

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

“Thought starter” background document on:

Exposure assessment

This thought starter paper has been prepared by ECHA with the support of the Scientific Committee following a structured expert consultation process. Workshops participants were requested to respond to three sets of questions covering the main discussions areas:

- Problem definition and conceptual model for sediment Risk Assessment
 - Protection goals and ecological relevance
 - Risk characterisation and environmental impact assessment
- Exposure assessment
 - Environmental fate and transfer of chemicals from water to suspended matter and sediment
 - Behaviour processes, within sediment distribution, ageing, bioavailability estimations
- Effect assessment
 - Effect assessment for epi-benthonic organisms, relevant taxonomic groups and experimental tools
 - Effect assessment for benthonic organisms, relevant taxonomic groups and experimental tools

This document reflects the feedback obtained from the participants regarding the second area, exposure assessment. Additional information has been obtained from the guidance documents, a review of available scientific literature and the input received from other experts in the field.

Disclaimer: This compilation has been prepared as a background document for facilitating the workshop discussions and does not represent a position of the European Chemicals Agency.

Index

1. Release and key environmental fate processes	2
2. Available exposure scenarios and metrics for sediment exposure assessment	5
3. Accounting for bioavailability and degradation/dissipation in sediment exposure assessment.....	10
4. Exposure assessment variability, uncertainty and prioritisation.....	15
5. Use of monitoring/field data and role of site specific emissions and assessments	18
Annex. Questions for TOPIC 2: Exposure assessment.....	21

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

1. Release and key environmental fate processes

The key processes to be considered depend on the type of assessment, and in general differ between local and regional assessments. At the local scale, two main situations are currently considered. For industrial chemicals the Predicted Environmental Concentration (PEC) is estimated for those substances originating from a point source emission, the sewage treatment plant (STP) outflow, therefore the most important processes will be those of dilution and distribution of the substance in the water body away from the outflow. Any models created need to realistically model these processes within the water body. Pesticides can enter a water body by spray drift, drainage, surface runoff and atmospheric deposition. Once in the water column, they are transported by advection and dispersion. Part of the pesticide may also volatilize and/or degrade. These environmental processes also regulate the movement of other chemicals directly released into soil, such as veterinary pharmaceuticals. It should be noted that processes such as drainage and runoff may also be relevant for industrial chemicals released into the terrestrial environment, e.g. through the application of STP sludge as fertilizer or soil amendment. For regional assessments, a low level, diffuse concentration is being considered and thus partitioning between water/sediment and degradation/dissipation will be most important.

Once the chemical has reached the water body sorption and de-sorption processes and those related to the dissipation (including degradation) are the most important for understanding determinants of the environmental fate. It should be noted that the assessment of sediment contamination is only relevant for chemicals that prefer to be in that compartment. Sorption and desorption direction goes from matrices/compartments with higher to lower chemical activity (concentration is generally a poor representative) and the rates of establishing equilibrium are strongly driven by the hydrodynamics in a system. These processes often significantly influence whether (for labile compounds) degradation occurs and to what extent. At the sediment-water interface and in the sediment layer advection, dispersion and diffusion are the important processes, and precipitation and dissolution of solids must also be considered.

Precipitation and dissolution of solids may be particularly relevant transfer mechanisms for ionic substances and metal compounds. Precipitation of metal sulfides in anoxic environments, and release of metals (and P) when re-dissolution of Fe and Mn oxyhydroxides occurs when oxic environments become sub-oxic or anoxic are particularly important near the sediment-water interface, and particularly dynamic by nature. Other processes that accompany diagenesis also need to be considered, including sulphide and Fe/Mn dynamics, as they operate in the distribution and speciation of metals in sediments.

Thus, the main mechanisms which define the environmental fate processes are: 1) partitioning between water and suspended particulate matter (SPM), and 2) settling of SPM to the bottom. From a scientific point of view SPM is an operationally-defined compartment, where the interface between what is 'suspended' and unconsolidated sediments is not always clear (e.g. where a sediment may be considered as suspended when water represents >90% of the volume). The Characterisation of SPM is also operational, because it depends on the kind of instrument (centrifuge, filtering, ultrafiltration) used to separate the aqueous and solid phases. It is also site-specific because it depends on the

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

physical and chemical characteristics of SPM. Many studies have concentrated in partitioning modelling under equilibrium conditions, but few took into account the dynamic hydrological processes. A more comprehensive and detailed modelling of the settling processes, which is needed to form the basis for understanding the actual distribution of contaminants at the regional scale, and maybe the local as well, is needed for more time and site / region specific purposes. At the site scale, the rate of consolidation of deposited SPM and the subsequent transition towards equilibrium (EqP concept) and perturbation from equilibrium (e.g., due to bioturbation, resuspension) need to be considered.

Sorption (and de-sorption when the bound concentrations exceed equilibrium concentrations) may be assessed and quantified using equilibrium partition coefficients; although in the real environment other processes such as ageing add complexity to the estimation. Issues that need addressing here are (i) how generally applicable are generic laboratory derived partition coefficients, and (ii) whether environmental conditions allow equilibrium to be achieved (e.g. how frequently resuspension occurs, or bioturbation processes that introduce oxygen to depth and influence redox gradients). Algorithms are needed to predict coefficients from basic variables of water and sediment. For non-ionic neutral organic chemicals partitioning in sediments is generally normalised to organic carbon, which is considered to be the main sorbing phase, i.e. K_{doc} , K_{poc} . Problems with this approach have been noted (e.g., when other binding phases such as black carbon are present; when sorption/desorption kinetics are slow relative to environmental transport processes). For pesticides, the Freundlich equation is the most commonly used equation to describe sorption.

In the case of metals and other ionisable compounds K_d is generally believed to be appropriate, although it must be recognized that distribution coefficients integrate binding of metals with multiple solid phases, meaning that K_d s can vary considerably among different sediments, in fact K_d estimations for metals do not work in case of anaerobic sediments. Metal partitioning will often require a more detailed description of chemical speciation to appropriately describe metal complexation with dissolved inorganic ligands (e.g., Cl^- , SO_4^{2-} , etc.), and non-linear binding to dissolved and particulate organic matter and to Fe and Mn oxide phases, and clays in very silty sediments. Competitive interactions of major ions (e.g., Ca^{2+} , Mg^{2+}) and other metals for the inorganic and organic binding sites also play a critical role. The use of the TICKET-Unit World Model has been proposed for metals¹.

