METALS ENVIRONMENTAL RISK ASSESSMENT GUIDANCE

FACT SHEET

MERAG 04

MARINE RISK ASSESSMENT: USE OF FRESHWATER DATA FOR THE DERIVATION OF ECOTOXICITY THRESHOLDS FOR MARINE SPECIES

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1. INTRODUCTION

The main focus of most risk assessment methodologies and Environmental Quality standards setting has been on the inland environment considered potentially at risk from urban or industrial development. Only recently it has been recognized by several regulatory bodies that there is need to extent these principles to assess the potential risks of a substance entering into the marine environment with the main focus on estuaries and coastal zones (OSPAR, 1998, TGD, 2003) Since marine ecosystems are a part of the largest aquatic system on the planet, covering over 70% of the Earth's surface including habitats ranging from the productive near shore regions to the barren ocean floor the extension of the existing risk assessment approaches to cover the marine environment seems to be a logical development. However, it should be acknowledged that the physico-chemical characteristics of saltwater environments show important differences compared to freshwater environments. For example, seawater is characterized by a higher ionic strength and the observed gradients in abiotic factors such as chlorine content have important consequences on composition, behaviour, physiology, reproductive strategies of species on one hand and could have consequences for the speciation and bioavailability of metals on the other hand. (Wright 1995, Verslycke et al. 2003, Nriagu 1980). Consequently, marine risk assessments should use, where possible, data relevant to the marine environment that is considered.

For the exposure assessment there are no specific aspects to be treated differently in the marine assessment. Processes are by large the same as in the freshwater, but input parameters need to be specific to the marine environment and should not be cross-read without a proper justification. The effects assessment should ideally be based upon data generated using a range of ecologically relevant saltwater species. However, due to the paucity of marine data for some metals it could be worthwhile to explore on a case-by-case basis if freshwater data can be used in absence of relevant marine data for some taxa. Therefore, a clear understanding is needed with regard to the speciation, bioavailability or possible species sensitivity differences between freshwater and saltwater environments for the metal/metal compound of concern. In the next paragraphs, dissimilarities/similarities between marine/estuarine and freshwater ecosystems that could possibly influence the impact of metals and metal compounds are explored.

2. CHARACTERISTICS OF THE MARINE ENVIRONMENT

2.1 Abiotic factors

Important physico-chemical differences exist between freshwater and saltwater systems. Seawater is characterized by a higher ionic strength and a relatively constant inorganic composition, with generally a smaller temperature range and a more constant pH (typically 7.8-8.2) and hardness than the freshwater environment. The most important factors influencing metal speciation in the marine environment are salinity (Cl^-), DOC, pH, ionic strength and hardness.

In open seas and oceans, the pH, salinity, hardness and ionic strength are fairly constant. Coastal or intertidal zones are typically characterized by a higher turbidity and more gradients with regard to the parameters mentioned above. Typically, DOC concentrations may be very variable in these areas. Estuaries are even more fluctuating systems with strong gradients of turbidity, pH, and salt concentrations. A number of marine environments have very specific conditions, deviating from these more general rules (the Baltic Sea with many gradients, highly productive coral reefs, submarine volcanoes with extreme temperatures…). However, the general nature of the enclosed and semi-enclosed seas is essentially dependent on whether or not the fresh water lost through evaporation is more or less than the amount of freshwater input from precipitation and direct runoff from the land. Another important physical feature is the retention time, most relevant to (semi-)enclosed seas.

These variations in physico-chemical parameters can have a pronounced influence on the speciation of metals. Further to chlorides, suspended solids, manganese, sulfur and oxygen dominate trace metal distribution/speciation, also in saltwater. The ionic composition of seawater has also a considerable influence on both solubility and partition coefficients. In absence of marine specific data there might be sufficient information available on a case-by-case basis to allow the relevant partition coefficient in seawater to be calculated from the freshwater data. Otherwise, measurements under marine conditions may be necessary. For charged chemicals, including many trace metals and their species, models indicate a general reduction in partitioning from river to sea, despite an accompanying increase in pH of about 1-2 units, because of competitive adsorption and complexation with seawater ions.

