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ZINC IN SEDIMENT

AN ENVIRONMENTAL QUALITY STANDARD OVERVIEW

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Introduction

Zinc is an essential trace element required by most organisms since it act as a cofactor for more than 200 metalloenzymes (Anu et al., 2018). However, when environmental concentrations exceed what is physiologically required for a specific species, zinc may compete with other essential elements for binding sites in proteins and cause negative effects. Thus, it is important to ensure that zinc concentrations in different environmental compartments (e.g. sediment and surface water) do not exceed levels that may cause adverse environmental impact for biota. In the European Union (EU), this process is handled through several legislations related to chemicals, e.g. REACH, the Plant Protection Product Regulation, the Biocidal Products Regulation and the Water Framework Directive which require member states to either use EU-wide Environmental Quality Standards (EQS) for priority substances and priority hazardous substances (WFD, 2013) or establish national EQS for specific pollutants that are discharged in significant quantities into a water body, so-called river basin specific pollutants (RBSP).

Zinc is frequently listed as a RBSP and EQS values has been established for freshwater by all EU member states except Finland and Italy (Vorkamp and Sanderson, 2016). The derivation of EQS values shall be performed according to the protocol described in the Technical Guidance Document No. 27 (European Commission, 2018), hereafter TGD 27. The first version of TGD 27 was published in 2011 and an updated version was issued in 2018. Briefly, TGD 27 focus on the steps involved for deriving an EQS, e.g. types and quality of data required, extrapolation and choice of assessment factors and how to account for background concentrations and bioavailability. The TGD 27 does not consider the implementation phase (e.g. design of monitoring programs, sampling strategies and chemical analysis etc), this is instead described in more detail in the Technical Guidance for implementing Environmental Quality Standards (EQS) for metals No 38, hereafter TGD 38 (European Commission, 2019).

Despite the guidance in TGD 27, there is a huge variation in national EQS-values between EU member states which for zinc (in surface water) range between 3.1 to 1300 µg/L, i.e. almost three orders of magnitude (Vorkamp and Sanderson, 2016). However, only 24% of the EQS values were established after 2010, i.e. the vast majority of the EQS values were developed prior to when the TGD 27 was developed. The analysis by Vorkamp and Sanderson (2016) also showed that deterministic approaches, i.e. the lowest chronic value combined with an assessment factor, was the main method to develop EQS values. The used endpoint and NOEC (No Observed Effect Concentration), the selected assessment factor as well as how background concentrations and bioavailability were treated in the establishment of the EQS all contributed to the large variation. This discrepancy between EU member states has also been observed for copper where national EQS values for surface water span from 0.87 µg/L to 50 µg/L (Lagerström et al., 2021). However, for sediment, many member states (including Sweden) have either not established EQS values for most metals or they are out of date and need to be revised based on the approach described in the TGD 27.

The overall aim of the proposed report was to develop an Environmental Quality Standard overview for zinc in marine and freshwater sediments, including to propose threshold values of zinc in these two compartments. In addition, the proposed threshold values were compared to measured zinc concentrations from monitoring data in freshwater and marine sediments.

Method selection and considerations

The derivation of a new EQS for zinc in sediment was based on the guidelines published in the TGD 27 (European Commission, 2018). As the guidance document primarily focuses on the water column, expert judgement and previous experience from the derivation of an EQS for copper in marine sediment served as additional complement (see Lagerström et al. (2021)). An extensive literature study, to collect critical and supporting ecotoxicological data, was conducted prior to the analysis (see section Data collection and selection below).

As suggested by the TGD 27 and supported by experts (workshop described in Lagerström et al. (2021)), the background concentrations of RBSP should be assessed during the implementation of the EQS rather than during the derivation process. In one of the tiered approaches, according to the TGD 38, it has been suggested that, as a second tier, the EQS should be recalculated with the added risk approach. However, as the TGD 38 also suggest deriving and implementing one specific EQS value and that this value should be of total concentration for tier 1 evaluations, the added approach might be used in the implementation phase, but this is out of the scope of this overview. Also, the organisms inhabiting the sediment have no ability to distinguish between zinc of natural and anthropogenic origin, making the added risk approach ecologically inappropriate. Therefore, the total risk approach is the preferred approach and will be used here to derive an EQS for zinc in sediment.

Bioavailability

The bioavailability, i.e. the fraction of the total concentration of metals that is available for uptake by organisms, vary with the physico-chemical properties (e.g. pH, salinity, grain size, organic matter and redox potential) of the sediment. The observed effect concentration can thus vary between ecotoxicological studies and will be impacted by the prevailing conditions within the studies. Biotic ligand models (BLMs) have been developed for some freshwater species to account for the bioavailable fraction of e.g. zinc in the water column (Heijerick et al., 2002a; Heijerick et al., 2002b) but no validated BLM exist for sediment. In sediment, metal bioavailability and toxicity is typically governed by the organic carbon (OC) content and the levels of acid volatile sulphides (AVS) (Zhang et al., 2014).