Discrete particle settling, coagulation processes and the settling of flocs need to be considered in defining an overall settling rate. The settling rates are ultimately a function of the characteristics of suspended matter (e.g., particle size distributions, surface charge) and the receiving water environment (e.g., turbulence, velocity of water current, water chemistry). The assumption that the only critical source of contaminants to the sediment compartment is via adsorption to suspended matter followed by sedimentation is a vast oversimplification of reality, and misses the key processes of sediment transport. Sediment transport plays a major role in the introduction, spatial distribution, and longevity of contamination in areas of low hydrodynamic energy. Deposition and

¹ <http://www.unitworldmodel.net>

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

re-suspension of sediment depend on the type of environment (e.g., marine, estuarine, riverine and lacustrine). The main processes may depend also on the season; for example in riverine environments, re-suspension of sediments occurs predominantly in specific season (flood events in spring). The role of these processes can be very relevant for site/region specific assessments. It should be noted that some regulatory processes (e.g., marketing authorisations) require generic approaches and models covering widely dispersed users and emissions.

Once settled, the sediment processes (initial diagenesis) that influence bioavailability need to be reconsidered (including the chemical changes and differing exposure pathways for organisms). For the contaminants, transformation and degradation processes (e.g., hydrolysis, photolysis, biodegradation), as well as dissipation processes such as volatilisation that affect the overall fate of the chemical within the water body need to be described.

The substance-specific physical-chemical parameters essential for a proper exposure assessment in the sediment compartment are those describing the processes mentioned above, and depend on the fate related properties of the substance.

For organic chemicals, solubility, pKa, Henry's Law constant, vapour pressure and the partition coefficients are the most relevant environmental fate parameters. For non-ionic organic chemicals the partition can be initially described by the K_{ow} or K_{oc} (with consideration of black carbon). For detailed assessments, the most important parameter is the activity of the chemical, which controls its partitioning. For routine determinations of exposure, the freely dissolved concentrations (C_{free}), a function of the chemical's activity, can be used to predict toxicity. In recent years, C_{free} has been measured with equilibrium sampling techniques (i.e., passive samplers), which often give more accurate data compared to traditional EqP calculations with generic K_d values. Ionic substances or substances capable to be ionised may also react with charged binding sites (i.e., clays, various ligand forming species; Mg, Ca, OH, CO₃ etc.); the ionic strength, type of charge (negative/positive), and the pH dependent K_{oc} , K_{ow} and water solubility are essential.

Regarding sediment characteristics, a major player for organic chemicals is the sedimentary organic carbon (OC) content that can be further divided between amorphous, soft or new and condensed, old or black carbon (BC). This division of types of sedimentary organic matter is especially relevant for non-ionic organics. For monitoring programmes parameters such as grain/particle size, surface area, organic carbon content, organic nitrogen content, iron, manganese, calcium and aluminium should be measured routinely. An understanding of the nature of binding or association of a contaminant with the sediment is critical.

For metals the most critical physical-chemical parameters defining their fate in the environment are the charge/covalence, speciation and bioavailability parameters further depending on environmental characteristics such as pH, redox conditions, organic carbon, acid volatile sulphides (AVS), Fe/Mn oxides, and sediment particle size should be considered. BCR (European Community Bureau of Reference, now known as the Standards Measurements and Testing Program) extraction schemes appear to be usual for metals and organometallics. Such

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

schemes are more problematic or less appropriate for organic substances which partition into organic matter.

Elements for discussion

Should contaminants associated with suspended matter and sediment be considered as the key element for triggering a sediment assessment?
Is partition under equilibrium conditions a proper first tier approach? And for higher-tier assessments? What bounds (see potential limitations identified below) need to be placed on the use of the EqP approach?
Are the parameters indicated above for the different groups of substances adequate or should we remove/add others?

2. Available exposure scenarios and metrics for sediment exposure assessment

Prospective risk assessment

The equilibrium partitioning theory has been mostly used to predict the exposure of benthic organisms to non-ionic hydrophobic contaminants. In the regulatory context stepwise RA approaches are frequently used, starting with very simple and cautious assumptions and models. If no potential risk is identified even for over-protective worst-case conditions, a conclusion of low concern is sufficient for the regulatory decision. Additional data and refinements to more realistic conditions are only needed if low concern cannot be concluded from the simplistic approaches. A tiered assessment process, including triggers for further assessment, is often included in the regulatory guidance.

For prospective risk assessment there are exposure scenarios available (e.g., for the EU biocides and REACH processes), but they are extremely simplistic in nature. Usually emission is implied to be from point (solution) sources and the contaminant is assumed to partition/adsorb instantaneously and homogeneously throughout the sediment layer. Emerging data indicate that the contaminant associates with the sediment less strongly than aged or pre-existent contaminants. Consequently, assessments of risk posed by contaminants spiked into sediments can be considered conservative, when compared to the same contaminants in many contaminated sediments at field locations. No realistic considerations of the fate processes are included. The equilibrium partitioning approach used for biocides/REACH only calculates the PEC in suspended sediment, is not volume limited and is then compared with ecotoxicology data dosed under laboratory test conditions under different exposure conditions (water only exposures for pelagic organisms or the chemical spiked directly to the sediment if sediment toxicity data are available) to that assumed in the model. Most existing models do not consider all major sources of contamination to the sediment compartment, such as sediment transport. Also lacking is exposure to discrete particulates that are contaminants themselves and may serve either as sources of contaminants or be bioavailable themselves. Examples are antifouling paint particles and tyre particulates, that may represent sparingly soluble sources of contaminants (e.g. Cu, Zn), but may transform to more bioavailable and potentially toxic forms with time. The emission of the chemical bound to suspended particles may be also relevant. For example, EUSES assume an emission to water of 15% for a non biodegradable chemical with a Log K_{ow} of 6 and Henry constant not higher than $1 \text{ Pa}\cdot\text{m}^3\cdot\text{mol}^{-1}$; in many cases this value will

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

exceed the solubility limit and most of the emission is expected to occur as particle bound substance.

Regarding active ingredients of pesticides, for which both knowledge on use profile and data availability is generally much higher than for industrial chemicals, at the European level active substances are evaluated using some ten FOCUS (FORum for the Co-ordination of pesticide fate models and their Use) surface water scenarios². Each of these scenarios should apply to the 90th percentile of exposure concentration in a large region. At the time of development of the FOCUS Surface Water scenarios, comprehensive databases for checking this assumption were not available, so it is not clear if the FOCUS scenarios are good predictions of this 90th percentile. EFSA (European Food Safety Authority) describes a consistent methodology for scenario derivation that could be applied to improve the exposure assessment³. At the national level, member states use exposure scenarios as well; for example the Netherlands use a specific scenario because Dutch surface water is relatively vulnerable. Actual fate data is used (i.e., sediment/water study) and values assume homogeneous distribution through the upper 5 cm and the comparison is with ecotoxicology data where exposure through this upper layer is assumed. This seems to equate better with behaviour in a pond system, however there are questions over how this could be applied to continuous releases and whether it is truly representative of aquatic systems.