Decreasing toxicity with increasing salinity, probably due to a decrease in bioavailability, has been observed by a number of authors (e.g., Sundae et al, 1978, Blust et al, 1992; Verslycke et al. 2003). Although each metal may react differently to increases in salt concentrations, free ion concentrations of for instance cadmium, zinc, copper and nickel tend to decrease with increasing salt concentrations, most probably by complexation with (chloride) ions (Nriagu 1980, Sadiq 1992, Verslycke et al. 2003, Wright 1995). Hence, the free metal ion, which is most bioavailable, is more abundant at lower salinities because of the reduced formation of chloro-complexes (e.g. $AgCl$, $CdCl_2$, $CoCl_2$). Other metals, however, are less influenced by salt concentration. Lead for instance, is mostly precipitated as lead(II)chloride at the normal pH of seawater and is less influenced by salinity. Further, the bioavailability of some metals (e.g. copper) is also strongly influenced by organic ligands, and this effect may sometimes override the effect of salinity (Verslycke et al. 2003).

As a general rule, total/dissolved metal concentrations in marine environments should be 'normalized to sea-values'. That is, the bioavailable fraction of the metal considered should be calculated for the saltwater environment under consideration. It is theoretically possible to run for instance a BLM for the specific saltwater conditions. However, up till now, no validated chronic BLMs for marine environments exist (Niyogi and Wood 2004). Arnold et al (2005) developed a BLM model for copper using the bivalve *Mytilus sp*.(the most sensitive taxa in the US EPA saltwater copper criteria database. From the results it is clear that the BLM predictions are to a high degree related to the presence of DOC and the BLM model could in fact be interpreted as a kind of DOC regression model. For open seas, little gradients exist (except with depth). For coastal/estuarine regions, very much variation exists, which might possibly lead to the fact that for each specific site, different models must be developed. Further, care must be taken to other specific situations in mixing zones (estuarine and coastal) compared with open oceans. For instance, low background levels of (essential) metals in open ocean environments occur. Coastal and estuarine environments on the other hand, may well have higher metal levels (Sadiq 1992). Care must be taken not to derive PNEC values or quality standards that are below natural background levels of (essential) metals, occurring in pristine saltwater environments. In general, different metals will act differently on increasing saltwater concentrations and this should be accounted for on a case-by-case basis (e.g. Turner et al. 2002, Pohl and Hennings 1999, Tipping et al. 1998, Toteja et al. 2001).

For marine sediments similar bioavailability parameters play a role as what has been encountered in freshwater sediments (see Fact sheet 4). Both Acid Volatile Sulfides (AVS) as organic carbon may mitigate metal toxicity in marine environments. Di Toro et al (1990) introduced and proved even his SEM-AVS concept for the first time using 10 d acute sediment toxicity tests with the marine amphipods *Ampelsica abdita* and *Rhepoxynius hudsoni* and using cadmium as model toxicant.

2.2 Biotic factors

2.2.1 Species diversity

Since the physico-chemical conditions encountered in the marine ecosystems differ from freshwater ecosystems, freshwater and saltwater organisms have developed very different strategies to cope with ion- and osmo-regulatory problems related to living in either very low or high dissolved salt concentrations. The observed gradients in ionic strength and dissolved nutrients resulted in both the marine and the freshwater environment to the development of very specific biological assemblages. A key question that has to be answered is to what level the observed differences in species diversity between marine and freshwater ecosystems might influence the vulnerability of the system.