If a relationship with OC can be discerned, the TGD 27 implies that a normalisation of the toxicity data to an EU ‘standard sediment’ of 5% OC content is preferred when deriving an EQS for the sediment. Norwood et al. (2009) demonstrated a linear correlation between higher OC content in the sediment and increased lethal concentration (LC50) (i.e. decreased toxicity) for *Hyalella Azteca* exposed to zinc. During a linear regression analysis, based on aquatic threshold values from more than 2000 Monte Carlo simulations, it was shown that the dissolved OC was the main variable affecting the calculated threshold value of zinc in limnic surface waters (Van Sprang et al., 2009). Further, Heijerick et al. (2003) concluded that the dissolved OC content appeared to be the factor that most strongly modifying the toxicity when modelling chronic toxicity on *Daphnia magna*. Similar relationship was observed with copper and adverse effects on benthic living organisms (Lagerström et al., 2021), where normalisation to 5% OC was supported by workshop participants and the TGD 27 as it reduced the variability within the dataset. In the case of zinc and the effect data in sediment, the max:min ratio, calculated from the compiled data within this study, is reduced from 54 to 50 for the entire dataset and from 9 to 6 for *H. Azteca* survival endpoint (n=7). *H. Azteca* was the only species where the number of tests justified the max:min ratio comparison (n>3). The relative standard deviation is similar for the OC normalised and the non-OC normalised dataset at 89 and 87% respectively. As the dataset for zinc is smaller as compared to the data set used for deriving an

EQS for copper (Lagerström et al. 2021), the results from the derivation of the copper EQS, is used as complement to support the normalisation of zinc to OC.

The TGD 27 highlights the importance of ecotoxicological studies to represent realistic worst-case scenarios. The affinity of zinc to AVS is not as high as for other metals (e.g. copper) (Simpson et al., 2000) and AVS thus have less potential impact on zinc bioavailability enabling a less strict limit than the 1 µmol/g dw proposed by the TGD 27. This, in turn, allowed a larger number of studies to be included in the final analysis but the additional uncertainty of potentially underestimating the toxicity was included in the determination of assessment factor. In this work, the ecotoxicological dataset has been limited to sediment with low to medium AVS (<10 µmol/g dw) content and the zinc concentrations will be normalised to 5% OC prior to the analysis. When AVS is not reported, evaluation was based on whether or not aeration was performed during the test period; if that was the case and if the OC content was not higher than 11%, the study was included.

Data collection and selection

The ecotoxicological data was collected from previously published risk assessment reports (JRC, 2010; Maycock et al., 2010), the REACH dossier (ECHA website) and studies published in scientific papers (Appendix 1). All data that was included in the final assessment have been analysed with respect to reliability and relevance (Appendix 1), either by previous assessors according to the Klimisch model (Klimisch et al., 1997) or, for more recent studies, according to the CRED model (Moermond et al., 2016).

To qualify as valid data for the final assessment, the targeted studies were chronic, single-species studies including both marine and freshwater benthic species being exposed to zinc-spiked sediments. Studies covering a total of 14 species fulfilled the criteria. Two of the studies (MacFarlane and Burchett, 2002; Guy et al., 2006) did however not report the OC content of the sediment, needed for the normalisation to 5% OC, and these were therefore excluded from the final analysis.

The final data set consisted thus of 12 different species (consisting of 7 taxonomic groups on order level) represented by a total of 21 end-point values (Appendix 1). All but three of the species were represented by only one study, the exceptions were *H. azteca* (n=7), *Chironomus tentans* (n=3) and *Chironomus riparius* (n=2). As illustrated in figure 1, the species represented different feeding and living conditions in both marine and freshwater sediment. A majority of the studies were conducted on limnic species in freshwater conditions while the marine species are represented by *Melita plumulosa* and *Quinquelaophonte sp*.

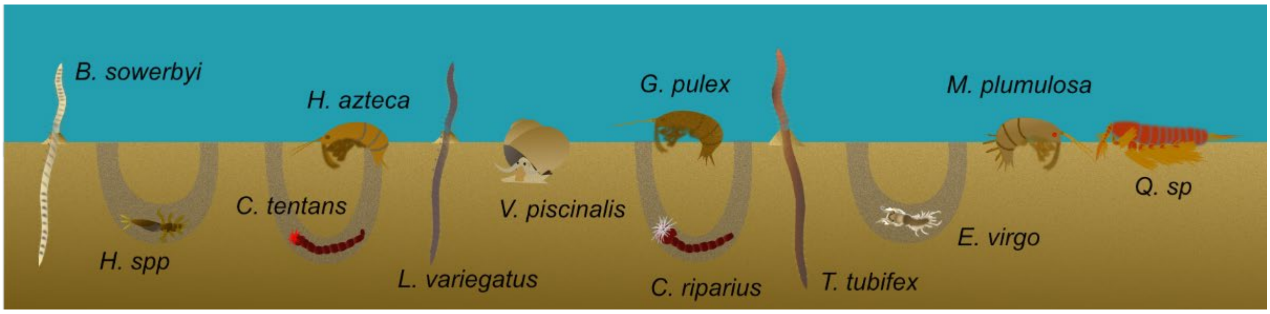

Figure 1: All the species, and a schematic illustration of their feeding/living conditions, that were included in the final species sensitivity distribution (SSD) analysis (Figure 4 and 5). Toxicity data from the two species to the right (M. plumulosa and Q. sp) where derived from tests using marine sediments while the rest of the toxicity data were derived from tests using freshwater species.