For regional/continental estimations multimedia models (e.g. MacKay Fugacity Models such as Level III EpiSuite) offer information on the expected relevance of the sediment compartment in the overall distribution of the chemical among the different environmental compartments.

Although environmental fate models have been developed for predicting metal concentrations in various environmental compartments, including sediments, as a function of metal loadings from natural and anthropogenic sources, the prediction of sediment metal concentrations is still problematic. In particular, although the existing models may be conceptually appealing, it is very difficult to validate them against field data⁴. This shortcoming is perhaps not a major problem for risk assessment per se, at least for the data-rich metals (since their ambient concentrations are reasonably well known), but it does constitute a major obstacle for the risk assessment of data-poor metals, and for risk management, (e.g., for deciding whether reducing a particular metal input will in fact result in a meaningful decrease in the ambient metal concentration).

Once metals have reached the benthic sediment compartment, there are three

² <http://viso.ei.jrc.it/focus/sw/index.html>

³ <http://www.efsa.europa.eu/en/efsajournal/doc/2562.pdf>

⁴ Paquin, P. R., R. C. Santore, K. J. Farley, C. Kavvas, K. B. Wu, K. Mooney and D. M. Di Toro (2003). A Review: Exposure, Bioaccumulation, and Toxicity Models for Metals in Aquatic Systems. Pensacola, FL, USA, SETAC Press.
US EPA (2007). Framework for Metals Risk Assessment. Washington, DC, USA, U.S. Environmental Protection Agency, Office of the Science Advisor, Risk Assessment Forum. EPA 120/R-07/001: 171.

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

recognised possible exposure routes: 1) the sediment porewater (for benthic organisms that burrow in the sediment); 2) the water overlying the sediment water interface (for epibenthic organisms; for benthic organisms that burrow in the sediment and create burrows that connect with the overlying water, and through which the overlying water circulates); and 3) the solid sediment (for sediment-ingesting organisms). The behaviour of the organism will govern which exposure route is most important, and the dominant exposure route may change for different life stages or due to different activities of an individual life stage.

The biotic ligand model (BLM) has been used to predict the toxicity of metals as a function of water chemistry, but to date, the application of the BLM to the sediment compartment is extremely limited. Such models are available for some metals, but they have not been widely developed and validated using sediment toxicity data. These EqP-based models predict only the dissolved exposure route and do not discretely consider dietary exposure. Many benthic organisms ingest considerable quantities of sediments, in which bacteria, algae, smaller organisms and organic matter may be the feeding target, whereas many organisms do not exclude other sediment materials.

There are also bioaccumulation and trophic transfer models and simplified calculations, e.g. based on BSAF, for estimating the expected exposure for fish and other organisms feeding on sediment dwelling organisms, but mostly focusing on non-ionisable organic substances.

The metrics for model-based equilibrium partitioning is not an issue as based on the parameters and default assumptions the model results can be expressed in different ways.

Elements for discussion
Are available exposure assessments tools and models sufficient? What additional tools/models should be pursued (accepted for trials)? Is there a need for developing generic but more realistic models for marketing authorisation?

Retrospective risk assessments

In the case of retrospective risk assessments, analytical measurement is usually the starting point for the exposure assessment. Direct sediment analysis can be complemented with or replaced by other tools, such as passive sampling devices that will absorb available concentrations of organic chemicals in a manner similar to, or at least reproducible of, the way adsorption occurs by biota. For non-polar organic chemicals, a very promising approach is to do equilibrium sampling into a polymer. The concentration in the polymer can then be multiplied by the lipid to polymer partition ratios in order to obtain an accurate prediction for equilibrium partitioning concentrations in lipids.

The metrics for defining and presenting the exposure levels are essential in retrospective risk assessments. Different sinks may require different metrics. Assuming that different organisms will be exposed via different routes, it is expected that one single metric will not be sufficient. Normally total dry weight (typically dry weight concentrations in settled matter or concentrations in the water column as an indication) and pore water concentrations are used. Pore water concentrations and total sediment concentrations (or both) will be needed,

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

depending on whether uptake is mainly via body surface (contact) or diet. There is a need to evaluate the state of the science regarding the relative importance of dissolved (pore water) versus dietborne exposure as they pertain to the manifestation of toxic effects, as toxicity is currently the predominant basis of risk assessment (unless food chain effects are evaluated). When both should be quantified, it should be considered that addressing dietborne exposure is not simple. How should dietborne exposure be quantified?

It is probably a good general target to use the same metrics as far as possible. However, if there are good reasons to use different/several metrics, for example because of differences in organism habitat use and/or life history characteristics or behaviour of the substances, then this should be performed. For example, the concentrations in suspended matter may be very useful for considering transport or exposure to certain species (e.g. filter feeders such as oysters) that live near but not in sediments. Normalising concentrations of organic chemicals to a fixed OC content reduces the variability but does not correct for differences on the nature of the OC.

When dealing with bioaccessibility/bioavailability, at the moment it appears that the best metric is the freely dissolved pore water concentration, both for metals and for organic chemicals. Note, however, that when using pore water concentrations to quantify sediment exposure, the assumption may reflect the conditions for uptake via the dermis, while for ingested sediment the physical-chemical conditions in the digestive system may affect the distribution. Therefore the geochemical environment of the digestive system must be characterised in order to begin modelling speciation and activity. However, as indicated above, different organisms may experience bioavailability in different ways.

Passive samplers can be used to determine freely dissolved pore water concentrations of organic compounds. Because good relationships with body burden concentrations have been observed there is a tendency to assume that all uptake is through the free dissolved phase, but this is not necessarily the case. The good relationships are found because the freely dissolved concentrations are directly reflecting the chemical activity in the system. Uptake by a passive sampler can also be transferred to a lipid basis; an excellent parameter for assessing exposure. The result is basically the lipid based concentration as if the organism was in equilibrium with the sediment. This approach can be used as (complementary) exposure metric in laboratory bioassays for reducing interlaboratory variability.