Overall it can be stated that there is a wider diversity of taxonomic groups in saltwater- compared to freshwater environments (Box 1). Yet, when one looks at the relative numbers of individual species the picture is reversed. While higher species diversity may lead to a wider sensitivity distribution there could also be a considerable functional overlap making the system less sensitive to the potential impact of a substance emitted to this environment. On the other hand there also exist distinct scenarios (e.g. brackish systems) with lower species diversity and a narrower sensitivity distribution. Such systems can be more vulnerable due to the ecosystem function dependency on a small number of key species. However, the latter is not marine specific. Based on the observations of Briand (1983) who evaluated the diversity, connections and mean maximum food chain length of 40 food webs, comprising marine, freshwater and terrestrial communities there is no scientific basis to assume that “low species diversity-high ecosystem dependence” is higher in marine systems than in freshwater systems. With regard to biomagnification, food chains on average do not appear to be longer in marine ecosystems than in other ecosystems. In both freshwater and marine ecosystem, it is generally possible to define five basic trophic levels (primary producers, primary consumers, detrital feeders, secondary consumers and degraders). Marine systems do tend to contain carnivorous invertebrates that are considerably larger than those in freshwater. Therefore, there may exist a *potential* for more links in the marine food chain and within the marine environment (although not necessarily exclusively) there is potentially a further emphasis relating to the impact upon larger marine species of mammal or birds (i.e. high trophic level species) (ECETOC 2001).

Box 1: Comparison number of species: taxa in marine and freshwater environments

ECETOC (2001) evaluated the number of species encountered in freshwater and marine ecosystems. The evaluation was based on gathering phylogenic data on fauna to determine species abundance within an ecosystem. However, when interpreting these results, one has to bear in mind that (1) many species have not yet been discovered, (2) phylologists classify species differently (e.g. dividing or grouping organisms in greater or smaller categories), and (3) it does not account for species at higher trophic levels being less numerous. Overall, more species groups, but less individual species are present in the marine environment.

An overview of the species diversity as reported by ECETOC (2001) is shown in Figure 1. The phylum *Arthropoda* has been separated into the two classes *Crustacea* and *Insecta* to highlight the differences between marine and freshwater environment. Likewise, elements within the phylum *Chordata* have been highlighted separately.

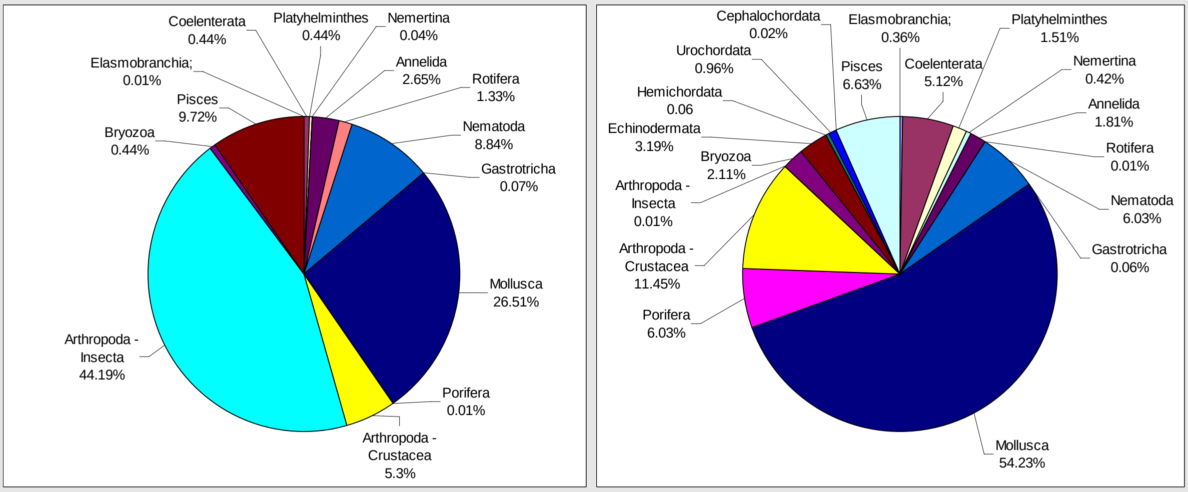

Figure 1: Species composition of freshwater (A) and marine (B) fauna (redrafted after ECETOC 2001).

In freshwater ecosystems, insects (44.2 %) represent the largest group, followed by mollusks (26.5 %), fish (9.7 %) and nematodes (8.8 %). Crustaceans amount for 5.3 % of the freshwater species. In saltwater ecosystems, mollusks represent 54.2 % of the species, crustaceans 11.5 %, fish 6.6 %, poriferans (sponges) 6 % and coelenterates *(*cnidarians*)* 5.1 %. When comparing the largest groups in both freshwater and saltwater ecosystems, it is striking that only a very limited number of insects are present in the marine environment (0.01 %), while this is by far the largest group of freshwater species. Further, mollusks constitute more than half of the marine fauna, but only a quarter of the freshwater fauna.

