

Draft Review of Geotechnical Services reports (2006 – 2007) and recommended on methods to calculate dilution factors, ongoing WET monitoring and appropriateness of using the copepod *Gladioferens imparipes* and Addendum.

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GLOSSARY

Acute – when the duration that an organism is exposed to the toxicant is a small portion of the lifespan of the organism. Typically these are between 24 and 96 hour long exposure. However, for animals with a short life span (e.g. micro-organisms) acute would mean exposures of less than 12 hours.

AF – assessment factor. This is a value that is used to convert one type of toxicity data to another (e.g. an acute EC50 value to a NOEC). These are also called safety factors and adjustment factors.

BurrliOZ – a software package that was used to derive the trigger values for toxicants in the Australian and New Zealand water quality guidelines.

Chronic – when the duration that an organism is exposed to the toxicant is a large portion of the lifespan of the organism. For fish this means longer than 7 days while for micro-organisms a 72 hour exposure is chronic.

DTA – direct toxicity assessment. This is where the toxicity of water bodies, effluents or discharges are determined.

EC5 or IC5 - the concentration of a substance (e.g. saline brine) that will cause a 5% inhibition or reduction in an endpoint (e.g. immobilisation, reproduction) or will cause a 5% effect to an endpoint (e.g. survival).

EC10 or IC10 – the concentration of a substance (e.g. saline brine) that will cause a 10% inhibition or reduction in an endpoint (e.g. immobilisation, reproduction) or will cause a 10% effect to an endpoint (e.g. survival).

EC50 or IC50 – the concentration of a substance (e.g. saline brine) that will cause a 50% inhibition or reduction in an endpoint (e.g. immobilisation, reproduction) or will cause a 50% effect to an endpoint (e.g. survival).

LEPA – low environmental protection area. The level of protection to be provided in this area is 80% of species. In other words a PC80 applies in this area.

LOEC – the lowest observed effect concentration. This is the lowest concentration used in a toxicity test that causes an effect that is significantly different to the control.

Margin of safety – the difference between the one value and another expressed as a multiple of the smaller value. For example the margin of safety between values of 5 and 50 is ten.

MEPA – medium environmental protection area. The level of protection to be provided in this area is 90% of species. In other words a PC90 applies in this area.

NOEC – the no observed effect concentration. This is the highest concentration used in a toxicity test that causes an effect that is not significantly different to the control.

PC – protective concentration. The concentration that should prevent a selected percentage of species in the ecosystem being considered from experiencing toxic effects. Thus PC99, PC95, PC90 and PC80 values protect 99, 95, 90 and 80% of species in the ecosystem being considered.

Safety factor – see margin of safety.

Sub-chronic – when the duration that an organism is exposed to the toxicant is a small portion of the lifespan of the organism but early life-stages which are very much more sensitive than later life-stages, are exposed to the toxicant. Such tests are considered to be estimates of chronic exposure.

WET – whole effluent toxicity testing. Toxicity testing using effluents.

EXECUTIVE SUMMARY

Whole effluent toxicity (WET) testing was conducted over three years in order to assess the toxicity of the saline brine discharged from the Perth Seawater Desalination Plant. WET tests were conducted using a fish, a microalga, a macroalga, a mussel and a copepod (a type of crustacean). There are limitations associated with the copepod toxicity data. The preferred approach of deriving the dilution factors was to use an estimate of the concentration that would cause a 10% reduction in copepod growth (i.e. Estimated EC10) value for the copepod (i.e. The concentration that would cause a 50% reduction in copepod growth (EC50) from the copepod 24 hour pulse exposure tests divided by a conversion factor of five) with the EC10 or toxicity data from the other species. This resulted in dilution factors of 16 and 13 to meet the limits set for the high ecological protection area and the moderate ecological protection area respectively. According to information provided, these dilution factors are being met with a considerable margin of safety. The recommended dilution factors were ground truthed against the toxicity data for the test species and they provided more than adequate protection.

Given the available information I believe that a decision on whether further routine WET testing is necessary is a matter to be resolved between the Water Corporation and the WA Department of Environment and Conservation. Further WET testing would be appropriate if any major changes to the Perth Seawater Desalination Plant occurred or if the chemical additives used were modified as these could lead to changes in toxicity of the saline brine.

I believe that the copepod *G. imparipes* is an appropriate test species to be used for WET testing in Cockburn Sound for the reasons provided previously in Warne (2007). However, I believe it would be wise to conduct additional *G. imparipes* toxicity tests to determine if more reliable EC10 estimates can be obtained. If this succeeds then I believe that *G. imparipes* EC10 data should be used to derive dilution factors, rather than converted EC50 data. If it is not successful and the limitations of the EC10 can not be resolved, then I recommend that *G. imparipes* still be used and that the EC50 values be divided by a conversion factor of five. The resulting value should be combined with the EC10 data from the other species to derive dilution factors. Alternatively a new WET test could be developed to replace *G. imparipes* however, there is no guarantee of success and this could involve considerable development costs and time delays.

