.8E-01

Note: Doses calculated using dose coefficient of immersion 3.98E-17 (Sv/s)/(Bq/m^3) and committed dose equivalent per unit uptake via ingestion 1.44E-8 Sv/Bq (Shleien et al., 1998).

6.2. Critical group assessment for human exposure

Discharge of effluent containing radionuclides to the marine environment may expose members of the public to radioactivity through fishing, consumption of seafood, irradiation by contaminated seawater, exposure to seaspray or through swallowing a small amount of contaminated seawater when swimming or surfing. Consumption of seafood from Potter Point is unlikely to provide any significant radiological dose as the flesh of blackfish (*Girella* sp.) that eat *Ulva* sp. from Potter Point has been analysed biannually for ^{131}I since 1995 and detectable levels of ^{131}I have only been found once (0.81 Bq/kg fresh weight, May 2000) (Hoffmann et al., 2000). However, immersion in seawater off Potter Point or outfall B, or consumption of algae from these outfalls may provide some radiological dose. Although *E. radiata* is not an edible algae species, fresh and dried *Ulva* sp. may be used in salads, spaghetti sauce and soups. On average, recipes using *Ulva* sp. suggest a serving size of 16 g fresh *Ulva* sp. per person. *Ulva* sp. are also used in some dietary supplement pills, but are unlikely to be harvested from near sewage outfalls for this purpose. Hence, two exposure scenarios involving immersion in seawater or consumption of *Ulva* sp. were investigated.

Scenario 1 A person swims or surfs 6 days per week for 2 h in seawater off Potter Point or outfall B and so is immersed in seawater containing ^{131}I and ingests 170 mL of water per surfing session (Stone et al., 2008).

Scenario 2 Consumption of *Ulva* sp. containing ^{131}I; A person eats 1 serve of *Ulva* sp. collected from Potter Point or outfall B per week.

The average ^{131}I concentrations in seawater and in *Ulva* sp. from Table 7 were used for this human dose assessment. The data in Table 8 show the dose rates from swimming/surfing or consumption of *Ulva* sp. at Potter Point and outfall B. The total annual dose to surfers or swimmers from scenario 1 is the sum of the annual doses from irradiation and ingestion. It can be seen in Table 8 that the annual dose from external irradiation in seawater is one order of magnitude less than the annual dose from ingestion of seawater. In all cases, radiological dose rates between Potter Point and outfall B were within the same order of magnitude. The annual dose from ^{131}I due to consumption of *Ulva* sp. is one order of magnitude higher than from swimming or surfing in seawater, but is still negligibly small. All dose rates were negligible considering the background dose rate in Sydney from terrestrial and cosmic radiation is 1.5 mSv/year (ARPANSA, 2009). All dose rates were well below the individual dose limit for public exposure of 1 mSv/year and the maximum dose constraint for public exposure due to radioactive waste disposal of 0.3 mSv/year (ICRP, 2007). All dose rates were also at least 2 orders of magnitude less than 10 μSv/year, the level considered trivial by the IAEA (1988).

If a person were constantly immersed in seawater off outfall B for one year their external dose (0.12 μSv) would still be two orders of magnitude below the IAEA screening value of 10 μSv/year. However, this scenario does not take into account the internal dose that may be received from accidental swallowing of seawater. In order to receive the maximum dose constraint for public exposure due to radioactive waste disposal of 0.3 mSv/year it would be necessary to consume 31 kg *Ulva* sp. from Potter Point or 27 kg *Ulva* sp. from outfall B every week. This is a highly unlikely scenario as these volumes of *Ulva* sp. (approximately 1600 kg from Potter Point or 1400 kg from outfall B each year) are unlikely to be present at these sites.

Titley et al. (2000) calculated the dose rate from ^{131}I to a critical group of members of the public in the UK who consumed high levels of locally caught seafood, spent considerable amounts of time on the banks of the estuary where sewage effluent was discharged and were exposed to seaspray. At Beckton STP in London the average dose rate was 0.25 μSv/year and at Knostrop, Leeds was 12 μSv/year. These dose rates are comparable with or substantially higher than those calculated in this study.