Regarding metals, the technique of diffusive gradient in thin films (DGT) is becoming increasingly used as a useful in situ passive sampling device for providing information of metal bioavailability at the sediment-water interface and in deeper sediments (the use of such tools need to be encouraged, so that they can be better developed and scope and limitations clearly identified).

It is generally recognised that total metal concentrations are often poor predictors of metal bioavailability and the risk posed by metals and also for metals, the best metric for bioavailability appears to be the the freely dissolved pore water concentration. For the assessment of metal bioavailability, the technique of diffusive gradient in thin films (DGT) is becoming increasingly used as a useful in situ passive sampling device for providing information of metal bioavailability at

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

the sediment-water interface and in deeper sediments (the use of such tools need to be encouraged, so that they can be better developed and scope and limitations clearly identified). Costello et al. (2012) showed that DGTs are useful in tracking the flux of Ni from sediments and among different binding phases, but DGT-measured Ni poorly predicted the invertebrate response to metal. Biodynamic model proposed by Rainbow and Luoma (2005) is a good starting point for assessing the dietborne exposure to metals.

For metals, the bioavailable fraction is a function of sediment AVS in anoxic sediments, with OC and Fe/Mn becoming increasingly more important in oxic/sub-oxic sediments. These factors and other sorptive phases like organic carbon and Fe/Mn oxides, and explain the majority of variability in toxicity to infaunal benthic organisms among sediments with different chemistries. Freely dissolved concentrations can be calculated from pore water concentrations if the partitioning coefficient KDOC and the DOC concentration are known or can be determined directly by measurements (e.g., using passive samplers). Chemical activity can be determined using speciation models, and this requires quantitative characterization of pore water geochemistry. It is very important to note that pore water geochemistry will vary substantially from that of the overlying water, e.g., higher DOC, different pH, higher hardness etc. At the sediment water interface, fluxes of metals (e.g. DGT) can be a particularly useful measure of potential metal exposure.

Based on the discussions on bioaccessibility/bioavailability above, there is good reasoning behind the use of the pore water concentration as the main metric to assess exposure; however, this is not often performed in ecotoxicological studies and may be difficult to perform due to specific testing conditions (e.g., the use of artificial sediment) and the lack of operational definitions and harmonised methods. In general but particularly for many metals, accurate pore water measurements can be difficult as the separation of pore water from sediments often results in changes in pore water concentrations, and pore water metals concentrations often displace large gradients in the top 1 cm of sediments due to redox gradient changing the dominant metal-binding phases (e.g. from Fe/Mn-oxides, that are then reduced and dissolved releasing metals, and then metal sulphide formation). The important consideration in the choice of metric is that the conceptual model used in the exposure estimation is complementary identical to that used in the ecotoxicological testing. Therefore, it is assumed that if the PEC is determined in suspended matter (representative of the upper few mm of sediment), then exposure in the ecotoxicology study should be performed with an equivalent material or, if not, it should be considered whether the effect measure could be normalised relative to the exposure measure. Alternatively to the later it should be considered how cautious the approach will be if different metrics for effects and exposure are being used. The metrics chosen may differ depending on the test species because for some species pore water is most likely to be the dominating exposure route, whereas for other species (sediment ingesters) dietary contribution to the pore water exposure route may be significant. At present under biocides/REACH, detailed guidance is only provided for the first tier, which is simplistic and precautionary in most cases, but not applicable to certain substances/conditions. The mismatch between the exposure used in the ecotoxicology studies (uniform distribution through the sediment) and the assumptions used for the calculation of PECs (suspended matter, upper mm), need to be solved when alternative or higher-tier methods are required.

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

In conclusion:

1. There is not one "universal" metrics covering all cases, thus the metrics to be used in the exposure assessment should be discussed in the conceptual model and aligned with the metrics to be used in the ecotoxicity assessment. Different taxa may require different metrics and even the same taxa may require complementary measurements if several exposure routes are relevant
2. Good metrics expressing the potential for partitioning are:
 - Measured freely dissolved concentrations
 - Measured chemical activities
 - Equilibrium partitioning concentrations in a polymer reference phase
 - Sediment-water interface fluxes (e.g. DGT technique).
 - Predicted equilibrium partitioning concentrations in lipids
3. There are situations where it is necessary to obtain (1) C_{total} , (2) freely dissolved concentration (or chemical activity) and (3) (bio)accessibility (or readily desorbing concentration).

Elements for discussion

Do you agree with the conclusions above?
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If not, which modifications/additions/deletions are needed?

3. Accounting for bioavailability and degradation/dissipation in sediment exposure assessment

Research during the last decades has led to several competing concepts of bioavailability and to many more methods to estimate bioavailability. One reason for disagreement is the confusion of two fundamentally different parameters, accessible quantity and chemical activity. The accessible quantity describes a mass of contaminants, which can become available to, for example, biodegradation and biouptake. It can be determined with mild extraction schemes or depletive sampling techniques. The chemical activity, on the other hand, quantifies the potential for spontaneous physicochemical processes, such as diffusion, sorption, and partitioning. For instance, the chemical activity of a sediment contaminant determines its equilibrium partitioning concentration in sediment-dwelling organisms, and differences in chemical activity determine the direction and extent of diffusion between environmental compartments. Chemical activity can be measured (estimated) with equilibrium sampling devices and, theoretically, is closely linked to fugacity and the freely dissolved concentration. The terms "bioaccessibility" and "bioavailability" as defined in human health toxicology can have relevance here if dietborne exposure is discussed (true bioavailability as a measure of the fraction of exposed substance that is ultimately absorbed in the blood or equivalent stream); nevertheless, there is not an equivalent level of information on toxicokinetics in sediment organisms.

Therefore, there is also a need for differentiating bioavailability from bioaccessibility. Bioavailability can be defined as the fraction of contaminants that are available for uptake by an organism of interest, and therefore is organisms specific, the time scale is also relevant, e.g. in relation to food

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

ingestion. Bioaccessibility is how much of a contaminant is accessible but not necessarily how much is assimilated into the organism. Although bioavailability assessments would be more preferable, the only real possibility for measuring bioavailability is by exposing relevant species to sediment (laboratory and/or field) and obtaining the tissue residues and toxicokinetic estimations– or if bioavailability is defined as a parameter related to toxicity to measure the chronic toxicity of the organisms. In the case of tissue concentrations, often the bioavailable fraction may not be correctly estimated as a considerable portion of bioavailable fraction may have already been ‘processed’ by the organism, and only the fraction that remain present in internally bioavailable forms is important (e.g. metals that have been accumulated slowly and stored as granules etc are often considered as non-bioavailable). Bioaccessibility can be measured using other techniques. The BCR extraction approach could be used for some chemicals to try and link solid-state speciation with bioaccessibility. What is really needed is a unified scheme to evaluate bioaccessibility using more representative biological fluids in validated and standardized new extraction methods (representing the effect of e.g., dietary tract, enzymes, proteins and surfactants).

