Toxikos
Toxicology Consultants

ABN 30 095 051 791
PO Box 74
Caulfield East
Victoria, 3145
Ph:+61 3 9569 3088
Fax: +61 3 9569 3618
rogerdrew@bigpond.com

Comment on Bell Bay effluent and potential impact on nearby seal colonies

Prepared for: GHD Pty Ltd (on behalf of Gunns Ltd)

Prepared by: Dr Roger Drew, DABT ,
John Frangos M. App. Sc. (Toxicol.)
Toxikos Pty Ltd.

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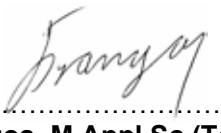

Roger Drew, Ph.D., D.A.B.T.
(Diplomate American Board of Toxicology)

John Frangos, M.Appl.Sc (Toxicology)

Executive summary

Toxikos was requested by GHD and Gunns to provide a brief commentary on the likely impacts of discharged mill effluent on a nearby seal colony. The scope of the commentary was restricted to potential bioaccumulative effects of substances in the effluent in order to address concerns articulated by a community member during consultation about the impacts of the Bell Bay mill.

The nearest seal colony to the ocean outfall site is on Tenth Island, approximately 12 km to the north east of the outfall site (Aquenal 2005) and is home to approximately 200-500 seals which are predominantly Australian fur seals (*Arctocephalus pusillus doriferus)*. The diet of Australian fur seal breeding colonies in Bass Strait is 80-90% fish. The most realistic potential for exposure of seals to substances in the mill effluent is through eating fish that may have accumulated the chemicals. Toxikos prepared a human health risk assessment (HHRA) that evaluated the potential for impact on human health by considering likely bioaccumulation of effluent constituents by fish caught in the vicinity of the Bell Bay effluent outfall and were subsequently consumed by humans. The HHRA report is titled "Human Health Risk Assessment - Bell Bay Pulp Mill Effluent", Toxikos Document TR081205-RJF, dated December 2006. Hence this commentary for seals draws upon the information in the HHRA.

Using a screening methodology that is based on *possible* bioaccumulation potential, described in detail in the HHRA, three metals (cadmium, mercury, selenium) and dioxins were identified as substances that may be present in the mill effluent and that required consideration with respect to potential impact on seals. The diffuser at the end if the effluent pipeline is to be designed to achieve a 1 in 100 mass dilution of the effluent within a short distance of about 100m either side of the diffuser. This dilution zone is called the DZ100 and the area at the edge, and consequently beyond the DZ100 is the focus of the HHRA and this commentary.

Selenium:

Selenium is an essential element required by fish and other animals for maintenance of normal biochemical functions. Its' uptake is controlled by homeostatic mechanisms and accumulation occurs when these are overwhelmed. This can happen when water concentrations are greater than 3 – 5 µg/L. Because the estimated selenium concentration at the margin of the DZ100 will be within the background range measured around the world, and at least an order of magnitude less than the lowest water concentration associated with accumulation in fish, and less than the ANZECC water quality guidelines for protection of aquaculture, an incremental increase in selenium concentrations in fish around the outfall is unlikely.

These considerations lead to the conclusion that selenium discharged in the mill effluent will have negligible impact on seals.

Mercury:

The theoretical incremental increase in fish Hg levels due to the mill effluent is very small and unlikely to be detectable with analytical techniques routinely used to monitor Hg in fish. Thus the effluent discharge is unlikely to statistically change the distribution pattern of Hg in fish that have been caught in the area of the proposed outfall.

Furthermore the average concentration of Hg in fish is approximately 50% of the standard set to protect human health, specifically the developing foetal brain, consequently it is presumed the health of seals will also be protected since the amount of fish that seals may catch at the outfall is likely to be a very small proportion of their total dietary intake.

It is concluded mercury discharged in the mill effluent is very unlikely to impact seals.

Cadmium:

Cadmium is bioaccumulated by sedentary organisms such as mussels but not readily by fish. For fish flesh to accumulate cadmium high water concentrations are needed and exposure of the fish needs to be prolonged. Nevertheless if muscle does accumulate cadmium it quickly loses the metal when the fish transfers to water of low cadmium concentration. Some of this cadmium is sequestered by kidney and liver. A mass dilution of the effluent cadmium gives a water concentration at the DZ100 of 0.012µg/L. Levels of cadmium in fish at the location t are less than analytical detection limits of 1 mg/kg fish. Existing levels of cadmium in fish are not recognised as causing issues in seals so it is not expected the small incremental increase in cadmium concentrations due to the effluent will have a significant impact on fish cadmium levels.

Overall it is considered unlikely that seals will be affected by cadmium in the effluent but there is large uncertainty associated with this opinion due to data uncertainty regarding existing cadmium levels in the area.

Dioxins:

The following conclusions regarding the bioaccumulation of dioxins by marine organisms have been formulated by review of the literature available for this project:

- Dioxins do not magnify through the marine food chain.
- Dioxins are not significantly bioaccumulated or biomagnified by fish.
- Fish living in local environments where dioxin concentrations are low also have low levels of dioxins.
- The theoretical increase in dioxin levels of fish after the effluent outfall becomes operational is markedly less than the analytical capability to measure.
- There will be no demonstrable increase in dioxin concentrations of fish that might reside around the ocean outfall.
- Animals higher in the food chain, including whales and seals, have relatively high metabolic and/or excretory capacity towards dioxins.
- Seals do not readily assimilate dioxins from their food and do not biomagnify dioxins.

In relation to effluent dioxin release to the ocean, an incremental increase in fish dioxin concentrations will not be demonstrable, and since seals do not easily assimilate dioxins from their food and data indicates dioxins do not biomagnify through the food web to seals, it follows there will be no demonstrable impact on seals from dioxins in the effluent.

The above, coupled with the fact the area of ocean potentially impacted by the discharged effluent is quite small relative to the total range of the fur seals[1], leads to the conclusion the viability of the seal colony at Tenth Island will not be affected by the effluent discharge

[1] The small relative area potentially affected by the effluent means only a small fraction of the seal diet will be sourced from around the outfall. This will also be the case should fish be attracted to the area of initial dilution (1/100) near the effluent diffuser.

1. Introduction

1.1 General background

Toxikos was requested by GHD and Gunns to provide a brief commentary on the likely impacts of discharged mill effluent on a nearby seal colony. The scope of the commentary was restricted to potential bioaccumulative effects of substances in the effluent in order to address concerns articulated by a community member during consultation about the impacts of the Bell Bay mill.

It is common practice in risk assessment to first conduct a screening evaluation to determine whether a full, formal quantitative risk assessment is required. Thus this commentary is not a formal risk assessment or a stand alone report; detailed logic and information underpinning the summary information contained in the commentary are referenced back to the human health risk assessment (HHRA)[2] for effluent discharge. This commentary is a qualitative and semi-quantitative analysis that indicates a formal quantitative risk assessment for seals is not warranted.

The HHRA evaluated the potential human health impact by considering likely bioaccumulation of effluent constituents by fish and their consumption by humans. Given that seals might also eat fish caught from near the effluent discharge area, information in the HHRA is pertinent for evaluation of risks to seals. This commentary therefore liberally draws upon technical background information contained in the HHRA and the reader is referred to the HHRA for the specific details that may be of interest.