SUMMARY OF ADDENDUM

Findings of a draft copy of this report (minus the Addendum) were presented to representatives of the WA Department of the Environment and Conservation at a meeting. During which it was pointed out that the Environmental Management Plan required the reporting of EC/IC5, EC/IC10 and NOEC values and the calculation of protective concentrations and dilution factors for each of these sets of data. This work was undertaken by Geotechnical Services. The author of the current report recommended some changes be made to the data used to calculate the dilution factors for the 2006 WET testing. These resulted in some changes to the resulting dilution factors. The dilution factors calculated using the 2007 WET testing data are correct.

It is appropriate to use EC/IC10 and NOEC data to derive water quality guidelines and dilution factors. The author of the current report does not support the use of EC/IC5 data to derive water quality guidelines and dilution factors, as there is too much variability in this type of data. The dilution factors derived using the EC/IC10 and NOEC data were well below the dilution that is achieved at the Perth Seawater Desalination Plant, with a reasonable margin of safety. All except the dilution factor that corresponds to protecting 99% of species that were derived using the EC/IC5 data were below the dilution achieved by the Perth Seawater Desalination Plant with a reasonable margin of safety. The dilution factor corresponding to protecting 99% of species that was calculated using EC/IC5 data was only marginally larger than the achieved dilution at the Perth Seawater Desalination Plant.

BACKGROUND

In the latter part of 2007 Dr Michael Warne (CSIRO) was approached by Ms Suzanne Brown to act as an expert advisor to the Water Corporation (Western Australia) on matters relating to whole effluent toxicity (WET) testing and the derivation of suitable dilution factors for saline brine discharges associated with the desalination plants being developed in Western Australia. He was subsequently approached by Mr Gordon Groth to provide advice on a report provided to Water Corporation by Geotechnical Services Pty. Ltd. (Woodworth, 2007). This report (Warne, 2007) reviewed the work done on copepod toxicity testing.

In January, 2008 Dr Warne was again approached by Mr Gordon Groth regarding three reports on the WET testing conducted by Geotechnical Services (2006a, 2006b, 2007). Specifically Dr Warne was asked to:

- comment on the appropriateness of the method used to derive dilution factors and if appropriate to recommend an alternate method and dilution factors;
- clarify the issue of representivity of species;
- comment on whether any further WET testing was required; and
- comment on the appropriateness of continuing the use of the copepod WET test.

CALCULATION OF DILUTION FACTORS

The report (Geotechnical Services, 2007) calculates the dilution factors that corresponded to the concentrations of saline brine that should protect 80% (PC80) and 90% (PC90) of species. In doing this the 2007 WET test data were used. I believe it is appropriate that separate dilution factors were derived based on the 2006 and 2007 WET testing results. The 2007 data used were either sub-chronic EC10 or NOEC values apart for the acute EC10 value for *Gladioferens imparipes*. It is not ideal to combine acute and chronic toxicity when using species sensitivity distribution methods such as BurliOZ (Campell et al., 2000) as they may have different distributions. However, based on the modelling of Yeates (2006) the 24 hour pulse exposure test for the planktonic *G. imparipes* will overestimate the likely life-long exposure of copepods in Cockburn Sound by more than 4 fold (i.e. 24 hours compared to 5 hours). Therefore from an environmental realism point of view it would be more appropriate to use an acute test such as that conducted rather than a chronic test.

Therefore one faces the situation of either using *G. imparipes* toxicity data from an acute but environmentally realistic test or use chronic data that is not environmentally relevant. Neither option is optimal.

There is also another issue with the *G. imparipes* toxicity data – the fact that the EC10 are not reliable. Invariably if toxicity data is plotted against the duration of the exposure then an asymptotic relationship occurs (i.e. the plot at both small and long exposure duration times will approach but never reach the axes). Such a plot was obtained for the EC50 data but not for the EC10. This raises the issue of whether the less reliable but more environmentally conservative EC10 values or the more reliable but less conservative EC50 values should be used to derive dilution factors. Again neither option is ideal. The Australian and New Zealand water quality guidelines (ANZECC and ARMCANZ, 2000) provided a means of resolving this. They recommend that a conversion factor of five is used to convert EC50 data to NOEC data (ANZECC and ARMCANZ, 2000; Warne 2001). This same factor can be used to convert EC50 to EC10 data as the NOEC is considered to be equivalent to the EC10 (Moore and Caux, 1997; USEPA, 1991 and Hoekstra and Van Ewijk, 1993).

Given the above I calculated dilution factors for four different scenarios:

1. using the 2007 sub-chronic EC10 values for the microalga, the macroalga, the mussel, and the fish and the acute EC10 value for the copepod;
2. using the same dataset as scenario 1 except the value for the copepod was the acute EC50; and
3. using the same dataset as scenario 1 except that the copepod value was omitted.
4. using the same dataset as scenario 1 but the copepod acute EC50 had been divided by an assessment factor of 5.