A method that tries to incorporate (mechanistically) an important basic phenomenon of the system, and which can also be cost efficient, is the estimation of freely dissolved pore water concentration. This can be performed either through models (EqP and BCF) or with chemical analysis for organic substances, but is more challenging for metals. The equilibrium between water/sediment/suspended matter is not an easy endpoint to address and the bioavailability of the substance is dependent on the equilibrium between the different matrices. Many scientific papers confirm the view that with non-ionic organic substances, freely dissolved concentrations in pore water are a good estimate for a pool that will attain steady state with biota through bioconcentration. This, of course, is modified by the organisms potential ingestion of the sediment and its gastrointestinal processes, biotransformation, as well as the duration of exposure. Passive samplers⁵ (PSDs) are very promising tools for the detection of freely dissolved concentrations of organics. Solid phase microextraction (SPME) is one application but it has limited sampler volumes. A larger volume is offered by coated vials and polyoxymethylene (POM) or polyethylene (PE) film. Some legacy contaminants can be sampled with passive samplers but new emerging substances need experiments. However, in addition to detecting environmental concentrations of active substance fraction, passive sampler concentrations can be converted to lipid based biota concentrations allowing direct effect assessment relying on empirical data from tissue residue approach (or CBR).

Passive sampling is an excellent tool to measure the bioaccessible fraction of

⁵http://c.ymcdn.com/sites/www.setac.org/resource/resmgr/publications_and_resources/executivesummarypassivesampl.pdf
http://www.norman-network.net/index_php.php?module=public/workshops/ispra_2012_pdf&menu2=public/workshops/workshops

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

sediments or suspended particulate matter (SPM) as well as the chemical activity (or the freely dissolved concentration), allowing bioavailability estimations. Passive samplers are also a good tool to give information on the potential for bioconcentration. This has been well demonstrated for legacy organics but their application for assessing exposure of emerging substances needs additional experimentation. However, it is also important to recognize that passive samplers do not reflect biomagnification or metabolism. Not all passive samplers are similarly suitable for measuring the bioaccessible fraction, the chemical activity/freely dissolved concentration or for estimating potential bioconcentration. PSDs based on partitioning of the chemical into a sampler consisting of one phase should probably be preferred. The questions for which PSDs are suitable and mature enough require clarification. Of high importance is standardisation of the passive samplers relative to the materials and the configurations used; e.g., the various polymers (or the varieties of one polymer) utilised show different partition coefficients for a given substance. Also calibration procedures require standardisation. The costs for PSDs, including preparation, extraction and clean-up of the PSDs, vary significantly between the different types of PSDs. Recommendations for introducing PSDs in sediment exposure assessment should take the costs into account. One shortcoming of traditional exposure assessment lies in sampling strategies that are discontinuous in time. Sampling is conducted in certain time intervals. Concerning sediments, this is not a problem if the surrounding conditions are more or less constant (i.e., concentration in water and depositing material). Usually, sediments are integrative over time and changes in contaminant concentrations are slow. However, concentrations in water can alter quickly. For water sampling, passive sampling resulting in time-weighted average concentrations can therefore be advantageous and constitute a more realistic exposure to biota. Note that passive samplers can assess porewater concentrations, which are linked to aqueous exposure, but cannot directly account for the uptake flux through particle ingestion.

In the regulatory context, if a PEC is calculated based on the assumption of 100% bioavailability, and a PEC/PNEC ratio of <1 is obtained, then basically there may not be a need to account for bioavailability any further. However, if the result is >1 , then the exposure assessment (e.g. release rate) of the substance should be refined and/or the bioavailability can be estimated based on physico-chemical data of the substance, monitoring data (if available) and properties of the environment. Bioavailability may depend on environmental conditions such as the content of organic matter in the sediment. Therefore, for a given compound, it can be appropriate to derive PECs specifically for different environmental conditions (such as sediment types). PECs can be calculated for different time periods after the beginning of the exposure, to account for the change in bioavailability over time. Modelling is acceptable for some situations; however not all chemicals exhibit partitioning behaviour according to theoretical models. For example ionic organic compounds can exhibit complex partitioning that needs to be better understood before its behaviour can be modelled. A bioavailability assessment approximated through the pore water concentration modelled using the Freundlich equilibrium equation is incorporated in most European pesticide fate models. To account for non-equilibrium sorption and ageing, rate-limited processes may be included as well if available.

For metals, in situ techniques such as DGT are increasingly being found to

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

provide useful information on labile metal concentrations (but like all techniques the limitations still require further examination). Furthermore, the bioavailability of some metals (Ag, Cd, Cu, Cr, Ni, Pb, Zn) can be modelled with SEM-AVS approach but the promising sediment Biotic Ligand Model (BLM) needs more demonstrations. The SEM-AVS approach for characterizing the bioavailable fraction of metals in the sediment includes: (1) consideration of organic matter content; (2) acid volatile sulfide (AVS) and simultaneous extractable metals (SEM). AVS in the sediment reacts with dissolved metals to form an insoluble metal sulphide. AVS and SEM are operationally defined terms and refer to the sulphide and metal fractions that are released upon a cold, weak acid extraction. The metal sulphide form is considered non-bioavailable to benthic organisms via the dissolved route. The amount of AVS in sediments, which is determined by the anaerobicity and the availability of sulphur (including that coming from decay of proteins of dead biota) in the sediment, serves as a critical parameter in determining metal bioavailability and toxicity in sediments. Metals, in essence, will exist in the form of their respective metal sulphide if the AVS is present in excess of the reactive forms of the sediment metals (SEM) (and the AVS is accessible to the metals in question), and as long as anaerobic conditions persist. On the other hand, if the total concentration of the metals is greater than the concentration of the AVS, then potentially, some fraction of the metals may exist as bioavailable metal, and then other complexation processes in the pore water (which will vary by metal) becomes the relevant process. Other solid phases (organic carbon, Fe/Mn oxides, clays) also bind metals, so the probability of risk under situations where SEM>AVS will vary according to these parameters. One limitation of the SEM-AVS approach is that it is not a predictive tool. The Ni case study will offer an illustration for how a predictive bioavailability model can be incorporated into a sediment risk assessment framework. Briefly, bioavailability models are developed for representative benthic organisms. These are then used to normalize sediment toxicity databases. Normalized ecotoxicity data are then used to populate a species sensitivity distribution, which can then yield site/region-specific PNEC values. A key consideration in any bioavailability-based approach is the need to collect data on the distributions of factors that affect contaminant bioavailability. For metal compounds, these should when possible include parameters like AVS, and organic carbon, Fe and Mn oxides and particle size.