Selected approach for the derivation of the environmental quality standard

Previous derivations of an EQS for zinc in sediments have been based on the deterministic approach, where the critical value (i.e. the lowest NOEC/EC10 value) is divided by an assessment factor (JRC, 2010; Maycock et al., 2010). However, the probabilistic approach, where all available data are accounted for in a cumulative probability curve fit (e.g. a species sensitivity distribution (SSD)), where the preferred methodology listed by experts in the expert elicitation workshop conducted by Lagerström et al. (2021) and this is also supported by the TGD 27.

For freshwater sediment, a total of 10 species, representing 6 taxonomic groups, could be included in an SSD for zinc. Thus, almost fulfilling the water compartment criterion for the number of species and taxonomic groups to perform a probabilistic approach. For marine sediment, only two species covering two different taxonomic groups were represented in the data set. With support from the expert elicitation, leading up to derivation of an EQS of copper in Baltic Sea sediments, pooling of marine and freshwater data was supported if this allows for an SSD analysis instead of the deterministic approach (Lagerström et al., 2021). Before pooling, the TGD 27 recommends that a statistical test (e.g. a two-tailed t-test) is conducted to check for differences between the datasets (freshwater vs. marine). Due to the limited marine data, this was not feasible for zinc but as the two NOEC values representing the marine species are distributed at the far left and far right of the SSD curve (Figure 5), indicating that a pooled dataset can be representative for marine species, the pooling was selected. Also, the brackish properties of large areas of Swedish waters, could also support a pooling of marine and freshwater sediment ecotoxicological data to derive the EQS of zinc in sediments. However, additional data might change the assumptions made in this EQS overview and more studies are requested in order to validate or reject the pooling approach.

The TGD 27 does not provide guidance on the minimum number of species or taxonomic groups, required to perform an SSD for metal in the sediment compartment but for water it should be at least 10 species representing 8 taxonomic groups (European Commission, 2018). Based on the pooled dataset, the number of species is fulfilled but the number of taxonomic groups is not (7 instead of 8). However, due to the lower level of biodiversity in sediment compared to the water column, less taxonomic groups can be accepted for the sediment compartment as long as different feeding and living conditions are represented as they are (figure 1).

Proposed environmental quality standard

The HC5 value (Table 1, figure 2), derived from the best fitting distributions (normalized to 5% OC) was used for the EQS setting. Considering the lower limit of the HC5 value and the absence of ecotoxicological data and high quality marine mesocosm/field data, an AF = 4 was applied. The proposed EQS in freshwater and marine sediments is:

$EQS_{sediment,pooled}$: 38 mg/kg dw (normalised to 5% OC)

Analysis

For the construction of the SSD curve, the US-EPA SSD Toolbox software was used (US-EPA, 2020). As recommended when performing an SSD for regulatory purposes, maximum likelihood was selected as the fitting method for the dataset (Carr and Belanger, 2019; Fox et al., 2021). The SSD Toolbox software allows for comparison of different distribution functions in order to obtain the best fit of the data. Initially, all distributions were considered, and the selection was based on the p-values, where the distributions resulting in lower p-values were excluded first. Thereafter, a comparison of assigned weight (according to AICc analysis of the software) and Goodness of fit analysis according to the QQ-plots narrowed the selection further. It is stated in the TGD 27 that "*If the data do not fit any distribution, the left tail of the distribution (the lowest effect concentrations) should be analysed more carefully*". The fit of the dataset, to any of the distribution functions, was far from perfect but in the end, the log-normal distribution function for both freshwater and pooled data was chosen as this is also preferred as a pragmatic choice according to the TGD 27 (European Commission, 2018).

All data was normalised to 5% OC prior to the analysis. The resulting SSD curves are presented in figure 2 and the details of the HC5 values with the corresponding 95% lower and upper limit of the confidence interval are presented in table 3.

Table 0-1: Results from the SSD analysis where HC5-50 represent the calculated HC5 value at the 50th percentile and the upper and lower limit give the range of the 95% confidence interval.

Dataset	HC5-50 (norm. 5% OC)	Lower limit (HC5-5)	Upper limit (HC5-95)
Pooled (n=21)	152 mg/kg dw	66 mg/kg dw	398 mg/kg dw