Comparison of the dilution factors generated by scenarios 1 and 2 would reveal the effect of using EC10 or EC50 values for *G. imparipes*. Comparison of scenario 3 with 1 and 2 would reveal the effect of using the acute *G. imparipes* toxicity values. Comparison of scenario 2 and 4 would show the effect of using the conversion factor and comparison of 1 and 4 would show the difference between using the unreliable EC10 values and the estimated EC10 value. The data that were used are presented in Table 1.

Table 1. Toxicity data (expressed in terms of % saline brine) used to derive dilution factors for three different scenarios.

Organism and measure of toxicity	Scenario 1	Scenario 2	Scenario 3	Scenario 4
Microalga (EC10)	13.7	13.7	13.7	13.7
Macroalga (EC10)	92.9	92.9	92.9	92.9
Mussel (NOEC)	12.5	12.5	12.5	12.5
Copepod (EC10)	16.8	29.8		5.9
Fish (EC10)	9.6	9.6	9.6	9.6

The resulting dilution factors generated by the four different scenarios are presented in Table 2.

Table 2. The dilution factors generated using the data in the four scenarios presented in Table 1.

Scenario	Dilution factors	
	PC90	PC80
1	10.8	9.2
2	10.8	8.8
3	11.7	9.6
4	16	12.9

It is clear that the dilution factors that correspond to the same level of protection (i.e. PC90 or PC80) are essentially the same for scenarios 1 to 3. The dilution factors derived using the estimated EC10 value (scenario 4) are considerably higher than the other dilution factors, reflecting the fact that the estimated EC10 value is considerably smaller than the other toxicity values.

None of the above methods of deriving the dilution factors are optimal. The limitations with scenario 1 and 2 have already been discussed. The limitation of scenario 3 is that it only uses toxicity data for four species that belong to four taxonomically different groups and therefore it does not meet the minimum data requirements of the BurrliOZ method (ANZECC and ARMCANZ, 2000; Warne, 2001). The limitation of scenario 4 is that it uses an estimate of the EC10 value. From a regulatory point of view the datasets that use reasonable assumptions and yet yield the most conservative dilution factors are preferred. Given this, the most appropriate approach would be to use the estimated acute *G. imparipes* EC10 value (i.e. the EC50 value divided by five) and the most appropriate dilution factors to adopt are 16 and 13 for the PC90 and PC80 respectively.

The diffuser for the Perth Seawater Desalination Plant was designed to achieve a dilution factor of 45 within 50 m of the diffuser and has always delivered at least this level of dilution

(Gordon Groth, *pers. comm.*). Assuming that the above dilution factors are achieved then the levels of protection established for the low ecological protection area (LEPA – PC80) and moderate ecological protection area (MEPA – PC90) will be met with a reasonable margin of safety. For example, if a 45 fold dilution was achieved the resulting water would be approximately 2.2% saline brine while the maximum permitted by the PC80 is 7.8% saline brine (i.e. 100/13.9 dilution factor) and the maximum permitted by the PC90 is 6.25% saline brine (i.e. 100/16 dilution factor).

The last step in the derivation of the Australian and New Zealand water quality guidelines was to ground-truth the trigger values (Warne, 2001) and if necessary to adjust the trigger values downwards by manipulating the calculations or the data in various ways (e.g. by increasing the level of protection from 95% to 99% or using a larger assessment factor¹ (AF)). This was done by comparing each trigger value to all the toxicity data that was available for that chemical. Particular attention was paid to comparisons of trigger values and field-based, mesocosm or microcosm toxicity data – as this type of toxicity data has the greatest environmental relevance. This ground truthing was also conducted for this report.

The PC80 values calculated for the four scenarios were compared to the toxicity data used to generate them. In scenarios 1 - 3 the PC80 was larger than the toxicity value for the fish *Pagrus aratus*. Thus, for scenarios 1 and 2, 20% of the test organisms (one out of the five species) would suffer a toxic effect which is exactly what would be expected from a PC80. For scenario 3, 25% of the test organisms (one out of the four species) would suffer a toxic effect which is slightly more than what would be expected from a PC80. However, in scenario four the PC80 was lower than the EC10 values for all the test species – thus less than 20% of the species tested would suffer a toxic effect at the PC80. Another ground truthing relates to the diffuser. If a 45 fold dilution is achieved by the diffuser then the resulting water would be 2.2% saline brine. There would thus be a 2.6 - fold safety factor between the dilution achieved and the lowest toxicity value used to derive the dilution factors (i.e. 5.9).

While the method used in the Geotechnical Services report (Geotechnical Services, 2007) is sound, I believe that converting the EC50 value to an estimated EC10 is more rigorous and defensible and also results in more conservative dilution factors. Provided the outfall diffuser is providing approximately a 45 fold dilution then there will be a reasonable margin of safety between the percentage of brine in the receiving water outside the mixing zone and the percentages that will exceed the PC80.