Moss (1988) and Barnes (1984) stated that 56 phyla were present in marine waters compared to 41 in freshwaters. Excluding a few groups comprising of solely parasites, species of all known phyla (70 to 90 depending on the classification system) are represented in the sea or in the transitional coastal fringe (Moss 1988, Barnes 1984). No phyla are confined to freshwaters only, while 15 phyla are found only in marine waters. Examples of these are the *Echinodermata*, *Ctenophora*, *Polyplacophora*, most *Gastropoda* and *Bivalvia*, *Cephalopoda*, and most *Polychaeta*. Other ‘smaller’ marine groups include the *Sipunculida*, *Echiuroidea* and *Pogonophora*. In contrast, land and freshwater organisms represent a smaller range of known phyla and classes. Yet, when one looks at the relative numbers of individual species the picture is reversed. Using a conservative estimate of 2 to 2.5 million described species of living organisms, 10 % of these (250 000) are marine. Of the species occurring in the marine environment, only 2 % inhabit the water mass itself, the other 98 % live in or on sediments or other substrates (Barnes & Hughes 1988, Janssen 2000).

Overall, although there are more taxa in the marine environment as a whole, there is no evidence that particular communities in the marine ecosystem are consistently more diverse than those in freshwater or terrestrial systems. However, there are a number of phyla and/or classes of organisms which occur mainly or exclusively in marine environments. Examples of these are the *Echinodermata*, *Ctenophora*, *Polyplacophora*, most *Gastropoda* and *Bivalvia*, *Cephalopoda* and most *Polychaeta.* Other ‘smaller’ marine groups include the *Sipunculida*, *Echiuroidea* and *Pogonophora*.

2.2.2 Species sensitivity

Freshwater and marine organisms face very different ion- and osmoregulatory problems related to living in either a very dilute or concentrated salt environment. Organisms living in estuaries face even more harsh conditions since they live in a salt gradient and have to cope with tidal and seasonal fluctuations in salt concentration. Depending on their evolutionary history, organisms have developed different solutions to deal with these physiological challenges. These differences in physiology may also have an impact on metal uptake and elimination processes (Smolders et al, 2004).

In general, gills and surface (skin, carapax, epidermis…) are the primary routes of metal uptake for most freshwater organisms. Freshwater organisms normally do not drink since they take up water by passive exchange across the permeable body surfaces. They live in a very dilute environment and have to actively retrieve major ions such as Na^{+}, K^{+}, Ca^{2+} and Mg^{2+} from the environment and the gills and digestive system are actively involved in this. In contrast, saltwater organisms live in a much more concentrated environment and depending on the exposure conditions and their ion- and osmo-regulatory physiology they are either hypo-ionic or hyper-ionic in relation to their environment. Saltwater organisms loose water and have to compensate this loss by drinking (Prosser 1991, Schmidt-Nielsen 1997, Willmer et al 2000, Smolders et al. 2004). This also means that they take in metals in solution via the digestive system. Both the epithelia of gills and gut are important and sensitivity targets. The observed differences in physiology, however, do not inherently lead to an overall greater sensitivity of marine species and it remains unclear whether the exposure through the digestive track also translates into a higher sensitivity to metals. Indeed, for the same internal metal concentration exposure via the gut does not necessarily results in the same toxic effect as exposure via the gill (Thomann et al, 1997, Szebedinszky et al 2001, Campbell et al , 2003, Sappington et al, 2003). Saltwater species may also have pelagic planktonic stages which can exhibit different sensitivities to chemicals and some reproductive strategies of marine invertebrates are less responsive to changing environmental conditions, which might be expected to lead to differences in sensitivity to toxicants (Hutchinson et al. 1998, Smolders et al. 2004).