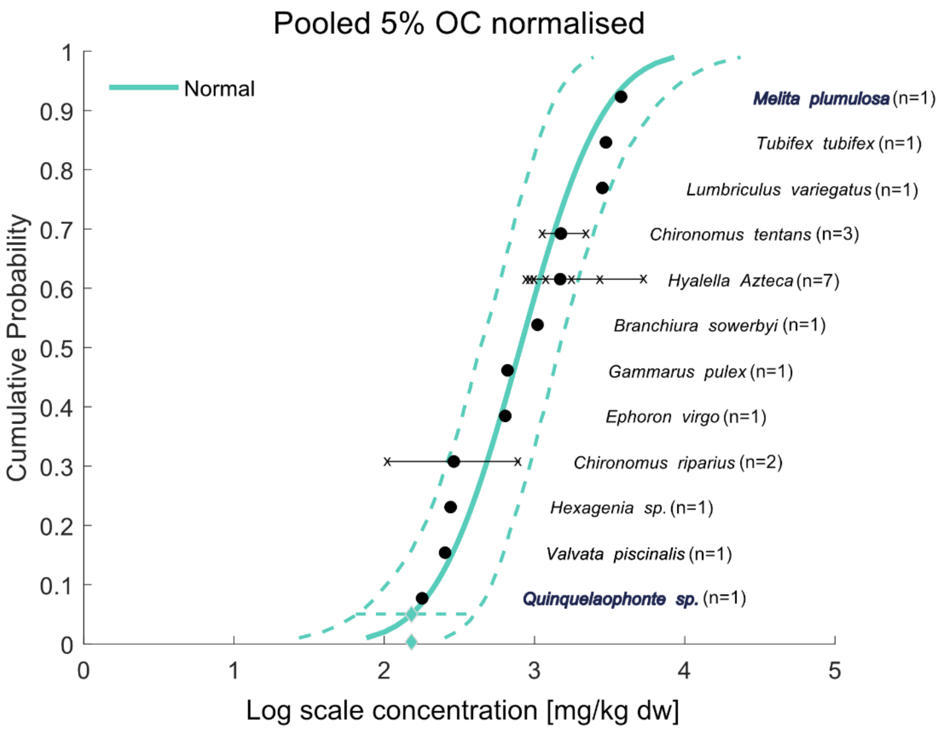

Figure 2: The SSD curves of the pooled dataset normalised to 5% OC. Black dots represent the geometric mean of the NOEC/EC10 value of the specific species (name to the right of each dot, marine species are bold dark blue) and the horizontal lines show the full range (X marks every discrete NOEC/EC10 value included in the analysis; n=number of NOEC/EC10 values). The full line showing the log-normal fitted curve and the dashed lines are the upper and lower 95% confidence interval. The diamond represents the calculated HC5 values with the horizontal dashed lines showing the 95 % confidence interval of the HC5 value. The diamond is plotted again on the x-axis for improved reading.

Uncertainties and determination of an assessment factor

Different feeding and living conditions are represented within the dataset but could be improved by introducing macroalgae, eel gras and higher plants. Also, benthic fishes might be of concern. The data availability can support an assessment factor lower than 5.

For the freshwater dataset, different species have been represented, but there is a lack in intra-species comparison yielding a high uncertainty in the final SSD curve. Intra-species comparison was only possible for *H. Azteca*, *C. tentans* and *C. riparius* and for these species a high variability (Figure 2) was observed. Similar variability was also observed when deriving an EQS for copper in marine sediment (Lagerström et al., 2021). Moreover, only two marine species are represented in the zinc data set, where one species was the most sensitive in the entire dataset, introducing a large uncertainty. Pooling of the data have been supported as an alternative to the deterministic approach, but it would be preferable to perform an SSD curve using a marine dataset.

The poor fit of the curves also represents a level of uncertainty which is also highlighted in the relatively wide range of the 95% confidence interval (Table 3). The lower limit of the confidence interval (HC5-5) is more than two times lower than the HC5-50 estimate meaning that the assessment factor should be at least 3 to be protective.

In a study by Bat and Raffaelli (1998), the calculated 50% effect concentrations (E(L)C50) of the marine amphipod *Corophium volutator* exposed to zinc for 10 days was 68 mg/kg dw for reburial and 76 mg/kg dw for mortality (both values normalised to 5% OC). This data was not included in the SSD since the ecotoxicological endpoints did not fulfil the criteria of chronic data but the derived effect concentrations are important when evaluating the assessment factor. The data show that an assessment factor of 2 (=76 mg/kg dw, 5% OC) is not protective to *C. volutator,* given that the LC50 value was 76 mg/kg dw, 5% OC. Also, in the treatment holding 50 mg/kg dw, 5% OC, no effect on survival was observed, 18 out of 20 amphipods survived which is similar as the controls where 19 out of 20 survived. Based on these results, an assessment factor of 3 (51 mg/kg dw, 5% OC) is adequate to protect *C. volutator* with respect to acute mortality but a higher assessment factor is needed to ensure protection of chronic exposure.

In an ecotoxicological study of the marine mollusc *Hydrobia ulvea (Campana et al., 2013),* no NOEC could be determined for any of the endpoints post zinc exposure and an unbounded NOEC of >797 mg/kg dw at 2.4% organic matter suggesting that *H. ulvea* does not belong to the most sensitive species.

In a field study, conducted by (Burton Jr et al., 2005), natural sediment was spiked with zinc at two exposure concentrations. A significant reduction on abundance was seen in some of the tests where the high exposure concentration was >100 mg/kg dw. The lowest concentration with a significant reduction in abundance was 175 mg/kg (Italian sediment; 0.34% OC) and 232 mg/kg dw (Dutch sediment; 4.97% OC), equivalent to 2574 mg/kg dw normalised to 5% OC and 233 mg/kg dw normalised to 5% OC. The results from this study indicate that the suggested EQS values would be protective to the benthic organisms.

Background concentrations

Another aspect to include when deriving an EQS, according to the TGD 27, is the natural background concentration. There are several different proposals of the regional natural background concentrations of zinc in sediments, but these are not normalised to OC content, making the comparison with the proposed EQS values difficult. In the marine and coastal areas of Sweden, the Swedish EPA have compiled a status classification of sediments based on comparative values and the deviation from these values (Naturvårdsverket, 1999). The comparative value (Class I) is derived from the median reference value, calculated from sediments sampled at 55 cm below the water-sediment interface which should correspond to pre-industrial times. The value separating Class I and II correspond to the background value (=85 mg/kg dw) and everything above that is classified as small to very large deviation. These values have not been normalised to the OC content.