¹ Assessment factors is the term used in the Australian and New Zealand water quality guidelines to describe factors used to adjust data. These are also called safety factors and adjustment factors.

Given the limitations of the *G. imparipes* toxicity data (discussed above) I believe that the best way to determine dilution factors is to divide the acute *G. imparipes* EC50 value by five and then combining the resulting value with the sub-chronic EC10 values for the other species to calculate dilution factors.

THE ISSUE OF SPECIES REPRESENTIVITY

The issue of representivity of the test species is treated differently in WET testing and in deriving dilution factors. There are two main approaches to WET testing

1. **to use generic species** that occur in that environmental media. For example, a WET test at Cockburn Sound would use species that occur within Australian marine waters. This is also sometimes called the Standard DTA approach (Van Dam and Chapman, 2001).
2. **to use endemic organisms** that actually occur in the ecosystem that is being assessed. For example, a WET test at Cockburn Sound would use species that are found in the marine waters around Cockburn Sound or closely related organisms (i.e. they are representative of the ecosystem being studied). This is also sometimes called the Site-specific DTA approach (Van Dam and Chapman, 2001).

There are strengths and limitations to both approaches. The main advantage of the endemic species approach is that the toxicity data generated are directly relevant to the particular ecosystem being studied.

I believe that it is generally accepted within ecotoxicology that the endemic species approach is the preferred approach. Van Dam and Chapman (2001) state

“For the purposes of Australian water managers, who generally oversee specific geographical regions and are concerned with local water quality, site-specific DTA is likely to be the most appropriate approach.”

This is the certainly the approach recommended for conducting DTA by the Australian and New Zealand guidelines for marine and fresh water quality (ANZECC and ARMCANZ, 2000).

The endemic species approach has been adopted for the Cockburn Sound study.

The BurrliOZ method (Campbell et al., 2000) is an example of a species sensitivity distribution (SSD) method. The minimum data requirements for this particular method are toxicity data for at least five species that belong to at least four taxonomic groups (Warne, 2001). There are no requirements for any particular organism type to be included in the dataset used by BurrliOZ. The reason for this is that BurrliOZ and other SSD methods are statistical methods. What they do is select the distribution that best fits the toxicity data and then using that distribution they calculate the concentration that should protect a chosen percentage of species. The reason these methods require toxicity data from a range of organism types is to make the data as representative as possible of the types of organisms that are present. For example, if toxicity data for five fish species was used BurrliOZ could only calculate the concentration that should protect a chosen percentage of fish species – not a certain percentage of all species. Within the context of using a SSD method each data point is simply representative of the sensitivity of all species within the ecosystem being examined.

WET MONITORING

For the majority of the test species there was no difference in the toxicity measured during the 2006 and 2007 WET testing. However, it is also clear that the EC50 and EC10 results for *Ecklonia radiata* were different – with the toxicity decreasing markedly in the second year. The EC10 value of the microalga *Isochrysis galbana* also showed a marked change in toxicity – in this case an increase. There are a number of possible explanations for these changes:

1. the *E. radiata* and *I. galbana* used in 2006 and 2007 had different sensitivities;
2. the toxicity of the saline brine changed; or
3. the change in toxicity was due to a co-contaminant from another release into Cockburn Sound.

The reference toxicant test values showed that the sensitivity of *E. radiata* and *I. galbana* to the reference toxicant had not changed. This does not necessarily mean that the sensitivity had not changed to other toxicants, but it does mean it is less likely. The presence of co-contaminants from other sources that affect the measured toxicity of the saline brine is certainly possible. For example, earlier in the WET testing program for the Perth Seawater Desalination Plant a whole set of WET tests were unusable due to the occurrence of a contaminant not related to the Desalination plant. Therefore, it is not possible to fully resolve the cause of this change in toxicity.

It should be noted though that despite the changes in the toxicity of the saline brine the dilution factors calculated using the four species excluding the copepod were essentially the same. The 2006 dilution factors that corresponded to the PC90 and PC80 were 10.4 and 8.6 respectively. The corresponding dilution factors for the 2007 data were 11.7 and 9.6. There were essentially no differences because the increased toxicity to *I. galbana* in 2007 was essentially countered by the decreased toxicity to *E. radiata*. The copepod toxicity data could not be used in these comparisons as the duration of the test exposures was so markedly different between the two years.

It is my experience that most licensing agreements that cover the discharge of effluents require ongoing monitoring of some sort – be it chemical or biological or a combination of the two. Often the frequency of testing and the number of tests conducted is initially higher and then tapers off if it becomes clear that the toxicity or chemistry is stable.