7. Conclusions

Much of the ^{131}I from nuclear medicine and ANSTO effluent is eventually discharged to the marine environment. Therefore, the radiological dose rate for marine biota and members of the public from discharge of ^{131}I to the marine environment in Sydney was modelled.

It was shown that during two sampling periods in 2007 and 2008 ANSTO's direct contribution to the levels of ^{131}I discharged into the marine environment at Potter Point ocean outfall were less than 2%. This indicates that direct discharges of ^{131}I from ANSTO liquid effluent to Cronulla STP and hence to the marine environment are minimal. However, it should be noted that indirectly ANSTO also provides the ^{131}I from nuclear medicine as it provides the radiopharmaceuticals to Sydney.

The behaviour of ^{131}I in effluent was explored. It was observed that lower variability in effluent ^{131}I concentrations at Cronulla STP appeared to be associated with a high rate of recirculation and mixing during treatment. It was also observed that effluent from the largest STP, Cronulla STP, which had the highest flows, had lower ^{131}I concentrations than the smaller STP B due to increased dilution. However, the total amount of ^{131}I passing through Cronulla STP was greater than at STP B, most likely because Cronulla STP has a larger catchment area and effective population.

The marine algae species *Ulva* sp. and *E. radiata* were shown to effectively concentrate ^{131}I from seawater. The CFs for *Ulva* sp. and

E. radiata were calculated to be 176 and 526, respectively. The derived CF for *Ulva* sp. agreed well with literature values but the derived CF for *E. radiata* was substantially lower than the literature CF for *Alaria*. From this we conclude that related species of algae do not necessarily have similar CFs. These derived CFs were calculated for a system that is not at equilibrium due to the discontinuous nature of ^{131}I discharges from the STPs and the relatively short half-life of ^{131}I. Therefore, future studies involving biokinetic modelling of the uptake of ^{131}I by marine algae would be of interest. *E. radiata* is able to detect lower concentrations of ^{131}I as it has a higher CF. However, *E. radiata* is more difficult to collect than *Ulva* sp. and so is not necessarily a better algae species to use as a biomonitor of ^{131}I levels. Uptake of ^{131}I by marine algae was shown to be dependent upon the species, age and location of the algae.

A Tier 2 ERICA dose assessment tool was used to calculate the dose rate to marine biota at Potter Point and outfall B from release of ^{131}I to the marine environment. Dose rates to marine biota under an average and worst case scenario were calculated for both outfall sites. All dose rates at both Potter Point and outfall B were below the ERICA screening value of 10 μGy/h. These low dose rates suggest that a more detailed dose assessment is not warranted. However, the use of a dynamic dose assessment approach (e.g. Vives I Batlle et al., 2008) in future, similar studies should be considered.

The radiological dose rate to humans from ^{131}I due to immersion in seawater or consumption of *Ulva* sp. from Potter Point and outfall B were investigated. The annual dose from consumption of *Ulva* sp. was higher than from immersion. However, dose rates under both exposure scenarios at both outfalls were below the IAEA screening value of 10 μSv/year. The radiological dose rates from Potter Point and outfall B were in all cases, for both biota and humans, comparable.

Iodine-131 was found to be effective as a tracer of effluent dilution and exposure of marine algae to effluent. Radiological dose rates to marine biota and humans at Potter Point and outfall B were investigated and found to be negligible.

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References

Australian Radiation Protection and Nuclear Safety Association, 2007. Potential Exposures to Sydney Water Employees and Members of the Public as a Result of Managing, Discharging and Reusing Effluent Collected from Catchments that Include Hospitals with Nuclear Medicine Capabilities. Commonwealth of Australia, Canberra.

Australian Radiation Protection and Nuclear Safety Association (ARPANSA), 2009. http://www.arpansa.gov.au/radiationprotection/Basics/understand.cfm.