Since there are generally fewer toxicity data available for saltwater species than for freshwater species (Hutchinson et al. 1998, Leung et al. 2001, Wheeler et al. 2002)[1] it could be worthwhile to explore on a case-by-case basis if freshwater data can be used in in case of absence of relevant marine data for some taxa (see section 3). As stated before, there are a number of important physico-chemical differences between the saltwater and freshwater environment. Furthermore different communities exist in saltwater ecosystems compared to freshwater ecosystems, with a number of taxa restricted only to marine and freshwater environments. Therefore, a relevant question to pose is whether or not marine species are more or less sensitive than freshwater species and if extrapolation from the freshwater compartment to the marine compartment should be allowed for the metal/metal compound under consideration without correcting for these differences. In this regard care should be taken in the interpretation of toxicity data generated with open sea species because these species may come from a metal-deficient area.

The sensitivity of marine species in comparison with their freshwater counterpart and the sensitivity of unique marine species have been addressed in a limited number of studies only (Box 2). Some of these studies assessed the sensitivity of freshwater versus saltwater species to metals and/or organic chemicals by studying existing effects databases. Other studies deal with a number of individual metals studied in one species.

[1] This is largely because risk assessments have been mainly focussing on freshwater systems and less standard test methods are available for saltwater species.

Box 2: Comparison sensitivity freshwater/marine species

Several authors have based their comparison in species sensitivity derived species sensitivity distributions (SSD) from the USEPA's AQUIRE and ECOTOX databases. Leung et al. (2001) and Wheeler et al. (2002) extracted acute median LC_{50} data were extracted from the database for a number of chemicals amongst which cadmium, copper, lead, mercury, nickel, chromium and zinc. No true quality check has been performed on the data retrieved from AQUIRE. Except for zinc, HC5 values were within one order of magnitude for freshwater and saltwater species. In general, the differences between freshwater and saltwater responses, as described by the SSDs, were not large and freshwater species appeared to be more sensitive than saltwater species. The use of freshwater data would thus be protective for saltwater species.

The USEPA ECOTOX database was also used by Smolders et al. (2004) to calculate HC5 values in freshwater and saltwater systems for different metals (cadmium, cobalt, copper, lead, mercury, nickel and zinc). As an example, the SSDs for lead is given in Figure 2. On a chronic level both freshwater and marine SSDs almost coincide.

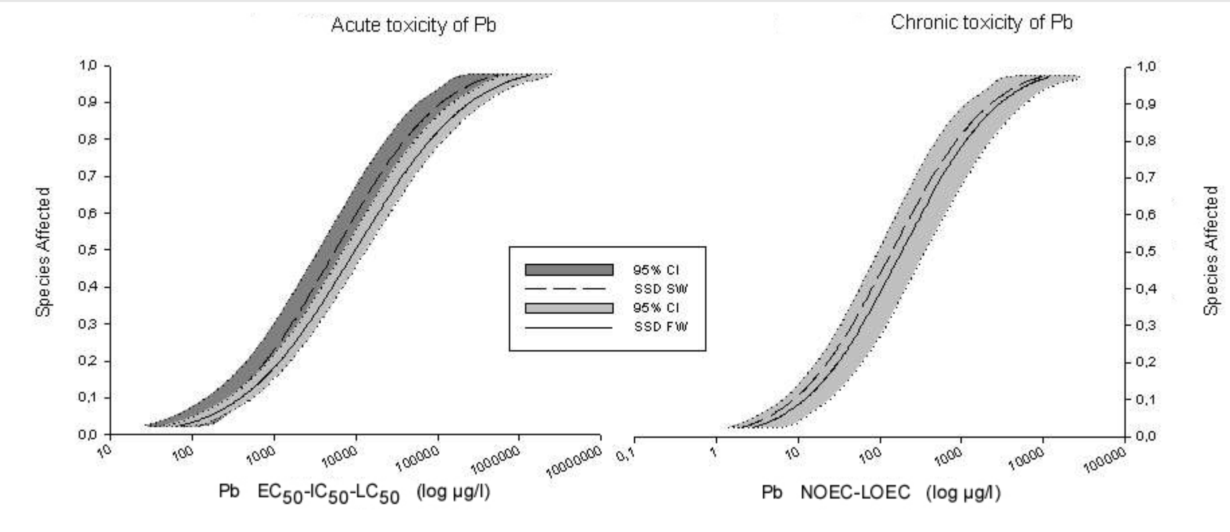

Figure 2: Comparison between freshwater (FW) and saltwater (SW) organisms for a) acute and b) chronic toxicity of lead (from Smolders et al. 2004).