Given the continuous nature of the desalination process, the margin of safety between the dilution being achieved and that needed to meet the MEPA, and the fact that similar dilution factors were obtained using the 2006 and 2007 toxicity data I believe that a decision on whether further routine WET testing is necessary is a matter to be resolved between the Water Corporation and the WA Department of Environment and Conservation. It would, however, certainly be appropriate to conduct further monitoring if there were any major changes to the Perth Seawater Desalination Plant or if the chemical additives used were modified as these could lead to changes in toxicity of the saline brine. As toxicity identification and evaluation studies (TIE) have not been conducted, the contribution of the various chemicals in the brine to the toxicity is not known. Thus, chemical analysis would not be an adequate system for determining if changes in toxicity were likely. Therefore if any further monitoring is to be done I believe that WET testing is the preferred method. The number of WET tests could be reduced in further testing, but if this was done the most sensitive species should be used.

ONGOING USE OF *G. IMPARIPES*

I believe that the copepod *G. imparipes* is an appropriate test species to be used for WET testing in Cockburn Sound for the reasons provided previously in Warne (2007). However, as stated above there are limitations to the EC10 data currently generated by this WET test. I believe that the best way forward would be to conduct additional *G. imparipes* toxicity tests using lower concentrations of the saline brine to determine if more reliable EC10 estimates can be obtained and to determine if this improves the relationship between EC10 values and

duration of exposure. If this succeeds then I believe that *G. imparipes* EC10 data should be used to derive dilution factors, rather than converted EC50 data. If it is not successful and the limitations of the EC10 can not be resolved, then I recommend that *G. imparipes* still be used and that the EC50 values be divided by a assessment factor of five. The resulting value should be combined with the EC10 data from the other species to derive dilution factors. Alternatively a new WET test could be developed to replace *G. imparipes* however, there is no guarantee of success and this could involve considerable development costs and time delays.

ADDENDUM

Background

The outcomes and recommendations of a draft copy of the above draft report were presented to representatives of the WA Department of Environment and Conservation. During the meeting the DEC advised that they would like the WET testing report to cover NOECs, EC/IC5 and EC/IC10 values and identify the protective concentration values (PC99, PC95, PC90 and PC80) and associated dilution factors for each set of toxicity data. This is consistent with the Environmental Management Plan for WET Testing Brine, 23 November 2006" established by the Water Corporation.

Geotechnical Services (2008) calculated the EC/IC5 values and calculated PC values and dilution factors for each set of toxicity data. The purpose of this addendum is to provide an independent review of the calculations and to comment on the various dilution factors that were derived.

Methods used to calculate PC values

EC50 values were available for all the test species for both sets of WET tests (i.e. 2006 and 2007). For the reasons mentioned in the above report EC10 and EC5 values for the copepod are not reliable and thus were not used. Rather, as was recommended above, estimates of the EC10 were obtained by dividing the EC50 by an assessment factor of five. Geotechnical Services (2008) suggested that EC5 values could be estimated by dividing EC50 values by an assessment factor of 10. They attributed this method to the Australian and New Zealand water quality guidelines (ANZECC and ARMCANZ, 2000) and to Warne (2001). However, whilst this attribution is not correct, it is a reasonable approach to adopt.