Barrington, S.F., Kettle, A.G., O'Doherty, M.J., Wells, C.P., Somer, E.J.R., Coakley, A.J., 1996. Radiation dose rates from patients receiving ^{131}I therapy for carcinoma of the thyroid. European Journal of Nuclear Medicine 23 (2), 123–130.

Bowen, H.J.M., 1979. Environmental Chemistry of the Elements. Academic Press, London.

Brown, J.E., Alfonso, B., Avila, R., Beresford, N.A., Copplestone, D., Pröhl, G., Ulanovsky, A., 2008. The ERICA tool. Journal of Environmental Radioactivity 99, 1371–1383.

Davis, E., 2006. The Impact of Radionuclide Discharge from Hospitals upon Wastewater Operations. Honours Research Project. School of Chemical Engineering and Industrial Chemistry, University of New South Wales.

Druehl, L.D., Cackette, M., D'Auria, J.M., 1988. Geographical and temporal distribution of ^{131}I in the brown seaweed Fucus subsequent to the Chernobyl incident. Marine Biology 98, 125–129.

Frechou, C., Calmet, D., 2003. Variations of ^{129}I activities in various seaweed collected on the French channel coast. Czechoslovak Journal of Physics 53, A83–A90.

Garnier-Laplace, J., Della-Vedova, C., Gilbin, R., Copplestone, D., Hingston, J., 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.

Hilditch, T.E., Connell, J.M.C., Davies, D.L., Watson, W.S., Alexander, W.D., 1991. Radiological protection guidance for radioactive patients – new data for therapeutic ^{131}I. Nuclear Medicine Communications 12, 485–495.

Hoffmann, E.L., Ferris, J.M., Sept 2002. Environmental and Effluent Monitoring Data for ANSTO Sites, 2001-2. ANSTO Environmental Management Project Report: EM/TN-05/2002. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., July 2002. Environmental and Effluent Monitoring at ANSTO Sites, 2001. ANSTO/E-747. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Farrar, Y., May 1996. Environmental and Effluent Monitoring at Lucas Heights Science and Technology Centre, 1995. ANSTO/E-725. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Farrar, Y., June 1997. Environmental and Effluent Monitoring at Lucas Heights Science and Technology Centre, 1996. ANSTO/E-730. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Farrar, Y., Nov 1998. Environmental and Effluent Monitoring at Lucas Heights Science and Technology Centre, 1997. ANSTO/E-732. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Farrar, Y., Mokhber-Shahin, L., 1999. Environmental and Effluent Monitoring at Lucas Heights Science and Technology Centre, 1998. ANSTO/E-737. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Mohkber-Shahin, L., 2000. Environmental and Effluent Monitoring at ANSTO Sites, 1999. ANSTO/E-740. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Mokhber-Shahin, L., 2001. Environmental and Effluent Monitoring at ANSTO Sites, 2000. ANSTO/E-745. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Ferris, J.M., Markich, S.J., 2003. Environmental and Effluent Monitoring at ANSTO Sites, 2002-2003. ANSTO/E-752. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Ferris, J.M., Harrison, J.J., Loosz, T., 2004. Environmental and Effluent Monitoring at ANSTO Sites, 2003-2004. ANSTO/E-755. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Ferris, J.M., Harrison, J.J., 2005. Environmental and Effluent Monitoring at ANSTO Sites, 2004-2005. ANSTO/E-757. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Ferris, J.M., Harrison, J.J., 2006. Environmental and Effluent Monitoring at ANSTO Sites, 2005-2006. ANSTO/E-761. Australian Nuclear Science and Technology Organisation, Sydney.

Hoffmann, E.L., Loosz, T., Ferris, J.M., 2008. Environmental and Effluent Monitoring at ANSTO Sites, 2006-2007. ANSTO/E-762. Australian Nuclear Science and Technology Organisation, Sydney.

Holmes-Farley, R., 2003. Chemistry and the Aquarium. Iodine in Marine Aquaria: part I. Advanced Aquarists's Online Magazine 2 (3). http://www.advancedaquarist.com/issues/mar2003/chem.htm.