In conclusion, the term bioavailability is defined in many different ways. The following is an attempt at a definition and delineation of bioavailability and related terms. The total concentration of a chemical in a sediment can be divided into an irreversibly bound pool (i.e., non-extractable, bound residues), reversibly bound, and freely dissolved pool. The reversibly bound and the freely dissolved pool constitute the (bio-)accessible pool. Accessibility is operationally defined. The accessible pool defines the fraction of the total concentration that can undergo degradation, be mobilised or taken up by organisms. However, it is a poor measure for the actual diffusion, partitioning or uptake process, which is rather driven by the freely dissolved concentration or the chemical activity. The chemical activity as well as the freely dissolved concentration can be measured with equilibrium passive sampling devices. Bioavailability is linked to (bio-)accessibility and to the freely dissolved concentration (or the chemical activity). Bioavailability also includes the uptake of a chemical by the organisms. Hence, bioavailability is not only driven by the characteristics of the sediment, but also dependent on the organism.

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

Elements for discussion

Do the discussions above provide good definitions for the bioavailability and bioaccessibility concepts? If not, what additional concepts should be considered? Are PSDs a suitable tool for exposure assessment? Can PSD results be compared with ecotoxicity data? How?

Is it possible to define default conditions for modelling bioaccessibility/bioavailability in generic risk assessments, e.g. at the ecoregion/ecotype level?

Degradation or dissipation half-life should be estimated and estimated half-life can be used in PEC calculation. It is important to evaluate how realistic the half-lives derived from laboratory experiments are in the field conditions. Actual degradation (rates) may be difficult to predict, as it depends on site-specific conditions (i.e., temp, pH, redox, bacterial community). When the dissipation half-life is used in PEC calculations, the fate of the substance should be known (e.g., binding to organic matter) and it should be considered whether the substance is transformed permanently or adsorbed (and whether it can be desorbed). Transformation products should also be considered as far as possible. For example, if there are intermediate or persistent/recalcitrant transformation products that can be of concern, they should be considered in the risk assessment. Some chemicals can exhibit a very fast half-life in water and tissue and a much slower half-life in sediment, which can also be highly variable depending on redox state. Many contaminants can last for years in sediment and may become bioavailable when disturbed or bioturbated. Degradability should be incorporated as it is a major factor, however, standard methodology available (e.g. OECD test guidelines and even "simulation" degradation guidelines such as OECD TG 307, 308, 309 & 314 protocols) are not in most cases really simulating the true environmental conditions in particular in relation to a environmental realistic low substance concentration, temperature and microbial species diversity over longer time frames. Furthermore often only one or a few degradation half-lives are being obtained with such methods and it is often not known how well test data represent the environmental variability impacting real environmental degradation rates. Hence in conclusion even "simulation" degradation test guidelines provide still highly uncertain results, which require normative decisions to be made in relation to their use in a regulatory context. Anyway fate models typically use degradability half-lives to estimate steady state sediment concentrations.

Some organisms have the ability to detect and avoid contaminated sediments, and as a consequence, healthy benthic communities may exist in sediments that have 'avoidable' areas (of varying size) that contain elevated concentrations of bioavailable contaminants. The consequences of short to long-term avoidance of contaminated sediments needs to be better understood.

For metals degradability is not an issue but speciation, ageing, burial below the depths that are explored/inhabited by benthic organisms and other relevant processes should be considered if feasible. Long term distribution processes occurring in sediments are not accounted for in short term laboratory toxicity tests. The contaminant avoidance discussed in the paragraph above has been shown to occur in metal contaminated sediments. Sediments with high metal concentrations may be avoided by larger bioturbating organisms and this will

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

reduce the frequency that oxygen mixes with deeper sediments, and consequently results in higher AVS concentrations and lower dissolved metal concentrations. While the dissolved metal exposure is now lower (due to metal-binding to AVS), many organisms may also avoid sediments with high sulfide (AVS) concentrations. For those that don't avoid sulphide, they will create oxidised niches (e.g. as exist around all burrows) and metals the metals may be more bioavailable near the organism. This leads into the need to consider exposure variability.

Regarding monitoring programmes, degradation processes are less relevant for sediments because of the long residence times in sediments, the compounds that are degradable will have "disappeared" from the matrix. When including bioavailability measurements, degradation is automatically accounted for however. Measurements can be repeated in time to provide information on rates. It is essential that measurement do not disturb the system and to ensure that the bioavailability measured in the samples really represent the bioavailability under real conditions.

The formation of metabolites should be considered. Metabolites should be assessed for their toxicity, their bioaccumulation potential, and their persistency. For metals such as mercury, the formation of organometalic compounds is relevant.

Elements for discussion

Should lab degradability data (or equivalent data for metals) be used for modelling PECs?

Should long-term chemical and diagenetic processes (e.g., speciation, ageing, burial) be explicitly considered in estimating biodegradability of sediment contaminants?

How this processes as well as the formation of metabolites should be addressed for allowing comparisons with effect data?

4. Exposure assessment variability, uncertainty and prioritisation

This is very challenging, as both contaminant concentrations and parameters that influence bioavailability (e.g., AVS, POC forms) exhibit considerable temporal and spatial variability. Additional aspects like seasonal and tidal fluctuations in hydrology which influences sedimentation, remobilization, resuspension of the sediment have to be also taken into account. It is a well-known fact that there could be a huge difference in sediment concentrations, within a very fine temporal and spatial grid. This has previously been reflected in the scientific literature, especially concerning legacy chemicals. In practical terms the spatial and temporal scales should be designed according to the risk assessment objectives, and should be coherent with the effect assessment, ensuring the optimal resolution of the spatial and temporal coverage of the risk characterisation output. For regulatory purposes, the goal should be to address realistic worst case scenarios, sufficient for decision making. Exposure scenarios/models should take into account the factors that affect the temporal and spatial variability in release, transport, ageing or any other factors that affect the exposure concentrations. The frequency, spatial distribution and type of emissions (continuous/not continuous) should be documented. In general, it can

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

be useful to obtain several PEC values (e.g., based on maximum and average emissions, or for different sites or areas), for risk assessment. Uncertainty related to each parameter used for calculation should be estimated and documented as far as possible. If the reasons for temporal variability are known, the timing of the assessment can be placed correctly (e.g. during the high exposure). Also, one approach would be to concentrate on season/time when the most vulnerable receptors are under exposure (e.g. fish eggs vs. adult fish). Acid Volatile Sulphides concentrations clearly depict a seasonal variability. Lowest concentrations are typically measured in spring season. Hence assessments should be performed at realistic worst case conditions prevailing in spring. Some indicator of overall uncertainty for PEC could also be useful. Quantitative uncertainty analyses using tools such as Monte Carlo simulations could provide significant insights.