1.2 Dilution zones

The diffuser at the end of the effluent pipe will be designed to achieve an effluent dilution of 100 fold within a short distance of release from the pipe[3]. The HHRA addressed the human health impacts of incremental increases in seawater concentrations of effluent constituents at these assumed dilutions. Furthermore the diffuser is to be engineered so the target dilutions will be

[2] "Human Health Risk Assessment - Bell Bay Pulp Mill Effluent", Toxikos Document TR081205-RJF, dated July 2006.
[3] Personal verbal communication Gunns Pty Ltd (2005), Jaakko Pöyry (2005) and GHD (2006). The diffuser will be 200m long with two rows of 15 cm holes in the upper half (in section at 10 & 2 o'clock) of the pipe which will be lying on the seabed.

achieved within a short distance[4] of the diffuser, approximately 100m. This commentary also deals with the impact of the incremental concentrations of effluent constituents at the edge of the 1 in 100 zone of initial dilution, for ease of reference this is termed the DZ100.

The 1:100 dilution zone (DZ100) is defined as the mass dilution of discharged effluent. The 1:100 dilution was selected based on the findings of eco-toxicological testing undertaken on representative effluent samples from similar operating pulp mills. The test reports concluded in reference to the 1:100 dilution mixing zone that assuming the effluent from the proposed pulp mill is the same as that sampled and tested from the representative mills, and that the sample was representative of effluent quality over time, then based on the eco-toxicology testing results, no acute or sub-acute lethal toxicity would be expected to be observed at the edge of the DZ100 mixing zone.

The DZ100 is different to the 'mixing zone' defined by RPDC (2004) and is likely to be smaller. The 'mixing zone' is defined in the RPDC (2004) emission limit guidelines for any new bleached eucalypt kraft mill in Tasmania as "a three dimensional area of the receiving waters around a point of discharge of pollutants within which it is recognised that the water quality objectives for the receiving waters may not be achieved (Sate Policy on Water Quality Management).

1.3 Routes of exposure

The diet of fur seals is 80-90% fish, the balance being cephalopods, thus fish are the primary route of potential exposure for seals to discharged effluent. Cephalopods (squid, octopus etc) were not observed within 1 km of the outfall site during survey conducted by Aquenal (2005).

Within the relatively small area of the DZ100, effluent concentrations will be higher than at the edge. This is not a significant issue for those effluent substances that have the necessary properties for bioaccumulation because the extent of accumulation in fish is determined by the average concentration to which the organism is exposed, which in turn is governed by the concentrations prevailing over the fish's entire range and the time spent in various portions of the range, and by the rate of removal of the substance by the fish. The area of the DZ100 is approximately 0.04km^2 whereas the foraging range of fish that may inhabit the area is larger.

[4] Estimates of the distances from the diffuser where the 1 in 100 dilution occurs will be determined by the hydrodynamic modelling being undertaken by GHD. The risk assessment herein has been conducted independent of the size of the DZ100, however the smaller the area of DZ100 relative to the foraging area of the fish the lower the probability that a fish will spend significant amounts of time within the DZ100. based on the assumption that the 1 in 100 dilution will be achieved within 100m of either side of the pipe

Brief exposures to the higher concentrations that may occur within the DZ100 zone have relatively little impact on the final concentration achieved in the organism, especially if such exposures are small compared to the half life of the substance in the organism in question[5].

The area in which the end of the effluent pipeline is to be located is barren, coarse grained sand with a few low rock ledges; this is a seascape typical of strong currents (Aquenal 2005). Fish are scarce within a 1 km radius. It is considered that if the effluent discharge was to increase nutrients in the area and attract fish, the fish would be rapidly depleted by seals should they feed upon them. The area of the DZ100 is approximately 0.04 km^2 which is a tiny fraction of the 100 – 200 km^2 foraging range of seals. Consequently seals will potentially spend very little time within the DZ100. These considerations, together with the negative ecotoxicity testing results of 1:100 effluent dilutions, indicate it is very unlikely seals will experience direct toxicity from exposure to effluent within the DZ100.

It is concluded that the most realistic route of potential exposure that seals may have to substances discharged in the effluent is by consuming fish that may have assimilated the compounds. Consequently this commentary focuses on the issues of bioaccumulation of effluent compounds by fish and the potential impact of their consumption by seals. This is directly analogous to the HHRA.

2. Bioaccumulation potential of Bell Bay effluent constituents

2.1 Screening procedures

A full description of the screening procedures is presented in the HHRA. The following sections briefly outline the methodology.

2.1.1 Identifying effluent constituents, screening for 'Chemicals of Interest'

The first step was to determine what may be in the effluent. A list of effluent constituents was compiled by reviewing the literature and any substance found to be reported in pulp mill effluent, regardless of age or type of mill, was regarded as being possibly present in the Bell Bay effluent. Then based on the bleaching chemistry of chlorine dioxide, mill engineering modifications and advice from Jaakko Pöyry this 'candidate list of chemicals' was culled to produce a list of 'chemicals of interest' for the Bell Bay effluent. If there was uncertainty about a

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the DZ100 area may be (100 + 100) x 200m = 40,000m^2 = 0.04 km^2.

[5] The time it takes to reach steady state body burden concentrations is about 3 – 5 half lives. A half life is the time required for a given body concentration to decrease to half its initial value.

historical effluent constituent being in the Bell Bay mill effluent, or not, it was assumed to be present. Final effluent concentrations for the 'chemicals of interest' were determined and these substances then subjected to screening processes for bioaccumulation potential.

2.1.2. Screening for bioaccumulation in biota.

The bioaccumulation screening methodologies detailed in the HHRA identified effluent constituents which have potential to be concentrated in either muscle or fat by any aquatic organism and thus perhaps pose a hazard to seals consuming those organisms. The screening methodology in the HHRA was technically based on molecular weight, known potential for bioaccumulation in any aquatic organism, and readiness of metabolism/excretion. If a substance in the mill effluent had been identified as having bioaccumulative potential then it was included. Substances coming forward from this screening process were considered to be 'chemicals of potential concern' and subject to further assessment. To these were added specific substances about which stakeholders had articulated concern.

2.2. Chemicals of potential concern

Using the screening methodology outlined in Section 2.1.2 above, the chemicals of potential concern (based on *possible* bioaccumulation), were determined to be three metals (cadmium, mercury, selenium) and dioxins. It should be noted however that because an organism at a low trophic level in the food web may take up a chemical, it does not automatically follow that predators of that organism also necessarily bioaccumulate the chemical. The latter is dependent on the range of organisms upon which the predator species feeds, the proportion of food containing the chemical that is sourced from the area, and the metabolic capacity of the predator species to metabolise and/or excrete the chemical.

With regard to dioxins, a review of the literature for bioaccumulation of dioxins by fish indicated minimal bioaccumulation potential. Nevertheless dioxins were included on the list chemicals of concern because of past association with old technology pulp mills and stakeholder interest in this group of chemicals.