Hou, X., Yan, X., 1998. Study on the concentration and seasonal variation of inorganic elements in 35 species of marine algae. The Science of the Total Environment 222, 141–156.

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

International Commission for Radiation Protection (ICRP), 2004. ICRP Publication 94: release of patients after therapy with unsealed radionuclides. Annals of the ICRP 34 (2), 1–80.

International Commission for Radiation Protection (ICRP), 2007. ICRP publication 103: the 2007 recommendations of the international commission on radiological protection, chapter 5: the system of radiological protection of humans. Annals of the ICRP 37 (2–4), 81–123.

Mairh, O.P., Ramavat, B.K., Tewari, A., Oza, R.M., Joshi, H.V., 1989. Seasonal variation, bioaccumulation and prevention of loss of iodine in seaweeds. Phytochemistry 28 (12), 3307–3310.

Marsh, K.V., Buddemeier, R.W., Wood, W., Smith, C., 1988. Radioiodine in kelp from western Australia. Journal of Radioanalytical and Nuclear Chemistry 123 (1), 199–213.

Martinelango, P.K., Tian, K., Dasgupta, P.K., 2006. Perchlorate in seawater. bioconcentration of iodide and perchlorate by various seaweed species. Analytica Chimica Acta 567, 100–107.

Nozaki, T., Saito, J., 1995. Isotopic double tracers for the measurement of the biological half-life of an element or compound, exemplified by ^{125}I-^{131}I Taken up in a Seaweed. Applied Radiation and Isotopes 46 (12), 1299–1305.

OSPAR Commission, 2002. Discharges of Radioactive Substances into the Maritime Area by Non-Nuclear Industry Radioactive Substances Series, Publication number 161.

Royal Observatory Hong Kong, 1995. Environmental Radiation Monitoring in Hong Kong. Radioactivity Bulletin 1994. Technical Report No. 12.

Saenko, G.N., Kravtsova, Y.Y., Ivanenko, V.V., Sheludko, S.I., 1978. Concentration of iodine and bromine byplants in the seas of Japan and Okhotsk. Marine Biology 47, 243–250.
Shleien, B., Slaback, L.A., Birky, B.K., 1998. Handbook of Health Physics and Radiological Health, third ed. Williams and Wilkins, Baltimore.
Stone, D., Hope, B.K., Harding, A.K., Slaughter-Mason, S., 2008. Exposure assessment and risk of gastrointestinal Illness among surfers. Journal of Toxicology and Environmental Health Part A 71, 1603–1615.
Sydney Water, 2008a. http://www.sydneywater.com.au/OurSystemsandOperations/.
Sydney Water, 2008b. Sydney Water Operating License Compliance Report. Schedule 3: Environmental Performance Indicators. Annual Report, 2007-08.
Teas, J., Pino, S., Critchley, A., Braverman, L.E., 2004. Variability of iodine content in common commercially available edible seaweeds. Thyroid 14 (10), 836–841.
Titley, J.G., Carey, A.D., Crockett, G.M., Ham, G.J., Harvey, M.P., Mobbs, S.F., Tournette, C., Penfold, J.S.S., Wilkins, B.T., 2000. Investigation of the Sources and Fate of Radioactive Discharges to Public Sewers. R&D Technical Report P288. Environment Agency, Bristol.
Twining, J., Hughes, C.E., 2008. Radioecological Risk Analysis of ANSTOs Monthly Effluent Releases, 2006-07. ANSTO Report E-764. Australian Nuclear Science and Technology Organisation, Sydney.
van Netten, C., Hopiton Cann, S.A., Morley, D.R., van Netten, J.P., 2000. Elemental and radioactive analysis of commercially available seaweeds. The Science of the Total Environment 255, 169–175.
Vives, I., Batlle, J., Wilson, R.C., Watts, S.J., Jones, S.R., McDonald, P., Vives-Lynch, S., 2008. Dynamic model for the assessment of radiological exposure to marine biota. Journal of Environmental Radioactivity 99, 1711–1730.