The authors conclude that there are no clear indications for important differences between freshwater and saltwater organisms in terms of acute and chronic toxicity for copper, cobalt, nickel, lead and zinc when expressed on a total metal concentration scale, with freshwater species being slightly more sensitive (acute and chronic freshwater and saltwater HC5 of the other metals are all within a factor of 4.4). In the case of cadmium, freshwater species were considerably more sensitive for chronic toxicity. For cadmium, the chronic HC5 value is 13.4 times lower in freshwater compared to saltwater.

Another series of studies have described the ECETOC EAT database (Hutchinson et al. 1998, ECETOC 2003). Freshwater versus saltwater species sensitivities are described and 'general chemicals', 'pesticides' and 'metals' are differentiated. A limited number of metal studies are available and thus the results must be interpreted with caution. Only relatively small differences between sensitivities of freshwater and saltwater species were found. A physiological similarity is suggested between species belonging to similar taxonomic groups regardless of their freshwater or saltwater origin. The authors conclude that the use of freshwater acute effects data instead of or in addition to saltwater effects data for risk assessment purposes is not contra-indicated by the empirical data reviewed.

Although no metals were included in their study, LeBlanc et al. (1984) confirmed previous data that relatively little differences in sensitivity exist between freshwater and saltwater species. Much greater differences between different taxonomic groups exist (e.g. different taxonomic groups of algae) than between related freshwater and saltwater species. Differences in sensitivity of different trophic levels were also as great as or greater than the paired species differences. Other authors concluded that marine species were more sensitive than freshwater species (e.g. Robinson 1999). Part of the higher sensitivity of invertebrate saltwater species may be attributed to the longer exposure period in saltwater invertebrate tests. Overall, several authors found that differences between freshwater and saltwater species sensitivities for metals and other compounds were generally within a factor of 10 (e.g. Calleja et al. 1994, Dawson et al. 1977, Hemmer et al. 1992, Sorokin 1999).

Although most often the used data in the previous studies (Box 2) were not thoroughly screened for quality there appears to be a relatively good agreement between the sensitivity of fresh- and saltwater fish. This is partly due to the fact that fish represent only 1 taxon. For invertebrates, less agreement is found, but this is probably largely due to the fact that several higher taxa are included in 'invertebrates' (contrary to fish). In general it can be stated that the results of the different studies do seem to indicate that the differences between freshwater and saltwater responses, are not large and freshwater species appear to be even more sensitive than saltwater species. The use of freshwater data would thus be protective for saltwater species. It is likely that different sensitivities of freshwater and saltwater species are at least partially a consequence of differences in speciation and bioavailability in the different media. In particular, the greater abundance of uncomplexed or free ionic forms of metals under freshwater conditions and the greater ability of saltwater organisms to regulate uptake of some metals such as zinc and copper probably contributes to the greater tolerance of these substances by saltwater organisms. Therefore, comparison of 'real' sensitivities between fresh- and saltwater organisms should ideally be performed based on the bioavailable fraction rather than on total/dissolved metal concentrations.

Differences in sensitivity between similar freshwater and saltwater species within a taxon also seem to be smaller than different between species belonging to different taxa. Hence, it is likely that extrapolating from freshwater to saltwater species will introduce less uncertainty than extrapolating between non-related taxa.