3. Seal colony at Tenth Island

The nearest seal colony to the ocean outfall site is on Tenth Island which is approximately 12 km to the north east of the outfall site (Aquenal 2005) and is home to approximately 200-500 seals which are predominantly Australian fur seals (*Arctocephalus pusillus doriferus)* (Pemberton & Gales 2004). Hence the following description of seal diet, foraging behaviour and general biology is intended, where possible, to be specific to the Australian fur seal.

The Australian fur seal is the most common seal in Tasmanian waters and breeds on small isolated rocks in Bass Strait between October and January. It also hauls-out at various rocky areas around the Tasmanian coastline, especially outside the breeding season when many seals disperse from the breeding colonies (Parks & Wildlife Service Tasmania 2003).

According to Arnauld & Warneke (2002) adult Australian seals weigh on average 229 kg and 85 kg for males and females respectively.

As lactating females must return regularly to suckle dependent young, they are relatively restricted in the duration and extent of foraging trips. Nevertheless telemetry studies with lactating female seals in Bass Strait have identified a wide foraging pattern, up to 500km but normally within 100 km of the colony (Arnould and Hindell 2001).. An Australian fur seal was observed just to the west of the diffuser site, within the 1 km radius study area during the marine and biota survey of the outfall site (Aquenal 2005). This indicates the outfall is within the foraging zone of Australian fur seals. The effluent pipeline diffuser will be located in the centre of a large area of coarse sand that currently has very limited marine organisms (Aquenal 2005). It is therefore unlikely that seals at Tenth Island obtain, or will obtain a large proportion of their food at the ocean outfall.

The dive behaviour and foraging locations of 36 lactating female Australian fur seals (*Arctocephalus pusillus doriferus*) from Kanowna Island (Wilsons Promotory Reserve Bass Strait) was determined throughout lactation during 1997–1999 (Arnould and Hindell 2001). A total of 664 foraging trips, 687 maternal-attendance periods, and 599 complete foraging cycles (foraging trip plus subsequent onshore attendance) were recorded, spanning all seasons and stages of the lactation period. The lactating females spent almost all of their time in Bass Strait and concentrated their time in waters within 100 km of the colony. Dives were benthic in nature which is consistent with the observation that most dives were to a depth of 70 to 85 metres. Post-lactation, females have a wider foraging range and sometimes left Bass Strait (Arnould and Hindell 2001). According to DPIWE (2005) six Australian fur seal cows from Tenth Island

have been equipped with satellite-linked-time-depth recorders and the results show that cow seals forage mostly within 200 km of the breeding colony. Dive information from one of these seals revealed that her dives, mostly at night, lasted about three minutes each and took her to depths below 45 metres.

Investigations of the diet of Australian fur seal breeding colonies in Bass Strait have been conducted (Gales and Pemberton 1994, Hume et al. 2004). Of the samples collected and analysed between 10 and 30% were collected from Tenth Island. The most important food source by analysis of faeces and regurgitates was fish (~80-90% of the diet) followed by cephalopods which contributed 10-20%; crustaceans and birds contributed negligible amounts to the diet. The main prey species were leatherjacket (family *Monocanthidae*), redbait (*Emmelichthys nitidus*), barracouta (*Thyristes atun)* and Jack mackerel (*Trachurus declivis*). The frequency of occurrence of flathead in the diet was less than 4% (Gales and Pemberton 1994, Hume et al. 2004).

4. Assessment of bioaccumulation potential in seals

4.1 Selenium

Selenium is an essential micronutrient in animals. Trace concentrations are required for normal growth and development, and at moderate water concentrations of selenium homeostatic regulation maintains body burdens within normal physiological ranges (Hamilton 2004, ANZECC 2000). Fabris et al. (2005) studied selenium concentrations in edible tissue of snapper, flathead, lobster and abalone in coastal waters of Victoria. Although the concentration of selenium in water or sediment was not reported an important observation was the concentration of selenium was approximately uniform atapproximately 0.5 µg/g in all finfish populations examined and was consistent with previous studies cited by the author for a large number of fish species obtained from near-shore environments of Australia. The explanation being that for essential elements biochemical uptake mechanisms maintain constant tissue concentrations (e.g. Skinner et al. 2004) therefore at low water concentrations, presumably at or about background, bioaccumulation is not expected.

A series of experiments in lakes in Sweden showed selenium bioaccumulated in fish via the food chain if concentrations were greater than 3-5 µg/L (Paulsson and Lundbergh 1989, 1991, 1994, Lindqvist et al. 1991). Similarly studies in Canada concluded selenium was accumulated

through the food chain of fish if water concentrations were high and recommended aquaculture "additions" of selenium be limited to 1 µg/L (Rudd et al. 1980, Turner and Rudd 1983, Turner and Swick 1983).

The estimated concentration of selenium at the periphery of the DZ100 around the outfall is 0.075 µg/L. This is within the range for coastal waters around Australia (<0.01-0.08 µg/L) and world wide sea water (0.009-0.45 µg/L) (Appendix 4 of HHRA). Data is not available for the selenium concentration of seawater local to the proposed outfall. However if it is assumed the concentration is at the high end of that for coastal Australian waters then at the edge of the DZ100 the concentration[6] of selenium will be 0.15 µg/L. Thus the predicted selenium concentration attributable to mill effluent at the DZ100 is much less than water concentrations identified as resulting in bioaccumulation of selenium by fish. Furthermore the concentration is less than the 10 µg/L ANZECC (2000) guidelines for protection of aquaculture and human consumption of seafood.

Conclusion

Selenium is an essential element required by fish and other animals for maintenance of normal biochemical functions. Its' uptake is controlled by homeostatic mechanisms and accumulation occurs when these are overwhelmed. This can happen when water concentrations are greater than 3 – 5 µg/L. Because the estimated selenium concentration at the margin of the DZ100 is within the background range measured around the world, and at least an order of magnitude less than the lowest water concentration associated with accumulation in fish, and less than the ANZECC water quality guidelines for protection of aquaculture, an incremental increase in selenium concentrations in fish around the outfall is unlikely.

It is concluded selenium discharged in the mill effluent will have negligible impact on seals.

[6] Detection limits for selenium in seawater in analysis conducted by Aquenal (2005) were 2 µg/L, results were <2 – 2 µg/L and are therefore not informative. Assuming background concentration is 0.08 µg/L then a 1:100 dilution of effluent with this seawater will give rise to a concentration at the edge of the DZ100 of [(1 x 7.5 µg/L) + (99 x 0.08)]/100 = 0.15 µg/L.

4.2 Mercury

It is expected the effluent treatment processes of the Bell Bay mill will remove most if not all detectable mercury before discharge. The highest estimated concentration of Hg in discharged effluent from pulped native eucalypt, plantation eucalypt or pine is 0.275 µg/L (JP 2005). The estimate was made by the mill designers and is based on analysis of pooled wood chip samples that are representative of feed stock for the Bell Bay mill and application of conservative mass balance assumptions. However based on Swedish data[7] the concentration of Hg in effluent could be less than 0.016 µg/L, i.e. approximately 20 times less. The form of mercury in the effluent is not known, but is likely to be mercuric salts (i.e. Hg^{++}).