As expected of an essential and yet toxic metal, the proposed EQS value is close to the natural background value. Although there is a risk that, on some locations, the EQS is even lower than the natural background value this is not a strong enough argument to lower the assessment factor further. The uncertainty in the determination of the EQS and the lack of knowledge regarding true background concentrations must be improved before potentially changing the assessment factor. The natural background should instead be assessed during the implementation of the EQS and efforts should be made to decrease the uncertainty in the derived EQS values.

Table 0-1: Deviation classification of metals in surface sediment based on the Swedish EPA report nr 4914. The concentration unit is mg/kg dw.

Class I	Class II	Class III	Class IV	Class V
Insignificant deviation	Small deviation	Apparent deviation	Large deviation	Very large deviation
<85	85-128	128-196	196-298	>298

Applying the EQS in Swedish waters

The derived HC5 value and the respective EQS values, dependent on the choice of assessment factor, are compared to the marine monitoring data from 2014 (collected by SGU and published on ICES dome (ICES, 2020)). Irrespective of the applied assessment factor, a majority of the sites will exceed the EQS, from approximately 50% if no assessment factor is applied to >95% if an assessment factor of 3 or higher is applied. Similar trends are observed for freshwater surface sediment (0-2 cm) concentrations of zink (normalized to 5% OC) from “background” environments according to monitoring data from 2019-2021 reported to and published in the SGU database of environmental pollutants (SGU, 2022).

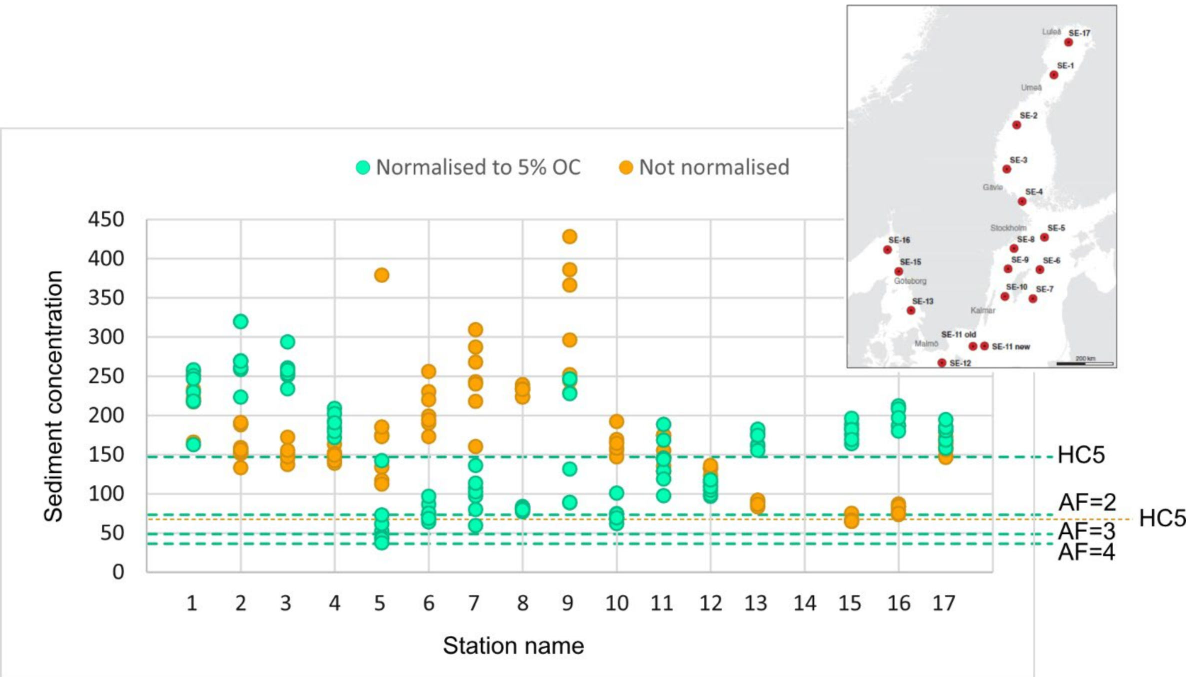

Figure 3: Surface sediment concentrations at Swedish marine stations (see map) from 2014. The green dots represent the normalised data (to 5% OC) while the orange dots are not normalised. The dashed green lines show the level of HC5-50 and the EQS values applying an assessment factor of 2, 3 and 4. The yellow dashed line show the HC5-50 value from an SSD analysis conducted on non-normalised data.