Spatial variability calls for proper sampling plans that consider source location, relation to vulnerable/valuable areas, gradients, sample number in relation to precision and cost efficiency. Assessors need to consider first the ecological relevance and then statistical significance vs. practical significance and possibility for stratified or composite or even adaptive sampling (instead of systematic). Most variability presently measured for organic chemicals in sediments are related to differences in composition of the sediment sampled in time or place but does not reflect the variability of the exposure level. Unless the system is very dynamic and close to pollution sources the exposure level in terms of chemical activity of organic chemicals (through passive sampling) will not vary a lot.

For pesticides, the current exposure models can simulate the time dependence of the exposure concentration in sediment (i.e., dissolved and total). Spatial variability could be assessed using a spatially-distributed modelling approach. Although there have been some initiatives for performing such a spatial analysis⁶, there is no commonly agreed methodology for pesticides available. Uncertainty of the exposure concentration leads to a shift towards higher exposure concentrations⁷. Uncertainty could be dealt with using a Monte Carlo approach, but EFSA proposes a simpler procedure. This procedure comes down to using a higher spatial percentile in the exposure assessment.

Several sampling paradigms have been proposed to derive data representative of large areas of contamination (particularly where the contamination may not be uniform). Several paradigms have been proposed using applications of GIS to derive weighted estimates and median values and to sub-divide large contamination areas into smaller geographic units. In many cases (especially many non-polar organic chemicals), tissue concentrations can be very important for exposure estimation because they integrate ambient concentrations over time and space. In other cases, a large number of samples taken according to a statistically robust sampling design are required for sediment or porewater to adequately characterize temporal and spatial variability. It is essential to rationalize sampling in terms of good experimental/monitoring design during the

⁶ e.g. www.eu-footprint.org

⁷ e.g. <http://www.efsa.europa.eu/en/efsajournal/doc/2562.pdf>

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

problem formulation stage, unfortunately this is too often overlooked.

The best alternative is using probabilistic approaches but, if this is not feasible, peak vs. average exposure estimations should be presented, and a time-weighted approach can be considered as the third option. For pesticide RA, we usually would calculate both and decide depending on available toxicity data which PEC is the appropriate one for the RA. Usually for pesticides, peak concentrations are of high relevance due to the pulsed exposures, however, considering accumulation in the sediment, TWA-concentrations might become more important. In any case, the concentrations in sediments do not vary with time as drastically as in the water column. Thus, threshold values should be built up to protect the benthic environment from chronic effects. Also, it might be a good practice to define site specific receptors with the most vulnerable time windows when special attention could be paid, e.g. with lower threshold values.

Priority is subject to the scope of the particular investigation being conducted and also depends on the chemical. Legacy substances are probably a threat for higher trophic levels through at long term steady exposure and even with decreasing levels in sediment. New, less monitored and less regulated substances can vary in quantities discharged and may be less persistent and thus, should especially be monitored more frequently until a better understanding of their behaviour and fate in both water and sediment is achieved.

Maximum (or worst-case) PECs can be used as a first check and if these do not indicate unacceptable risk, then there is not necessarily a need to refine the PEC any further (for regulatory purposes). If the worst case scenario shows possible risks, we have to investigate at a more precise level. If the aim is to protect the community it is necessary to consider the life cycle of the organisms. There are some periods of the organism's life cycle which are more sensitive to contaminants. In these cases the priority is to measure peak concentrations. Higher tier risk assessments should strive to replicate reality to the extent practical. That dictates preferences in risk assessment to characterize "reasonable conditions" and "reasonable-worst-case conditions" and to consider exposure scenarios that are plausible. In the case of monitoring programmes, maximum values may be reflecting outliers and errors and will discourage sampling: the more data, the higher the value. For chronic, long term assessments, averaging seems reasonable. Considering the "recovering capacity" of ecosystems this pragmatic approach can be considered as acceptable. However, peak-exposures may also need to be considered because of their acute ecological relevance (e.g. 90th P?). For remediation purposes one should aim for realistic concentrations, as worst case estimates might lead to unnecessary remediations, and averaged concentrations might miss hot-spots that do should be prioritized for remediation. Such a realistic assessment would be best obtained by performing space-resolved bioavailability measurements in sediments, e.g. using passive samplers when appropriate. Based on such measurements, hot-spots might be identified (maximum concentrations) that could selectively be remediated and locations that do not need remediation.

Elements for discussion

Are current generic regional scenarios (e.g. REACH and EU TGD) and models (e.g. fugacity models) sufficient for regional exposure estimations?
Should the ecoregion and ecotype concepts be incorporated as refinement

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

options?

What are the best metrics/strategies for peak & short-term exposure assessments?

What are the best metrics/strategies for long-term exposure assessments?

How should spatial variability (patchiness and hotspots) be best considered?

5. Use of monitoring/field data and role of site specific emissions and assessments

Monitoring and field exposure data are valuable tools for retrospective risk assessment and sediment quality assessments. It helps to identify the general status of a specific concrete environment and characterise possible threats. Monitoring and field data can also be invaluable in determining the realism of a risk assessment, provided that the field data include measures of biological effects that can plausibly be linked to exposure to a particular contaminant. This has proved difficult in the past, as the link between exposure and effect is difficult to establish in field situations where mixtures of contaminants are normally present and many non-contaminant factors can modify exposure and confound interpretation of impacts. In this context, recent work in the toxicogenomic field shows some promise that we will be able to identify particular transcriptomic "signatures" that can be linked to a particular contaminant and ideally linked as well to ecologically relevant outcomes/effects.