At the edge of the DZ100, the 100 fold mass dilution will result in a Hg concentration of approximately 0.003 µg/L. While this is within the range of Australian coastal and open ocean waters the existing background concentration of Hg in the receiving water should also be taken into consideration. Unfortunately an accurate quantitation of Hg in the water around the proposed diffuser site is unavailable; recent seawater analyses by GHD (2006a) placed the Hg concentration at less than the analytical detection limit of 0.1 µg/L. The Department of Primary Industry, Water and the Environment ( ) found the concentration of mercury in water to be below detection (<0.05 µg/L) at Hebe Reef and other locations in the Tamar estuary[8]. Background concentrations of Hg for Australian coastal waters are recorded to be <0.001 – 0.02 µg/L (DEH 1995). Assuming the Hg level at the diffuser site is at the midpoint of this range the concentration would be approximately 0.01 µg/L.

Using this value as the existing background seawater Hg concentration gives a 1:100 dilution of the effluent Hg using seawater as the diluent as follows:

$$[(1 \times 0.275\ \mu g/L) + (99 \times 0.01\ \mu g/L)] \div 100 = 0.013\ \mu g/L$$

This assumed DZ100 concentration is within the concentration range reported for Australian coastal waters and it might therefore be expected that background fish Hg concentrations will not alter very much and there will be negligible impact on seals. However there is a great deal of uncertainty associated with the background Hg concentration assumption; the calculation

[7] Email to Toxikos from ,Jaakko Pöyry Oy dated 24/11/2005.
[8] The Australian and New Zealand Environmental Conservation Council (ANZECC) have established a water quality guideline (WQG) for mercury in marine waters of 0.1 µg/L. The WQG is intended for the protection of aquatic organisms however ANZECC consider that the use of the WQG designed for the protection of areas of high conservation value (i.e. 99th percentile level of protection) to be precautionary for bioaccumulation and biomagnification in aquatic organisms.

should only be used to provide a rough indication of the possible impact of discharged effluent Hg on Hg concentrations in seawater surrounding the diffuser.

Mercury (Hg) in the aquatic environment exists mainly as inorganic mercury or in methylated forms (ANZECC 2000, Morel et al. 1998). The major components of mercury in seawater are complexes of the divalent inorganic form, Hg^{++} (ANZECC 2000). Under certain conditions, inorganic mercury in sediments and water can be biologically converted into methylated mercury by microbes (Morel et al. 1998, Mauro et al. 2002, ANZEC 2000).

Methylmercury (MeHg) can easily penetrate the biological membranes of microorganisms and is efficiently accumulated by these organisms by covalently binding to protein sulphydryl groups. Subsequent accumulation in aquatic food chains is mainly due to ingestion of MeHg containing microorganisms (Morel et al. 1998). Fish bioaccumulation factors for MeHg are consequently very high (US EPA 2001, ANZECC 2000).

The HHRA estimated the incremental increase in MeHg that might occur in fish around the effluent outfall and added this to the Hg concentrations determined in fish caught from the area. This required information on the extent of methylation of Hg discharged in the effluent and the accumulation of MeHg by fish.

A literature search for the proportion of inorganic Hg converted to MeHg in either open ocean or coastal waters revealed a range of 0.03% to 3% based on empirical observations (Mason et al. 1999, Mason & Sullivan 1998, Benoit et al. 1998, Ullrich et al. 2001, Rolfhus and Fitzgerald 1995). For conservative calculation of the amount of MeHg that may be formed from Hg discharged in effluent the top of the observational range was chosen, i.e. 3%.

The bioaccumulation factor (BAF) is defined as the ratio of MeHg concentration in fish flesh divided by the concentration of dissolved MeHg in the water column. The BAF represents the accumulation of Hg in fish of a specific trophic level from both direct uptake from water and uptake from the food web (i.e. it includes consideration of sediment and sediment dwelling organisms).

A database of bioaccumulation factors has been established by the US EPA (1997) using data either directly obtained from field studies (most of the field studies were conducted in the Great Lakes region of the USA) or obtained indirectly by estimating a BAF from the bioconcentration

factor (i.e. transfer factor of mercury from water column only) and 'food chain multiplier' (a factor to account for food chain exposure to mercury). The data was then used to derive probability distributions of bioaccumulation factors for fish (US EPA 1997).

The US EPA (1997) considered the variability in BAF values either directly measured or indirectly estimated and considered the median value of the directly measured BAF was most representative for Hg accumulation in fish in fresh water lake environments. For foraging/predator fish the median BAF was 1,600,000 dry-weight which at 80% water content of fish translates to a BAF of 320,000 on a wet weight basis. The same BAF value has been used by Environment Canada to establish tissue residue guidelines for the protection of wildlife consuming fish (EC 2001) and also by the US EPA in establishing tissue residue guidelines for human consumption of fish (US EPA 2001).

Thus the incremental increase in MeHg concentration due to discharged Hg in effluent is:

$$IC_{FISH} = (Hg_{EFF} \times C_{Hg\text{-}MeHg}) \times BAF$$
$$= 0.00275 \mu g/L \times 0.03 \times 320{,}000 \text{ L/kg}$$
$$= 26.4 \text{ µg/kg wet weight fish (0.026 mg/kg fish).}$$

Where:
IC_{FISH} = Incremental increase in fish MeHg due to effluent.
Hg_{EFF} = Hg concentration at DZ100 (0.00275 µg/L) due to effluent.
$C_{Hg\text{-}MeHg}$ = Conversion of effluent Hg to MeHg (3%, see HHRA).
BAF = Bioaccumulation Factor for MeHg (320,000 L/kg; see HHRA).

In two rounds of sampling, April 2005 and February 2006, a total of 39 fish of different varieties have been caught within 250m to the east and west of the proposed diffuser site and analysed for Hg concentration (Aquenal 2005, GHD 2006a).

The concentration of Hg in the 39 fish was 0.22 ± 0.15 mg/kg (mean ± SD). Twenty four of the fish had Hg levels below the analytical detection limit[9] hence to calculate the statistics the level of Hg was assumed to be at half the detection limit for these animals. The distribution of Hg in fish is shown in Figure 1. If the discharged Hg in the effluent caused an incremental increase in fish Hg content of 0.026 mg/kg as conservatively calculated above, the average fish Hg concentration will increase from 0.22 mg/kg to 0.25 mg/kg. The consequence of this incremental

[9] The analytical detection limit (DL) for Hg in fish was 0.1, 0.2 or 0.5 mg/kg depending on the analytical run. The assignment of 0.5 DL for fish with Hg analytical non-detects in the calculation of statistics was done according to the respective detection limit for the batch within which the specific non-detect fish resided.

increase on the distribution pattern of Hg in fish is also shown in figure 1. It should be noted an incremental increase in fish Hg of this magnitude will not be statistically demonstrable if measurements are conducted on fish. This is especially so given the estimates of Hg in the effluent are considered to be over predictions and a high conversion of Hg to MeHg is assumed in the calculations for estimating incremental increases in fish Hg.