Most of the comparative studies are based on the comparison of taxa occurring in both the freshwater and marine environment. However, as indicated before there are number of phyla and/or classes of organisms which occur mainly or exclusively in marine environments. Few data on the sensitivity of these species are available in the literature. Ghirardini et al. (2001) found similar sensitivity with the sea urchin (*Paracentrotus lividus*) exposed to a range of surfactants and their biotransformation products compared with different freshwater species. In another study Geffard et al. (2001) demonstrated that oyster embryos (*Crassostrea gigas)* were somewhat more sensitive than the sea urchin larvae when exposed to natural sediments contaminated with PAHs and metals. The authors concluded that the sensitivity between the two species remains within an order of magnitude, confirming the findings of His et al. (1999) with the same species. Based on these limited number of publications, there are no indications that the sea urchin (representative from the key marine taxon of the *Echinodermata*) is specifically more sensitive than the oyster (representative from the more common marine taxon of the *Mollusca*). Hunt et al. (2002) studied nickel toxicity in a marine fish, mollusk and crustacean. Acute to chronic ratio's (ACRs) for nickel toxicity to the three species were similar to another saltwater value (for mysids) and significantly lower than the existing values for freshwater species. However, these authors used total nickel concentrations to describe and compare the effects measured and did not account for bioavailability.

2.2.3 Conclusions

Although there are more taxa in the marine environment as a whole, there is no clear evidence that particular communities are consistently more diverse than those in freshwater or terrestrial systems, and that there are consistent differences in sensitivity between marine taxa and those in other environments. Existing marine effects databases generally lack data on certain key marine taxa (echinoderms, mollusks) which make it difficult to judge that such phyla are potentially more sensitive and that any effects assessment conducted in the absence of such data may not capture this sensitivity. From the limited data available there are, however, no specific indications that species from (key) marine taxa are consistently more sensitive than freshwater species, and the same critique can also be leveled at the freshwater effects database which similarly lacks information relating to key taxa such as mollusks and insects.

3. DERIVATION OF ECOTOXICITY THRESHOLDS (PNEC OR EQS) FOR MARINE SPECIES

Deriving an ecotoxicity threshold (PNEC or EQS) for the marine environment should follow the same principles as for the freshwater environment. Meaning that all available effects data for marine species belonging to different taxa have to be gathered and screened for quality ((see Fact Sheet 3 for more details on this process). Since there are only a limited number of standard guidelines on toxicity testing in the marine aquatic and sedimentary environment mostly fewer species mean NOECs/EC10s will be available for marine organisms than for freshwater organisms.

In appraising toxicity data for the development of water quality guidelines (EQS) or Predicted No effect Concentration (PNEC) two approaches can been used (Warne, 1998 in Batley et al, 1999, MERAG fact sheet 2). The first and traditional approach uses assessments- or safety factors applied to toxicity data. This empirical approach is based largely on intuition, factors that have typically been assigned order of magnitude numbers such as 10,100, 1000. These factors represents a typical example of the Precautionary principle and as such are conservative and over-protective in most cases. In the EU even higher assessments factors have been suggested in order to account for the presence of key marine sensitive taxa (TGD, 2003). The second and scientifically preferred method involves the fitting of all available and acceptable data to a statistical distribution, known as the Species Sensitivity Distribution (SSD) approach.

If sufficient marine ecotoxicological data are available to build a saltwater Species Sensitivity Distribution (SSD) the use of the statistical extrapolation method is recommended (Figure 3). However, due to a general paucity of reliable marine data (using standard test species there is often an inability to utilize the SSD approach for deriving an ecotoxicity threshold for the marine environment. In those cases it could be evaluated whether or not freshwater ecotoxicity data can be pooled with marine ecotoxicity data in order to increase the size of the marine database and thereby allowing the use of the SSD approach for deriving a PNEC or EQS. If there is proof that freshwater species exhibit a similar sensitivity or are even more sensitive than saltwater species it can be considered to pool both databases and as such a hybrid SSD can be applied. Pooling of freshwater and marine data should, however, only be considered when one dataset is not sufficiently robust to derive an SSD on its own. Pooling freshwater data with a less robust marine database is appropriate, but the inverse should not be performed, as this will increase uncertainty in the ecotoxicity threshold for the freshwater. Also care should be taken to avoid pooling freshwater taxa that do not occur in marine ecosystems (e.g. amphibians) or are unlikely to occur in strict marine ecosystems (e.g. insects). Criteria for determining the adequacy of taxonomic coverage for marine databases have not yet been addressed but the initial criteria of the London workshop can be used as a starting point[2].