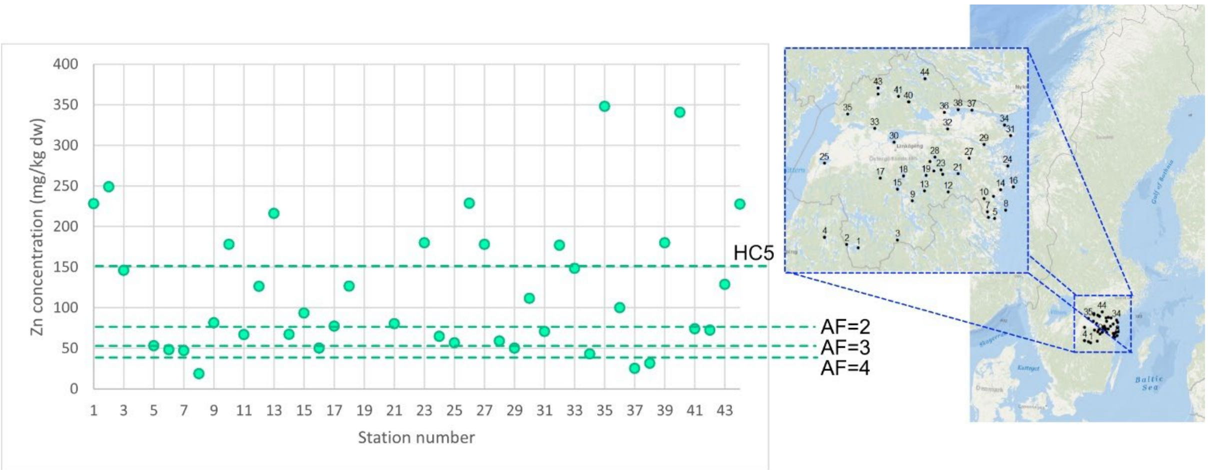

Figure 4: Surface sediment (0-2 cm) concentrations of zink (normalized to 5% OC) from "background" environments according to monitoring data from 2019-2021 reported to and published in the SGU database of environmental pollutants. The dashed lines show the derived HC5 values and the respective threshold values depending on what assessment factor is applied. At station number 19, 20 and 22, the measured concentration exceeded 500 mg/kg dw (2285, 725 and 1918 mg/kg dw normalised to 5% OC) and have not been included in the figure.

Conclusions

As the proposed EQS value is close to the natural background we propose a tiered approach where the EQS act as a trigger value for further investigation.

Normalising to OC is one tool to account for some of the varying properties of the sediment where high OC content often results in lower bioavailability, both due to the binding of zinc to OC but also related to the redox conditions where high organic matter often means low oxygen which can yield higher levels of AVS complexing with the metals, including zinc.

The complex nature of sediments and the high diversity of species makes it impossible to account for all uncertainty, and hence an assessment factor is required. Based on current knowledge, an assessment factor of 4 should be applied to the HC5 value to account for most of the uncertainties within the dataset.

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Appendix 1 – data included in the SSD

Table 0-1: Summary of all data included in the species sensitivity distribution (SSD) analysis. All studies have been assessed with respect to reliability and relevance, either from previous assessments according to Klimisch et al. (1997) in the REACH dossier, where K1 means reliable and relevant without restrictions, according to JRC (2010) with classes … and/or in the current study according to Moermond et al. (2016) where: C1 Relevant without restrictions: The study is relevant for the purpose for which it is evaluated. C2 Relevant with restrictions: The study has limited relevance for the purpose for which it is evaluated. R2 Reliable with restrictions: The study is generally well designed and performed, but some minor flaws in the documentation or setup may be present.

Genus	Species	Toxicity value (not OC normalised)	Toxicity value (5% OC normalised)	OC content (%)	AVS (µmol/g)	Endpoint	Water type	Equilibration time	Reference	Comment	Relevance/reliability
Avicennia	Marina	250				Growth	Estuary	14 d	MacFarlane and Burchett (2002)	Not included in the final SSD with OC normalisation	K2 (From new REACH dossier submission)
Branchiura	Sowerbyi	269	1121	0.5-1.8	Not reported	Survival	FW	7 d	Lobo et al. (2016)	OC content calculated from OM (%)/1.7 according to TGD 27. Calculation for OC normalisation based on 1.2% OC content (average of range). Only 1 week equilibration time.	C2/R2 – missing raw data for calculations of EC10.
Chironomus	Riparius	80	800	0.5	Not reported	Growth	FW	3 w + 6 months ageing	Norwood et al. (2009)		• K1 (From new REACH dossier submission) • C2/R2 (CRED, this study, due to not reporting AVS and added Zinc salt)
Chironomus	Riparius	232	105	11	Not reported	Growth	FW	3 w + 6 months ageing	Norwood et al. (2009)		• K1 (From new REACH dossier submission) • C2/R2 (this study CRED due to not reporting AVS and added Zinc salt)
Chironomus	Tentans	435	2175	1	1.1	Growth	FW	14 d	Farrar and Bridges (2002)		Study was deemed valid by JCR but rejected based on the fact that 10 days is not enough for

											chronic assessment.[3] As this is the most sensitive value for C. tentans (and chronic values are often lower than acute) it has been included.
Chironomus	Tentans	522	1373	1.9	1.8	Growth	FW	30 d	Farrar and Bridges (2002; 2003)	Also presented NOEC at 639 mg/kg dw	• Useful JRC (i.e. relevant and reliable)
Chironomus	Tentans	855	1125	3.8	3.9	Growth	FW	14 d	Sibley et al. (1996)		• Useful JRC (i.e. relevant and reliable) • K1 (From new REACH dossier submission)
Ephoron	Virgo	219	638	1.7	4.9	Growth	FW	40 d	Nguyen et al. (2005c)		K1 (From new REACH dossier submission)
Fundulus	heteroclitus	1103				Growth	FW	14 d	Guy et al. (2006)	Embryo-larval stage (referring to master thesis). Not included in the final SSD with OC normalisation.	
Gammarus	Pulex	201	661	1.5	6.5	Growth	FW	40 d	Nguyen et al. (2005a)		K1 (From new REACH dossier submission)
Hexagenia	Sp.	608	276	11	Not reported	Growth	FW	3 w + 6 months ageing	Norwood et al. (2009)		• K1 (From new REACH dossier submission) • C2/R2 (this study CRED due to not reporting AVS and added Zinc salt)
Hyalella	Azteca	526	5260	0.5	Not reported	Survival	FW	3 w + 6 months ageing	Norwood et al. (2009)		• K1 (From new REACH dossier submission) • C2/R2 (this study CRED due to not reporting AVS and added Zinc salt)
Hyalella	Azteca	2111	960	11	Not reported	Survival	FW	3 w + 6 months ageing	Norwood et al. (2009)		• K1 (From new REACH dossier submission) • C2/R2 (this study CRED due to not reporting AVS and added Zinc salt)
Hyalella	Azteca	1713	921	9.3	Not reported	Survival	FW	3 w + 6 months ageing	Norwood et al. (2009)	High background Zinc content (339 mg/kg dw)	• K1 (From new REACH dossier submission)