For the type of fish caught in the area of the proposed outfall, Foods Standards Australia and New Zealand (FSANZ 2004) have established a limit of 0.5 mg Hg/kg fish (as an average) for human consumption. This is to protect the developing foetal brain against Hg-induced subtle learning defects, higher doses than those that occur from consumption of fish with Hg levels at the standard are required to induce effects that might be detectable in seals. Although seals eat more fish than humans it is probable only a very small amount of their diet will be sourced from the ocean outfall because seals have a wide foraging range and the outfall will be quickly fished out.

Keeping in mind the conservatism embedded in the calculations, and the fact that a measurable change in fish Hg levels will not be demonstrable, the small theoretical change in fish Hg concentration will have no impact on seals.

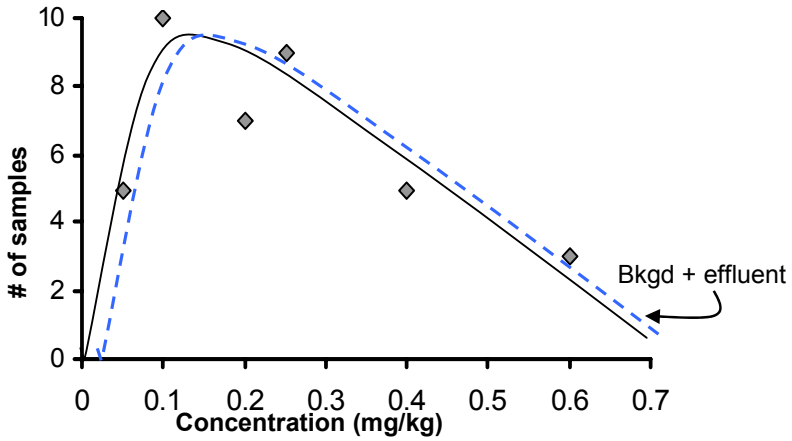

Figure 1: Background fish Hg concentrations (mg/kg) and theoretical impact of effluent Hg .
The solid curve represents the distribution of Hg in fish caught in the area of the proposed outfall.
The dashed curve represents the theoretical increase in fish Hg level due to effluent is 0.026mg/kg.

Conclusion:

The theoretical incremental increase in fish Hg levels is very small and unlikely to be detectable with analytical techniques routinely used to monitor Hg in fish. Thus the effluent discharge is unlikely to statistically change the distribution pattern of Hg in fish that have been caught in the area of the proposed outfall.

Furthermore the average concentration of Hg in fish is approximately 50% of the standard set to protect human health, specifically the developing foetal brain, consequently it is presumed the health of seals will also be protected consequently it is presumed the health of seals will also be protected since the amount of fish that seals may catch at the outfall is likely to be a very small proportion of their total dietary intake.

In considering the above it is concluded mercury discharged in the mill effluent is very unlikely to impact seals.

4.3 Cadmium

In contrast to mussel, fish are not avid accumulators of cadmium (Jarvinen and Ankley 1999, ANZECC 2000). This may be because cadmium residues in fish muscle reach steady-state only after long exposure periods and muscle loses accumulated Cd when the fish moves to clean water (Sangalang and Freeman 1979, de Conto Cinier 1999). Pharmacokinetic modelling of data showed that while whole body cadmium levels (represented by blood, gill, gut wall, liver and alimentary canal contents but not muscle) in trout reached steady state in approximately 50 days, the concentrations in kidney did not attain steady state even after 350 days (Thomann et al. 1997). Whole body depuration occurred with an initial half life of approximately 6 days followed by a longer terminal phase with a half life of approximately 24 days [10], on the other hand kidney concentrations continued to increase during depuration. This is consistent with the experimental data of de Conto Cinier (1999) who showed exposure of carp to high concentrations of water borne cadmium resulted in concentrations of the metal increasing sharply in kidney and liver, but not muscle. For muscle, cadmium concentrations only became significant after 106 days. After 127 days exposure to 53 µg/L, the cadmium concentration in kidney was 4 fold higher than in liver and 50 fold higher than muscle; at 443 µg/L these ratios were 2 and 100 respectively. With depuration however the loss of accumulated cadmium was rapid and immediate in muscle but no decrease was observed for liver and kidney. Similar

[10] These depuration half lives are graphical estimates made by Toxikos from Figure 7 of Thomann et al.

results have been found for Japanese eels (Yang et al. 1996), zebrafish (Wicklund et al. 1988) and girella (Kuroshima 1987).

Metallothionein is generally regarded as a high affinity sink for some non-essential metals (Segnar and Braunbeck 1998). This is the case for cadmium, it is avidly bound by metallothionein and tissue sequestering of cadmium in aquatic animals, as in mammals (Goyer and Clarkson 2001) and sea birds (Stewart et al 1996, Elliott et al. 1992, Elliot and Scheuhammer 1997), correlates with metallothionein concentration and inducibility which is highest in liver and kidney but low in muscle (Roesijadi 1992). This explains the relative inability for fish muscle to accumulate and retain cadmium, cadmium accumulated by muscle quickly redistributes to liver and kidney where it is sequestered. Hence kidney concentrations continue to rise when the levels of cadmium in other tissues are decreasing during depuration.

The background concentration of Cd in fish around the proposed outfall site was below analytical detection limits (<0.1 mg/kg) in 34 of 39 samples; the level in the remaining 5 samples were 0.1 (n=4) or 0.2 mg/kg.

It is thus concluded that cadmium does not easily accumulate in fish muscle. High concentrations in the aquatic environment and long term exposure are required before significant concentrations are observed in fish muscle (de Conto Cinier 1999, Kraal et al. 1995, Papoutsoglou and Abel 1988).

The concentration of cadmium in effluent is conservatively estimated to be 1.2 µg/L. With a mass dilution of 1: 100 the concentration at the fringe of the DZ100 would be 0.012 µg/L. While this concentration is markedly less than those associated with accumulation of cadmium in fish flesh (de Conto Cinier 1999, Kraal et al. 1995, Papoutsoglou and Abel 1988) it does not take into account background levels of cadmium in seawater.

The predicted concentration of 0.012 µg effluent cadmium/L after mass dilution at the edge of the DZ100 is clearly below the range of cadmium concentrations of <0.2-0.4 µg/L measured by Aquenal (2005) at the proposed location. Elsewhere in the world background concentrations for marine waters have been measured to be between 0.001-0.7 µg/L with Australia tending towards the upper end of the range (HHRA Table A4.1, Appendix 4). Unfortunately quantitative data for cadmium in seawater and fish from the proposed site of the outfall are not available, these measurements being recorded as less than analytical detection limits by Aquenal (2005).

(1997).

An assessment similar to that undertaken for mercury is therefore not possible.

Conclusion:

For cadmium a literature review showed it is bioaccumulated by sedentary organisms such as mussels but not readily by fish. For fish flesh to accumulate cadmium high water concentrations are needed and exposure of the fish needs to be prolonged. Nevertheless if muscle does accumulate cadmium it quickly looses the metal when the fish transfers to water of low cadmium concentration. Some of this cadmium is sequestered by kidney and liver.

A mass dilution of the effluent cadmium gives a water concentration at the DZ100 of 0.012µg/L. While this level is not expected to result in marked incremental accumulation of cadmium because it is less than background concentrations around the world and the latter are not associated with cadmium accumulation by fish it does not take into account the background concentrations at the site. In addition levels of cadmium in fish at the location have not been quantitated but are less than analytical detection limits of 1 mg/kg fish. Existing levels of cadmium in fish are not recognised as causing issues in seals.