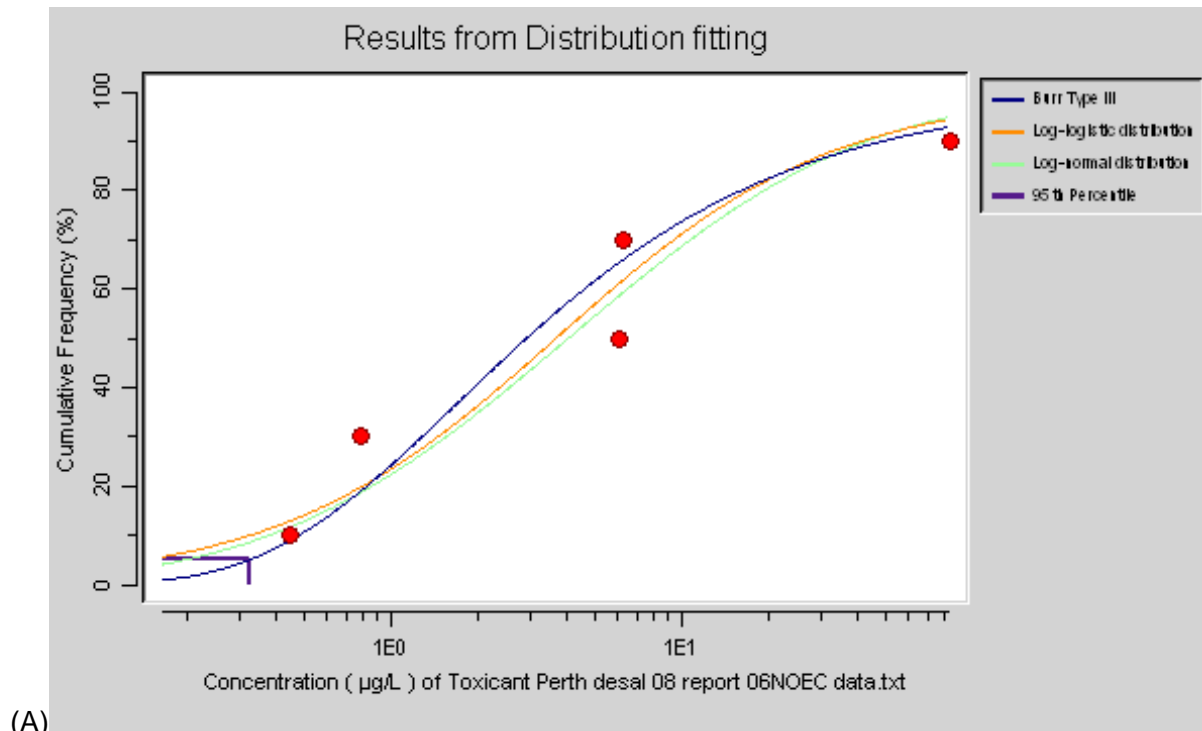
2006 WET data

I agree with all the toxicity values that were selected to calculate the PC values and dilution factors for the 2006 WET testing that are presented in Table 3 of Geotechnical Services (2008), with one exception. The selected NOEC value for the fish is 11.0% saline brine. However, this is not the actual NOEC value, rather it is the EC10 value. The EC10 value was adopted as the NOEC on the basis that EC10 are often considered the same as a NOEC (Moore and Caux, 1997; USEPA, 1991; Hoekstra and Van Ewijk, 1993). However, in this instance there is a NOEC value (<0.78 % saline brine). As the NOEC value is less than the EC10 value it would be more conservative (i.e. environmentally protective) to use the actual NOEC value rounded off to 0.78 % saline. A similar procedure of adopting the estimated EC10 value (i.e. $EC_{50} \div 5$) rather than using an actual NOEC value was also done for the copepod reproduction data. However, in this instance the estimated EC10 value (i.e. 0.45 % saline brine) was smaller and thus more conservative than the NOEC value (i.e. < 0.78). Therefore, this is a reasonable approach to adopt in this case.

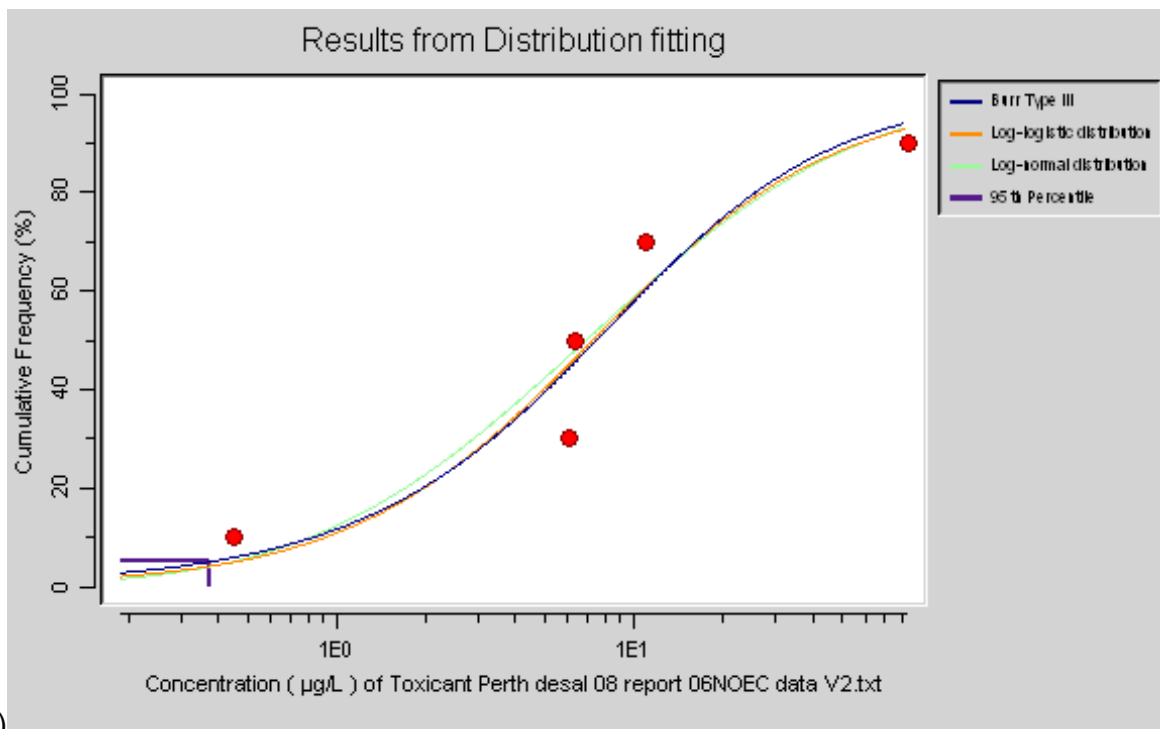
The recommended change to the toxicity value to be used in the analysis of the NOEC data for the fish was made and the PC values and corresponding dilution factors calculated. These results and those obtained by Geotechnical Services (2008) are presented in Table 3.