Usually the data are not there to assess the similarity between the sensitivity of freshwater and marine species. In case not enough marine data are available and the evidence of the comparative sensitivity evaluation is not conclusive to allow cross-reading between the saltwater and freshwater data then the assessment factor approach may be the only alternative. An assessment factor should be applied on the lowest marine data point. In the latter case since there is no scientific evidence that marine ecosystems are more sensitive than freshwater systems the size of the assessment factors applied should be similar to the ones used for the derivation of a freshwater ecotoxicity threshold[3].

[2] The London 2001 workshop provided criteria for the use of SSD analysis for freshwater databases. However, these criteria are not directly applicable to marine systems due to fundamental differences in organisms represented in marine and freshwater environments. For example, insects are an essential freshwater taxonomic group according to the London criteria; however, insects do not live in strict marine environments. Likewise, representative marine taxonomic groups such as echinoderms do not occur in freshwater environments.

[3] In some legislative systems the application factors for the marine environment have been raised (TGD, 2003). But as also indicated by the Scientific Committee on Toxicity, Ecotoxicity and the Environment (CSTEE) there are no scientific grounds for raising the assessment factors (CSTEE, 2002)

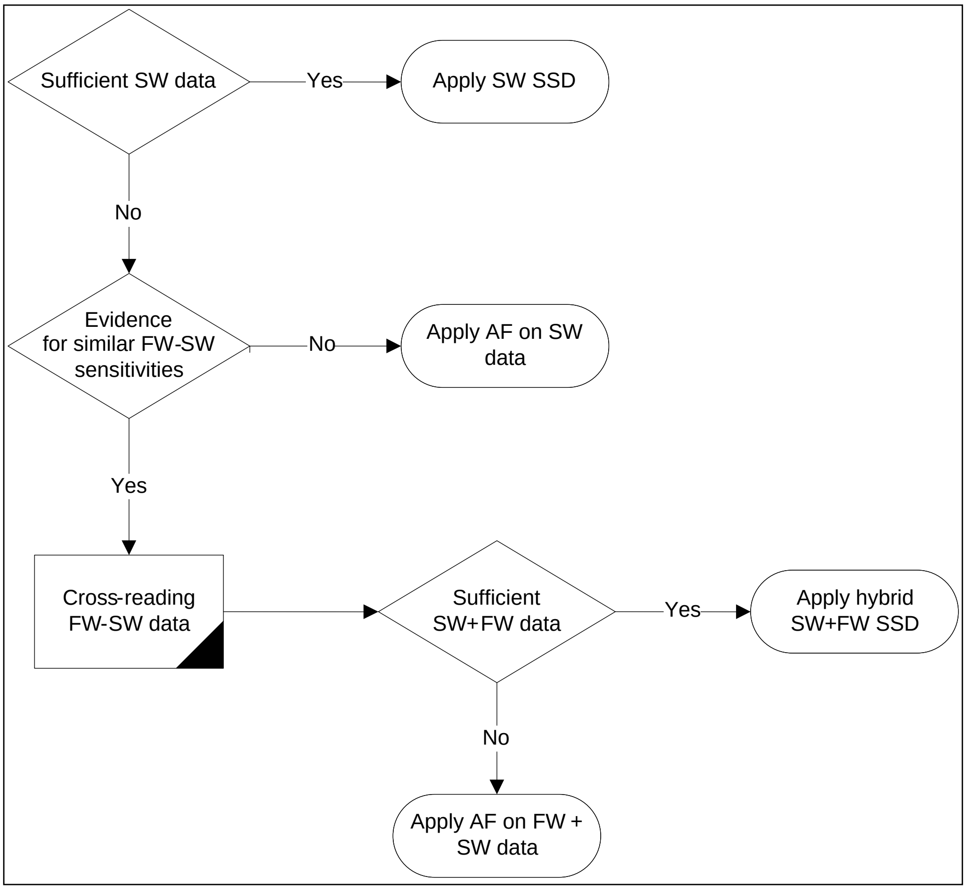

Figure 3: Methodology outline for the derivation of marine risk assessments (PNEC derivation). AF = assessment factor, SSD = species sensitivity distribution, SW = saltwater, FW = freshwater.

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