Overall it is considered unlikely that seals will be affected by cadmium in the effluent but there is large uncertainty associated with this opinion.

4.3 Dioxins

4.3.1 Dioxin content of effluent

The bleaching technology to be used at the proposed Bell Bay mill means dioxin formation will be very limited, which, together with efficient effluent treatment, results in anticipated dioxin concentrations in the discharged effluent that are non-detectable (JP 2005) The maximum concentration[11] of dioxins in discharged effluent is estimated by the mill designers to be 0.074 pg/L (conservatively assumed by Toxikos to be TCDD equivalents[12]) and is well below the RPDC guideline for the final effluent (at the point of discharge) of 10 pg/L and 30 pg/L for 2,3,7,8-TCDD and 2,3,7,8-TCDF respectively, these limits are equivalent[13] to 13 pg TEQ/L. Thus the amount of TCDD equivalents discharged in the mill effluent are 175 times less than the RPDC requirements.

Because the effluent concentration of dioxins will be markedly less than the RPDC approval guidelines it might be anticipated impact on marine fauna would be non-significant. Nevertheless the potential for seals to accumulate dioxins has been examined in detail below.

4.3.2 Potential for bioaccumulation of dioxins by fish

Any fat soluble substance present in high concentrations in fish is a potential candidate for bioaccumulation by seals due to their diet being primarily fish, and seals having high levels of body fat and long life span. As discussed below, whether such a substance in fish bioaccumulates in seals is dependent on the capacity of seals to metabolise and excrete the substance. Historically several authors have reported relatively high levels of *dioxin equivalents* in marine mammals such as whales, seals and porpoise. However one has to be extremely careful in interpreting this information because levels of PCBs in fish and marine mammals are

[11] Analytical detection limits for dioxins in effluent range from 0.3 – 9 pg TCDD/L (Shariff et al. 1996) to up to 10 pg TCDD/L (ALS Environmental 2005). Thus the concentration of dioxins in the discharged effluent will be 4 – 135 times lower than the analytical detection limit. It is noted Australian Laboratories tend to have higher dioxin quantitation limits, therefore, in round numbers the amount of dioxin in effluent will be about two orders of magnitude less than the ability of the laboratory techniques likely to be used to monitor this parameter in effluent.

[12] The mill designers, Jaakko Pöyry, provided Toxikos with a mass estimate of dioxins in the final discharged effluent. This could be made up of a number of low chlorinated dioxin congeners. By assuming the mass estimate of Jaakko Pöyry was as TEQ effectively assumed all the discharged dioxin was TCDD and is therefore conservative.

[13] The WHO TEF for 2,3,7,8-TCDD is 1.0 and for 2,3,7,8-TCDF is 0.1 (van den Berg et al. 1998). Hence for the RPDC limits of 10 pg/L TCDD and 30 pg/L TCDF the TCDD TEQ of the RPDC limits = (10 x 1) + (30 x 0.1) = 13 pgTEQ/L.

also often reported as dioxin equivalents and while PCBs have significant potential for bioaccumulation and biomagnification, dioxins do not. The difference between the bioaccumulation of PCBs versus dioxins in marine mammals is due to dioxins being metabolised whereas PCBs are not. Thus reports of high concentrations of dioxin equivalents in blubber or flesh of marine mammals relate to bioaccumulation of PCBs and not dioxins (see discussion below). PCBs are not formed in the Bell Bay pulping process and no PCBs are released into the environment.

In an Australian ecological risk assessment conducted for the Department of Environment and Heritage, Gatehouse (2004) makes an overview comment that available field-based aquatic bioaccumulation studies generally show a progressive increase in tissue dioxin like material from low to high trophic levels. This statement appears to be contradictory to detailed investigations of dioxin bioaccumulation, notably in the Great Lakes, where biomagnification of TCCD through the food chain "*is significant between fish and fish-eating birds but not between fish and their food, or fish and sediment"* (Gatehouse 2004) [Indicating no bioaccumulation by fish to greater levels than is in their environment].

The latter conclusion by Gatehouse (2004) is supported by Australian studies in which biosediment accumulation factors[14] (BSAF's) for a variety of fish species and bivalves have been empirically determined to be less than unity (Gatehouse 2004, discussed in more detail below), and by studies not reviewed by Gatehouse (2004) but described later in this section.

Potential bioaccumulation of dioxins in transplanted cultured mussels from an ocean outfall at the Ninety Mile Beach (in Bass Strait) in Victoria was investigated by Haynes et al. (1995). The outfall is situated 1.2 km offshore in 15m of water and at the time was discharging approximately 40 ML/day of secondary treated complex effluent. The effluent consisted of domestic and light industrial waste waters (~16 ML/day), effluent from a bleached kraft pulp and paper mill (~15 ML/d) plus oil and gas production waters from the Bass Strait region (~9ML/d). Mussels were deployed in approximately 16m of water at the outfall, 1, 7 and 10 km down current from the outfall and at a control site 50 km up-current. Mussel dioxin profiles were similar to those in mussels grown in unpolluted seawater and the authors concluded "*tissue concentrations of dioxins and furans in deployed mussels provided no evidence that bioaccumulation of dioxins discharged from the ocean outfall was occurring".*

[14] The biosediment accumulation factor (BSAF) for TCDD is the ratio of contaminant concentration in the organism (normalised for lipid content) to the concentration in dry weight sediment (normalised for organic carbon content). Because it is empirically derived from field data it takes into consideration dioxin accumulation that occurs not only by direct transfer from water or sediment to the organism but also from

The fish's food is the most important exposure source for uptake of dioxins; direct uptake from water via gills and skin is negligible due to very low dioxin water solubility (Gatehouse 2004). According to Gatehouse (2004) benthic feeding fish (bottom dwellers and demersal feeders) consistently contain more dioxins than other fish. The relationship between sediment concentration and fish tissue concentration is therefore more important than the relationship between water column concentration and fish tissue concentration (i.e. the bioaccumulation factor BAF is more important than the bioconcentration factor, BCF). This is consistent with the fact that in polluted waterways around the world virtually all the dioxins partition into sediment.

The relationship between sediment dioxin concentration and fish tissue dioxin concentration is determined from field trials and is called the biota-sediment accumulation factor (BSAF). It is simply the ratio of the concentration in the fish to the concentration in sediment after both media have been normalised for certain factors.

Because BSAFs are based on field data, the values also incorporate the effects of chemical bioavailability and uptake from all segments of the animal's environment; including sediment, water column, and food web. Other factors such as metabolism, depuration, biomagnification, fish growth effects and others are inherently accounted for in the BSAF (Hendricks et al. 1998, Burkhard and Lukasewyez 2000, US EPA 2004, US EPA 1995, Cook and Burkhard 1998). Because the BSAFs are empirically determined they are particularly useful for chemicals, such as dioxins, which may be detectable in fish tissues and sediments but are difficult to detect or measure precisely in the water column. For this reason US EPA (2004) consider the BSAF for dioxins to be a more reliable measure of bioaccumulation potential than bioaccumulation factors (BAFs) and/or bioconcentration factors (BCFs).