Protection level (% species)	Calculated by Geotechnical Services (2008)		Calculated in the current study	
	Protective Concentration (% saline brine)	Dilution Factor	Protective Concentration (% saline brine)	Dilution Factor
99	0.06	1666	0.17	590
95	0.37	270	0.33	300
90	0.84	119	0.48	210
80	1.95	51	0.83	120

There is a considerable change in the dilution factors that are required for the various protection levels compared to the values obtained by Geotechnical Services (2008). At high levels of protection (e.g. PC99) the dilution factor required is considerably reduced, but at all lower levels protection (e.g. PC95, PC90, PC80) the dilution factors actually are larger. This reflects the change in the distribution that these changes made (Figure 1). Thus at the LEPA where the PC80 must be met, the dilution factor would need to be 120 fold. While at the MEPA where the PC90 must be met, the dilution factor would need to be 210 fold.



(A)



(B)

Figure 1. The graphical output of the BurrliOZ software for the dataset recommended by the current study (A) and used by geotechnical services (2008) (B).

However, to a large degree the above change is academic in nature as it is the dilution factors from the 2007 WET testing using the saline brine discharged by the Perth Seawater Desalination Plant that is important in terms of the EIS for the plant.

In general, the magnitude of the dilution factors decreases from using EC5 data, to EC10 data to NOEC data. However, as the level of protection decreases (i.e. PC99 to PC80) the differences between the dilution factors for each set of data decrease. In fact, there is little difference in the dilution factors required to meet the PC80 that were generated by the various datasets.

2007 WET data

I agree with the vast majority of the data selected by Geotechnical Services (2008) to calculate the PC values and dilution factors. The EC10 value for the copepod has been rounded off incorrectly. The value is 5.96 which should be rounded off (to one decimal place) to 6.0, not 5.9. There is also a potential problem with the values for the mussel used for the EC10 and EC5. In both cases the NOEC value of 12.5 % saline brine has been used. However, the EC50 value could be divided by assessment factors of five and ten to provide estimates of the EC10 and EC5 respectively. However, in this case I support the approach that has been adopted by Geotechnical Services (2008); because the NOEC value of 12.5 % saline brine exerts no effect on the mussel (refer to page 57, Geotechnical Services, 2008). The estimated EC10 and EC5 values would be 3.6 and 1.8 % saline brine. However, as 12.5% saline brine exerts no detrimental effect on the mussel, it would be unnecessarily conservative to use these estimated EC10 and EC5 values. I therefore agree with the PC values and dilution factors presented in Geotechnical Services (2008).

Differences between the dilution factors derived using '06 and '07 data

The differences in the magnitude of the dilution factors derived using the 2006 and 2007 WET toxicity data are quite large and warrant explanation. These differences occur predominantly because different copepod toxicity data is used in the two sets of data. Warne (2007) argued that the 24-hour continuous exposure of the copepod to the saline brine was a gross overestimation of the likely life-long exposure of copepods in Cockburn Sound. He estimated, based on data provided by Yeats et al. (2006), that the average life-long exposure of copepods to the plume of saline brine would be approximately 5 hours (Warne, 2007). He therefore argued that pulse experiments where the duration of exposure much more closely

relates to the likely exposure should be used to estimate the toxicity of the saline brine. Hence for the 2007 round of WET testing a 24-hour exposure period was used.

Appropriateness of using EC5, EC10 and NOEC data to derive PC values and dilution factors

The current Australian and New Zealand Water Quality Guidelines use chronic no observed effect concentration (NOEC) data to derive high reliability Trigger Values (TVs) but acute EC/IC50 toxicity data to derive moderate and both classes of low reliability TVs (ANZECC & ARMCANZ, 2000; Warne, 2001). There has been discussion in the literature over the relative merits of NOEC and LOEC toxicity data (collectively referred to as hypothesis-based toxicity data) compared point estimate toxicity data (e.g. EC50, IC20 values). Critics such as Hoekstra and Van Ewijk (1993), Noppert et al. (1994) and Chapman et al. (1996) feel that it is not appropriate to use such data for regulatory purposes. They would prefer to use toxicity values that correspond to a fixed biological effect (e.g. an LC5 or EC10). Problems with the use of NOEC and LOEC data revolve around the fact that these values are determined using hypothesis based statistical techniques. Specifically the problems are that:

- only tested concentrations can be NOEC values (therefore NOEC values are to a large degree affected by the concentrations used in the toxicity test);
- the NOEC title is misleading. A NOEC is the highest concentration used in a toxicity test that causes an effect not significantly different to the control(s). It therefore does not correspond to 'no effect'. Typically, the NOEC corresponds to a 10 to 30% effect (Moore and Caux, 1997; USEPA, 1991 and Hoekstra and Van Ewijk, 1993);
- this measure of toxicity can easily be manipulated and does not encourage high quality work. For instance, lax procedures would increase the variability between replicates. This in turn, would increase the size of the difference needed between the treatment and control means in order for a statistically significant difference to be found (i.e. the NOEC value is likely to increase).
- a problem related to the third dot point is that TVs derived using this data do not have as clear a definition as those derived using EC10 data. The TVs based on NOECs would theoretically protect X% of species from experiencing statistically significant inhibitory impacts. The TVs based on EC10 data would theoretically protect X% of species from experiencing inhibitory impacts greater than 10%.