It is more difficult to use monitoring for prospective risk assessment in which you have to authorise or deny the environmental release or use of a certain chemical. The source of the chemical that you are tracing can vary enormously and can even be a metabolite of other contaminants. The empirical approaches that involve laboratory and semi-field studies are much more reliable tools in risk assessments, but may not address some processes occurring in the technosphere. Nevertheless, field data are invaluable in terms of validating models, approaches, and conclusions that are derived from laboratory observation. When dealing with a site specific risk assessment the situation varies. In this case, it would be very valuable to assess the general quality of the receiving environment, the species present, the physico-chemical properties and many other factors that would help to understand the potential effect of a chemical in that specific environment. However, monitoring/field data can, if performed in a comprehensible way covering the environment of interest sufficiently both in time and location, be used in order to calibrate Multimedia Fate Models, to estimate partition coefficients, to identify local sources, etc., and provide essential validation to any desk-top predictions which are typically fraught with uncertainties. If there is field monitoring data available for the same substance or similar substances, then this information could possibly be used to evaluate the reliability of the PEC calculations or exposure scenarios/models. It should be noted that field data and monitoring cannot always be assumed to be representing the actual environmental exposure and that such data also includes cofounders (e.g. contribution the chemical concentration of a substance from degradation processes of other substances and from other natural and anthropogenic sources and processes). Field data and monitoring data can however be used in a weight of evidence (WoE) approach when evaluating single substances in a regulatory context. Results from synoptic measurements on biology, chemistry and ecotoxicology as measured with the TRIAD approach could

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

be used as a kind of validation. Site-specific information should be taken into account when it is considered to be affecting the behaviour or fate of the substance. Monitoring, if done only in the water column, needs to include substances expected to distribute into sediments. After that a simple multimedia model could estimate concentrations in sediment. Even better would be a scheduled sampling and analysis of sediments. Of particular importance are monitoring data for incorporating bioavailability-based approaches (e.g., AVS for metals, passive samplers for organic chemicals) and data that can be used to validate model-based exposure estimates. Monitoring and field exposure data can also be used in calibrating site-specific models that are often applied to predict future risks and in evaluating various management options for risk reduction.

In a standard risk assessment approach (e.g., under REACH), monitoring is a higher tier methodology and limited to substances already on the market. For site-specific assessments, however, monitoring is key in an integrated (e.g., TRIAD) approach.

Sediment serves as "an archive" for many particle associated contaminants and includes all possible routes of depositing material at that specific site. Source characterization and control is essential, not only in exposure assessment but also in possible remedial actions. The quantity of the emissions from point sources and diffuse sources should be estimated (as far as possible) and if both are considered relevant, then they should both be included in the exposure assessment. To include point source discharges and diffuse sources, one would need to be able to model contaminant transport from these sources to the sediment compartment, and thus evaluate their contribution to sediment metal concentrations. As mentioned earlier, such models exist in conceptual form and as computer programs, but they have been seldom validated. This is a challenge for modelling (to be dealt with via emission data/loads, area size etc). Measuring will integrate this automatically, that is, if the space resolution is high enough to catch point sources. If the location of a point source is known, the campaign can be tuned to this. If point sources cause local concentrations to exceed SQCs, local remediations should be performed. The linkage of point and diffuse sources to exposure concentrations and bioavailability in sediment has been performed using mass balance models. Screening-levels models for organic chemicals and metals are available and have been used in generic risk assessments. Site-specific models have also been developed and have been used in support of many regulatory programs. Transport of contaminants from the point source or diffuse sources requires models coupling hydraulic processes (e.g., advection, dispersion of water and SPM), sedimentological processes (e.g., deposition/resuspension) and chemical processes (e.g., adsorption/desorption). From a modelling point of view, both point source and/or diffuse sources can be included as input data in such integrated models.

Current practices for marketing authorisation risk assessments focus on generic assumptions and scenarios. For industrial chemicals, the sediment compartment in the present is assessed with data from the water compartment. It is evaluated in a local scale, where the sources are well identified and at regional scale where the sources are more diffuse but in terms of the calculation, the characteristics of the receiving sediments are always the same. For pesticides, diffuse sources including spray drift, drainage and run-off are included in the FOCUS SW

**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

scenarios and models. Point sources are not included in the exposure assessment for pesticides as the starting point of the assessment is good agricultural practice.

Elements for discussion

Monitoring and direct assessments are essential tools for retrospective assessments. What is the added value of applying modelling tools if measurements are available?

Current practices do not include diffuse sources (e.g. soil run-off and drainage) as relevant emission sources for industrial chemicals and consumer products. Should these emissions be considered?

Are sediment exposure models based on equilibrium/fugacity sufficiently developed and validated to be used as generic tools in the regulatory context?

Are sediment exposure models based on non-equilibrium dynamics and processes sufficiently developed and validated to be used as generic tools in the regulatory context?

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**Topical Scientific Workshop on
Risk Assessment for the Sediment Compartment**
7-8 May 2013

Annex. Questions for TOPIC 2: Exposure assessment

1. What are the main processes to be considered when assessing the environmental fate and transfer of chemicals from water to suspended matter and sediment for local/regional and generic/site-specific⁸ assessments ?
2. Which physical-chemical parameters are essential for a proper exposure assessment in the sediment compartment, covering release, environmental transfer/partition and within sediment distribution processes (e.g., redistribution, partitioning, ageing, etc.) for the following: metals, non-ionic organic chemicals, ionic organic chemicals, organometallic chemicals, polymers, and nanomaterials?
3. Which exposure scenarios⁹ are currently available for predicting the exposure of sediment organisms including the epi-benthic community?
4. What are the best metrics for quantifying sediment exposure? (e.g. total dw or ww sediment concentration, suspended/settled matter concentration, pore water concentration). Is one single metric sufficient or do different ecological receptors require different exposure estimations and/or different metrics?
5. How should bioavailability be accounted for in sediment exposure assessment?
6. How should degradability and dissipation be considered in sediment exposure estimations?
7. How can temporal and spatial variability and uncertainty be assessed and expressed in sediment exposure estimations?
8. Which exposure estimations should be prioritised? For example, maximum vs. averaged concentrations, realistic vs. worst-case concentrations, peak vs. averaged concentrations, time-weighted averages adapted to the ecological receptor?
9. How can monitoring and field exposure data be used in local/regional and generic/site-specific⁷ sediment RA?
10. How to include point source discharges and diffuse sources in the sediment compartment exposure assessment?

⁸ Site-specific include the quality status assessment of defined water bodies.

⁹ Exposure scenarios include the environmental release and fate processes allowing quantitative estimations of the expected sediment concentration (including spatial and temporal variability and uncertainty)