BSAFs are specific for the locations from which sediment and fish samples were obtained. In order to make the BSAF applicable to other locations the BSAF needs to be made location specific for dioxin sediment concentration, sediment organic carbon content, and location specific dioxin concentration in fish and the lipid content of the fish.

As a component of the Australian National Dioxins Program, PCDD/PCDFs were measured in sediment and biota from a number of sites around Australia and BSAFs calculated for bivalves and various Australian species of fish (Gatehouse 2004, Müeller 2004). Commercial fishermen supplied fish samples that were caught in close proximity to the sediment sampling locations.

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the food the organism eats and therefore potential biomagnification.

The BSAFs for fish were calculated by dividing the fish tissue TCDD concentrations by the average surface sediment TCDD concentration in the catching vicinity. Gatehouse (2004) normalised the BSAF according to the lipid content of the fish and the organic carbon content of sediment from the locality from which the fish was caught using the data provided in Müeller (2004). All BSAF for dioxins were much less than 1.0. This result is consistent with most fish BSAFs reported in the literature and indicates dioxins are not bioaccumulated to concentrations higher than are in sediment (Gatehouse 2004).

The lack of significant bioconcentration does not however mean fish do not take up dioxins from their environment. In locations where there has been high, point source contamination of sediment some fish have been found to have higher levels of dioxins compared to fish from non-polluted areas, however dioxins do not biomagnify between fish and their food or between fish and sediment (Gatehouse 2004).

Dioxins are considered for a quantitative risk assessment herein because ANZECC (2000) has nominated 2,3,7,8-TCDD as being bioaccumulative (Section 5.2 of HHRA). The ANZECC bioaccumulative opinion for dioxins is not well documented but is apparently founded on "*elevated dioxin levels found in fish, shellfish and sediments in some localised urban and industrial areas in Australia (Thompson et al. 1992) and overseas (US EPA 1984, CCREM 1987, Palmer et al. 1988)".* ANZECC (2000) do not provide discussion of the content of the references they cite nor any specific technical information relating to bioaccumulation of dioxins.

The US EPA, CCREM and Palmer references quoted by ANZECC (2000) are not readily available. However the Thompson et al. (1992) reference relates to *very high local pollution* in Homebush Bay where large amounts (between 200-300 tonnes) of dioxin-contaminated wastes were produced between 1949 and 1976 by industry on the banks of the Bay. Subsequent surveys in the 1980s showed mean concentrations of TCDD varied from <4 to 181 pg/g - wet wt fish depending on species and 29 to 116 pg/g in invertebrates. Surface sediments, down to 50mm depth, contained an average of 9.6±18 pg TCDD/g dry weight sediment. As a result of these surveys, fishing has been banned in Homebush Bay (Rubinstein & Wicklund 1991). These data suggest bioaccumulation of TCDD from high point source contamination of sediment into some types of fish. On face value the data appear at variance to other studies. It is noted however the mean concentration data reported by Thompson et al. (1992) has very large standard errors and it is not possible to determine whether the fish have bioconcentrated dioxins to a greater extent than is present in their environment.

The US EPA (2000) dioxin reassessment makes some general observations in relation to the data it reviewed:

- *"For fish, the concentrations of CDDs and CDFs are dependent on the exposure level, fat content, living habits, and the degree of movement of the species. Comparatively high fat-content bottom fish, collected close to the contaminant source, generally have the highest CDD/CDF levels; whereas, lower fat content, non-stationary fish have much lower concentrations, even in the vicinity of the contaminant source.*
- *The US National Dioxin Study indicated that the levels of 2,3,7,8-TCDD in fish from the Great Lakes Region were higher than those from urban areas. Comparable levels were detected in whole bottom feeders and predators from the Great Lakes Region".* [This latter statement signify's lack of biomagnification of dioxins].

Conclusion:

The information discussed above collectively supports the view that:

- dioxins are not significantly bioaccumulated or biomagnified by fish or bivalves.
- fish living in local environments where dioxin concentrations are low also have low levels of dioxins.

4.3.3 Dioxin concentrations in fish near the effluent discharge

Commensurate with low concentrations of dioxins in the receiving waters background concentrations of dioxins in fish (flathead, wrasse and perch) at the proposed ocean outfall site were less than quantitation limits of 0.1 – 3.4 pg/g tissue wet weight for dioxin congeners grouped according to degree of chlorination. Flathead have the highest average oil content of fish in the area, they are bottom dwelling and feed on benthic organisms, in addition they are the species most likely to be caught by recreational anglers. For these reasons the HHRA calculated the *theoretical* incremental increase in dioxin content of flathead using a BSAF from Australian field studies and Australian data for fish lipid and sediment organic carbon.
The theoretical incremental increase in dioxin concentration of the fish was calculated to be 0.00018 pg TEQ/g, in the context of existing dioxin content of fish being less than quantitation limits of 0.1 – 3.4 pg/g tissue, the incremental intake is not measurable. The change in fish dioxin content would not be measurable even if the incremental increases were 100 – 1000 times higher than estimated.

For reasons outlined above the calculations for dioxin accumulation in the HHRA relate to

flathead. Although flathead make up only 4% of the diet of seals these fish are the species most likely to exhibit dioxin accumulation because of their benthic feeding habit and higher oil content compared to other fish that seals feed upon. Therefore if there is no demonstrable incremental increase in flathead dioxin concentrations it is probable this is also the case for other fish species within seal diet.

Conclusion:

Quite apart from the increase in fish concentration not being measurable because it is far below the capabilities of analytical techniques likely to be brought to the problem, if the anticipated change was measurable it is likely to be within the current background variation in dioxin content of fish in the area. Thus a statistical increase in fish dioxin levels would not be confirmable.

In these situations the logical conclusion is there will be no demonstrable increase in dioxin concentrations of fish that will reside around the ocean outfall.

4.3.4 Potential for bioaccumulation of dioxins by seals

Direct evidence for lack of biomagnification of dioxins through trophic levels of a food web has recently been provided by Wan et al. (2005). In an investigation of a food web in Bohai Bay [15], north China, these workers found lipid concentrations of low chlorinated 2,3,7,8- substituted dioxins and furans did not exhibit significant trends with trophic layers. In fact the concentrations of higher chlorinated dioxins and furans declined significantly with increasing trophic levels. There were however significant positive relationships between trophic layer and accumulation of PCBs. Given the similarity in fat solubility between dioxins and PCBs the authors concluded the difference in trophic transfer (none or little for dioxins but positive for PCBs) is mainly due to their different metabolic transformation rates in the higher trophic levels.

The difference between PCBs and dioxins in bioaccumulation potential is also seen in marine megafauna. Muir et al. (1996) studied beluga whales in the St Lawrence seaway; in males the geometric mean concentration for TCDD TEQ for PCBs was 1070 pg/kg blubber but that for dioxins only 0.1 pg/kg. Other studies in different whales and porpoise reported by Muir et al. (1996) showed similar relationships. Based on the ratio of mirex to TCDD in Lake Ontario and

[15] Bohai Bay is an enclosed inner sea in north China. It is a highly developed economic area and about 1 billion tons of waste water have been discharged into the bay. The study analysed 9 dioxins, 11 furans and 12 PCBs in samples of phytoplankton/seston, zooplankton, 3 invertebrate species, 6 fish species and one marine mammal.

the assumption of similar transport and accumulation properties, the authors considered the TCDD concentration in beluga blubber should be 5 – 6 orders of magnitude (10^5 to 2 x 10^6) times higher than measured.