											• C2/R2 (this study CRED due to not reporting AVS and added Zinc salt)
Hyalella	Azteca	936	2463	1.9	1.8	Survival	FW	30 d	Farrar and Bridges (2001; 2002; 2003)		Study was deemed valid (interpreted as relevant and reliable) by JCR but not used as the growth LOEC = lowest exposure. We use survival endpoint and keep study.[1]
Hyalella	Azteca	229	1145	1	1.1	Survival	FW	14 d	Farrar and Bridges (2002)		Study was deemed valid by JCR but rejected based on the fact that 10 days is not enough for chronic.[2] As this is the most sensitive value for H. azteca (and chronic values are often lower than acute) it has been included.
Hyalella	Azteca	455	1379	1.7	8.2	Survival	FW	40 d	Nguyen et al. (2005b)		K1 (From new REACH dossier submission)
Hyalella	Azteca	347	868	2	7.2	Survival	FW	35 d	Nguyen et al. (2012)		C2/R2
Lumbriculus	Variegatus	933	2827	1.7	8.2	Growth	FW	40 d	Nguyen et al. (2005a)		K1 (From new REACH dossier submission)
Melita	Plumulosa	754	3768	1	<0.5	Fertility	SW	14 d	Gale et al. (2006)	High background concentration (240 mg/kg dw)	K1 (From new REACH dossier submission)
Quinquelaophonte	Sp.	39	179	1	Not reported	Reproduction	SW	14 d	Stringer et al. (2014)		Not considered in ECHA PNEC since the concentration of the control sediment was not determined and they derived $PNEC_{add}$. However, the non-spiked sediment was a control and the first exposure concentration showed no effect so the study is included here as it applies TRA and measured concentrations are provided. • C1/R2
Tubifex	Tubifex	1135	2986	1.9	1.8	Reproduction	FW	30 d	Farrar and Bridges (2003)		• Useful JRC Useful JRC (i.e. relevant and reliable) • K1 (From new REACH dossier submission)

Valvata	Piscinalis	200	253	3.9	Not reported	Growth	FW	20 d	Ducrot et al. (2006)		C2/R2

[1] From the JRC report: *The study as such is valid, but rejected, based on quality criterion (Growth was affected at the lowest concentration tested, but the sensitivity for endpoint growth compared to endpoint survival is not confirmed by the results of the two other long-term tests with H. azteca).* (p. 488)

[2] From the JRC report: *The study as such is valid but the test is rejected for chronic NOEC derivation, based on the following Quality criterion: For H. azteca a 10-d test is a short-term test which cannot be used to derive a chronic NOEC value.* (p. 491)

[3] From the JRC report: *The study as such is valid but the test is rejected for chronic NOEC derivation, based on the following Quality criterion: For C. tentans a 10-d test is a short-term test which cannot be used to derive a chronic NOEC value*. (p. 492)

Appendix 2 – alternative EQS values for marine and freshwater sediment (no pooling)

Summarising table

Dataset	Method	Threshold value (mg/kg dw 5% OC)	AF	EQS (mg/kg dw 5%OC)
Pooled	Probabilistic	152	4	38
Freshwater	Probabilistic	198	4	50
Marine	Deterministic	179	10	18