Despite the above problems NOEC data were recommended in preference to point estimates of toxicity such as the EC10 in the Australian and New Zealand guidelines (ANZECC and ARMCANZ, 2000) for the following reasons:

- there was a general lack of EC10 type data in the scientific literature;
- there are large amounts of NOEC data available in the literature; and
- this situation is unlikely to change in the near future until an authoritative organization (e.g. USEPA, OECD, EU) declares that NOEC/LOEC data should no longer be used and make a statement about what new form of data should be used.

However, it was pointed out in the WQGs (ANZECC & ARMCANZ, 2000) and in the document that proposed the framework for developing the WQGs (Warne, 1998) that the methods used to derive the trigger values are not data specific as long as only one type of data is used. Thus, TVs could be derived using EC10 values if there was sufficient data. In fact, these same documents suggested that the use of NOEC data “be phased out” as EC10 type data become available (ANZECC & ARMCANZ, 2000; Warne, 1998).

There are essentially an infinite number of point estimates of toxicity. However, the main ones reported in the literature are those that measure a 50%, 20% and 10% effect. Other values that measure a 5% and 1% are occasionally reported. When relationships between toxic effects and concentration are drawn up they are typically sigmoidal. The error associated with estimating effects of various magnitudes becomes larger at the extremes (i.e. 0 and 100%) and smaller towards the median (i.e. a 50% effect). This is why most toxicity data report a 50% effect concentration. However, basing water quality guidelines or dilution factors on toxicity data that permit a 50% effect is not particularly protective. But as lower % effects are used the uncertainty at which concentration effects actually occur increases. Therefore, the final choice of the appropriate toxicity data to use to derive water quality guidelines and dilution factors will be a compromise between these two factors. The general consensus appears to be that EC/IC10 or EC/IC20 values are the most appropriate. I have not seen any EC/IC5 data being used for this purpose, in fact very little EC/IC5 toxicity data is ever reported in the literature. I therefore agree with Chapman (2005) that EC/IC5 data are inappropriate to derive water quality guidelines and dilution factors.

Comparison of the dilution factors and the dilution attainable by the Perth Seawater Desalination Plant

The highest dilution factors (i.e. those that correspond to protecting 99% of species) calculated by Geotechnical Services (2008) for the 2007 WET testing using EC10 and NOEC

data are all smaller than the dilution achieved by the Perth Seawater Desalination Plant. Okley et al. (2007) reported that the dilution factor achieved was always greater than 50 fold. Only the dilution factor corresponding to protecting 99% of species based on EC5 data, exceeded the dilution factor reported by Okley et al. (2007) and only very marginally (i.e. 50.5 versus 50). But as stated above it is not appropriate to use EC/IC5 data to derive dilution factors.

The dilution factors that correspond to the level of protection to be met in the LEPA and MEPA are all well below the dilution achieved by the Perth Seawater Desalination Plant. The smallest margin of safety is approximately 1.8 fold and this is based on the dilution factor calculated using EC/IC5 data. When the dilution factors calculated using EC/IC10 and NOEC data are compared to the obtained dilution the smallest safety margin is approximately 4.6 fold.

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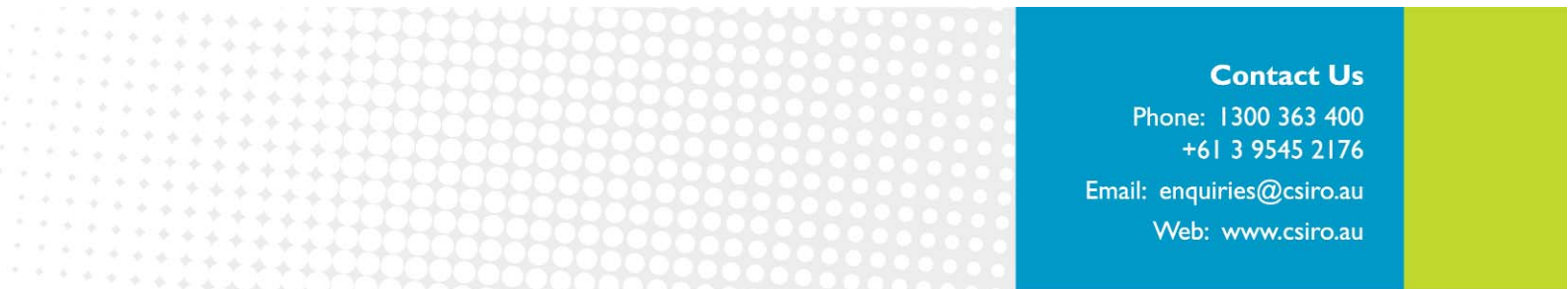
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