In relation to seals, the Swedish Dioxin Survey measured concentrations of 17 dioxin and furan congeners in tissues of several fish species, in fish eating birds, and in marine mammals[16] obtained from different sites along the Swedish coast. While there were geographical differences between dioxin/furan levels and patterns in fish, which could be linked to specific point sources of dioxins/furans, and levels were higher in fish eating birds compared to their prey, dioxin concentrations in seals did not indicate biomagnification (de Wit et al. 1992). This conclusion is further supported by a study (Bignert et al. 1989) of four seal species[17] from widely different areas around the Scandinavian Peninsula that are variously impacted (or non-impacted) by anthropogenic sources of dioxins The study showed no substantial species or spatial differences in levels of dioxins/furans. Among marine mammals highly persistent organochlorines (e.g. DDT or PCBs) normally increase with increasing age, this relationship was not observed for dioxin/furan concentrations in seals (Bignert et al. 1989), a phenomenon also seen in other studies (Addison et al. 2005). The results are consistent with expectations for a substance that is not highly bioaccumulative and does not biomagnify, and are explained by rapid metabolism of dioxins/furans by seals.

De Swart et al. (1995) fed two groups of approximately 1 year old harbour seals for two years with fish from two sources. One source had approximately ten fold more TCDD TEQ than the other. While at the end of the two years there was a 3½ difference in blubber concentration between the groups of seals, in both groups the levels of dioxins were lower than in the fish fed to them and estimated body burdens were less than 0.1% of the cumulative intakes of the compounds. These data show dioxins in fish are not readily assimilated by seals and the authors suggest seals may have an efficient mechanism for either excreting or metabolising dioxins.

Blubber of harbour seals from the Strait of Georgia, British Columbia, Canada (in 1991 and 1992) contained higher concentrations of PCDD/F than did samples from Quatsino Sound on western Vancouver Island (Addison et al. 2005, Addison & Ross 2001). Historically the Strait of Georgia received effluent from six coastal pulp mills which used elemental chlorine in the bleaching process with poor effluent control systems, they also used a wood-chip feedstock that

[16] Blubber of grey, harbour and ringed seals.
[17] Grey seal, common seal, harp seal and ringed seal.

had been preserved with pentachlorophenol (Addison & Ross 2001). The Strait of Georgia receives treated and untreated industrial and domestic wastes from surrounding communities. Thus the effluent quality and quantity discharged into the Strait of Georgia was quite different from that to be discharged by the proposed Bell Bay mill. In contrast Quatsino Sound receives effluents from some mining operations and a single sulphite pulp mill which has only used mild chlorine bleaching and a wood feedstock free of PCDD/F precursors. The total dioxin levels in harbour seals in Strait of Georgia were approximately 5 to 8 times those in Quatisino Sound; the latter were 10-30 pg/g blubber lipid wet wt which is comparable with data recorded from uncontaminated sites in eastern Canada, western Europe and the Arctic (Addison et al. 2005). Although still different from the Bell Bay mill processes, which will use no elemental chlorine in its bleaching process, the combination of mine and sulphite pulp mill effluent into Quatisino Sound did not result in increased dioxin concentrations in seals.

As is the case with fish, *relatively high* concentrations of dioxins in a local environment can result in higher concentrations of dioxins in marine mammals compared with those from pristine areas. This does not mean however dioxins in mammals in the polluted area are the result of biomagnification through the food chain.

Conclusions:
The weight of evidence of the studies reviewed above indicates dioxins are not avidly bioaccumulated and are not biomagnified by marine mammals, including seals.

4.3.5 Impact on seals

From the previous section the following conclusions regarding the bioaccumulation of dioxins by marine organisms have been formulated by review of the literature available for this project:

- Dioxins do not magnify through the marine food chain.
- Dioxins are not significantly bioaccumulated or biomagnified by fish.
- Fish living in local environments where dioxin concentrations are low also have low levels of dioxins.
- The theoretical increase in dioxin levels of fish after the effluent outfall becomes operational is markedly less than the analytical capability to measure.
- There will be no demonstrable increase in dioxin concentrations of fish that might reside around the ocean outfall.
- Animals higher in the food chain, including whales and seals, have relatively high

metabolic and/or excretory capacity towards dioxins.

- Seals do not readily assimilate dioxins from their food and do not biomagnify dioxins.

In relation to effluent dioxin release to the ocean, an incremental increase in fish dioxin concentrations will not be demonstrable, and since seals do not easily assimilate dioxins from their food and data indicates dioxins do not biomagnify through the food web to seals, it follows there will be no demonstrable impact on seals from dioxins in the effluent.

The above, coupled with the fact the area of ocean potentially impacted by the discharged effluent is quite small relative to the total range of the fur seals[18], leads to the conclusion the viability of the seal colony at Tenth Island will not be affected by the effluent discharge.

[18] The small relative area potentially affected by the effluent means only a small fraction of the seal diet will be sourced from around the outfall. This will also be the case should fish be attracted to the area of initial dilution (1/100) near the effluent diffuser.

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About Toxikos Pty Ltd

Toxikos Pty Ltd is a consulting company formed on December 1st 2000 to provide clients with independent excellence in toxicology and health based risk assessment. Its charter is to assist industry and government make science based decisions regarding potential effects and management of environmental and occupational chemicals. For over twelve years, prior to and since the establishment of Toxikos, staff have provided toxicology and health risk assessment advice to clients in a wide range of industries and government in Australia, New Zealand and South Africa.

About the author: Dr Roger Drew is one of the Directors and Principal consultants of Toxikos Pty Ltd. He has primary degrees in biochemistry and pharmacology and postgraduate degrees in toxicology. Postdoctoral training was undertaken at the National Institutes of Health, National Cancer Institute in the USA and he spent twelve years teaching medical students and conducting toxicological research at Flinders University of South Australia. He was corporate Toxicologist to ICI/Orica Pty Ltd for twelve years before creating Toxikos Pty Ltd. Dr Drew is the only consultant toxicologist in Australia certified by the American Board of Toxicology.

Dr Drew has been a toxicology consultant to Australian Federal and State Authorities; a member of several standing expert committees of the National Health & Medical Research Council of Australia and the National Occupational Health and Safety Commission of Australia. He has been a member of many expert task groups of the World Health Organization for the International Programme on Chemical Safety.

Dr Drew was Adjunct Professor of Biochemical Toxicology at RMIT University and teaches various aspects of toxicology to undergraduate and postgraduate students at RMIT, Monash and Melbourne Universities. He is a member of several professional toxicology societies and is a recognised national and international expert in toxicology and risk assessment. He is currently on the editorial board of the international scientific journal "Regulatory Toxicology and Pharmacology" and a board member of the Australian National Research Centre for Environmental Toxicology.