Freshwater SSD

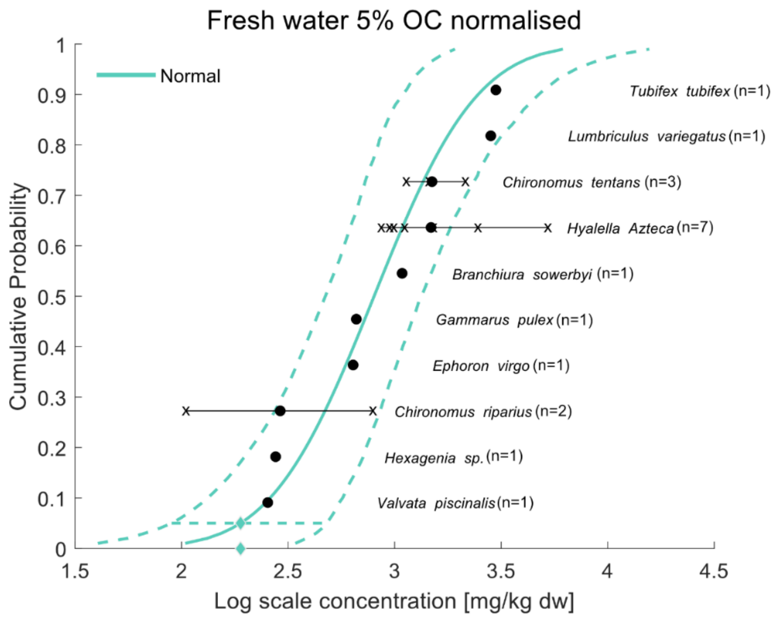

Figure 5: The SSD curves of the freshwater dataset normalised to 5% OC. Black dots represent the geometric mean of the NOEC/EC10 value of the specific species (name to the right of each dot) and the horizontal lines show the full range (X marks every discrete NOEC/EC10 value included in the analysis; n=number of NOEC/EC10 values). The full line showing the log-normal fitted curve and the dashed lines are the upper and lower 95% confidence interval. The diamond represents the calculated HC5 values with the horizontal dashed lines showing the 95 % confidence interval of the HC5 value. The diamond is plotted again on the x-axis for improved reading.

Marine critical value

Only two marine species are represented within the dataset. The most sensitive is the reproduction endpoint of *Quinquelaophonte sp* where the NOEC value (normalised to 5% OC) =179 mg/kg dw. According to table 11 and 13 in the TGD 27, the assessment factor should be somewhere between 10-50. As we have more than three long term tests but only two representing marine species it can be argued to use AF=10, resulting in an EQS of 18 mg/kg dw.

Appendix 3 – previously determined threshold values and background concentrations

Threshold value (mg/kg dw)	Method	Remark	Matrix	Country	Year	Reference
123	TRA	ISQG – Interrim Sediment Quality Guidlines	FW sediment	Canada		Canadian Sediment Quality Guidelines for the Protection of Aquatic Life
315	TRA	PEL – Probable Effect level	FW sediment	Canada		Canadian Sediment Quality Guidelines for the Protection of Aquatic Life
124	TRA	ISQG – Interrim Sediment Quality Guidlines	SW sediment	Canada		Canadian Sediment Quality Guidelines for the Protection of Aquatic Life
271	TRA	PEL – Probable Effect level	SW sediment	Canada		Canadian Sediment Quality Guidelines for the Protection of Aquatic Life
150	TRA	ER-L Effect range Low	SW sediment	US	1990	US-EPA: https://www.nj.gov/dep/srp/guidance/ecoscreening/esc_table.pdf
120	TRA	LEL - Lowest Effects Level	FW sediment	US	1990	US-EPA: https://www.nj.gov/dep/srp/guidance/ecoscreening/esc_table.pdf ; Long Morgan 1990 NOAA
122	Background	Background value	SW sediment	OSPAR	2009	The Background Assessment Concentration (BAC) is normalised to 5% aluminium in all subregions except the Iberian Sea and Gulf of Cadiz, where BACs are not normalised.
200	TRA	GV (guideline value)	SW sediment	AUS		Guideline value AUS ANZEC/ARMCANZ interim sediment quality guideline
410	TRA	GV-high (guideline value high)	SW sediment	AUS		Guideline value AUS ANZEC/ARMCANZ interim sediment quality guideline
9.5	ARA **or** bioavailability (HC5 = 19 mg/kg dw. Lower 90% limit = 4.4 mg/kg; upper = 44 mg/kg.)	EQS	Pooled sediment	Denmark	2021	Not published yet
49	ARA (deterministic) AF=10	PNEC(add)	FW sediment (but also applied to SW sediment)	EU	2010	JRC Zinc RAR
190	TRA concentration (background)=140 and PNEC=49	KN= kwaliteitsnormen	FW sediment	Netherlands	2009	Afl eiding van milieukwaliteitsnormen voor Rijnrelevante stoff en
49	ARA (refers to the studies in JRC 2010)	PNEC(add)	FW sediment	UK	2012	WFD UKTAG

49	ARA (TRA=51 mg/kg dw)	GV=Gränsvärde	FW sediment	Sweden	2008	Naturvårdsverket rapport 5799
<85	Klass I Ingen avvikelse	Comparison value (=jämförvärde) totalanalys	SW sediment	Sweden	1999	Naturvårdsverket rapport 4914
85-128	Klass II Liten avvikelse	Comparison value (=jämförvärde) totalanalys	SW sediment	Sweden	1999	Naturvårdsverket rapport 4914
128-196	Klass III Tydlig avvikelse	Comparison value (=jämförvärde) totalanalys	SW sediment	Sweden	1999	Naturvårdsverket rapport 4914
196-298	Klass IV Stor avvikelse	Comparison value (=jämförvärde) totalanalys	SW sediment	Sweden	1999	Naturvårdsverket rapport 4914
>298	Klass V Mycket stor avvikelse	Comparison value (=jämförvärde) totalanalys	SW sediment	Sweden	1999	Naturvårdsverket rapport 4914
276	N1	French sediment quality guideline for dredging (Below the level N1, the ecological impact is view as negligible. Between N1 and N2, chemical analyses must be supplemented with toxicity tests.)	SW sediment	France	2001	Mamindy Pajani et al 2013
552	N2	French sediment quality guideline for dredging (Below the level N1, the ecological impact is view as negligible. Between N1 and N2, chemical analyses must be supplemented with toxicity tests.)	SW sediment	France	2001	Mamindy Pajani et al 2013