

Report to the Connecticut Department of
Energy and Environmental Protection

on

**The Draft Proposed Program Outline
for a Transformed Cleanup Program**

**Topic: Remediation Standard
Regulations Evaluation
Soil and Sediment**

November 20, 2012

Submitted to Support the Transformation of
Connecticut's Cleanup Program

Table of Contents

Introduction	1
Workgroup Membership	1
Workgroup Meetings	2
Report Organization.....	3
DIRECT EXPOSURE CRITERIA SUBGROUP SUMMARY	3
I. Background	3
II. Discussion and Recommendations.....	4
POLLUTANT MOBILITY CRITERIA SUBGROUP SUMMARY.....	7
I. Background and Recommendations	7
II. Discussion	8
SEDIMENT SUBGROUP SUMMARY	12
I. Background	12
II. Recommendations.....	13
III. Discussion.....	14
Figures.....	14
Appendix A.....	19
Appendix B	24
Appendix C	31

Introduction

The Department of Energy and Environmental Protection (DEEP) is working to improve Connecticut’s cleanup program through an interactive stakeholder process. As part of the transformation of the statutory and regulatory components of the cleanup program, DEEP solicited volunteers for and formed six transformation workgroups. DEEP asked these workgroups to comment on and make recommendations regarding certain aspects of the transformation, as summarized in the [Draft Proposed Program Outline for a Transformed Cleanup Program](#).

This transformation workgroup was asked to provide DEEP with comments and recommendations regarding the evaluation of the Remediation Standard Regulations (RSRs) in relation with the two soil remediation standards: Direct Exposure Criteria (DEC) and Pollutant Mobility Criteria (PMC). In addition to the evaluation of soil remediation standards, the workgroup was tasked with assessing the possibility of including sediment remediation within the RSRs.

Comments and recommendations contained in this report are the opinions of the workgroup members. Care was taken to identify areas where consensus was not reached among workgroup members.

Workgroup Membership

Workgroup 5 Members:

Participant	Representing
Larry Hogan (Co-Lead)	AECOM
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Benjamin Rieger	AnteaGroup
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Workgroup Meetings

The workgroup spoke about how the DEEP could modify the RSRs and recommend changes in the statutes as they pertained to the following elements of remediation: DEC, PMC and sediments. In general, while the workgroup did not reach consensus on all of the topics discussed, there was overall concurrence that significant progress could be made with respect to streamlining the regulatory requirements and allowing for additional flexibility within the governing statutes and regulations, while still preserving overall protection of human health and the environment.

The workgroup first met on October 10th, 2012. At this meeting the group discussed the scope, deliverable deadlines, ground rules and set all future meeting dates. Preliminary discussions began on the topics of RSRs DEC and PMC revisions. The group decided to table sediment discussions until additional information was gathered.

The workgroup next met on October 15th, 2012. The group discussed the August 2012 RSR Lean Event Concepts and how it related to the prior meeting discussions regarding the DEC and PMC. Additional discussion occurred as a group for DEC and PMC evaluation and associated revisions. Considerable time was spent on discussions regarding institutional controls. The group also reopened discussions on recommendations for sediment evaluations.

At the October 22, 2012 meeting, the workgroup began general discussions on prioritizing DEC, PMC and sediment revisions. The group agreed on forming three individual subgroups to perform a more detailed evaluation of each of the three topics. Each subgroup began to outline and prioritize revisions and recommendations on their respective topic. Also, each subgroup identified and assigned research deemed necessary.

On November 2, 2012, each subgroup reported on the general discussions and recommendations of their respective topic and discussed some of the results of their research. The subgroups broke out and continued to work on concurrence to specific recommendations and began to outline slides for the public presentation.

During the workgroup meeting on November 5, 2012, the public presentation was discussed and slides were finalized by each subgroup. Additional discussions as a group centered on writing assignments for the final report.

On November 15th, the workgroup as a whole revisited and discussed DEC topics and final writing assignments related to the DEC and Sediment recommendations. The workgroup discussed the timeline and agenda for the final meeting.

At the November 19th meeting the workgroup reviewed the draft report and associated Appendices. Additional edits and clarifications by the workgroup were incorporated and the report was finalized.

Report Organization

The Connecticut RSRs protect human health and the environment with two sets of soil cleanup standards, the DEC, designed to protect citizens from exposure to soil pollutants and the PMC, meant to protect groundwater resources from polluted soil leaching. Through almost 17 years of experience working with the RSRs, the workgroup acknowledges that remedial activities performed to date to meet these criteria have improved Connecticut's environment, but the rigidity and cost for compliance are often unattainable. Often the fundamental assumptions for the criteria do not fit the specific setting and use of the Site. Our recommendations offer greater flexibility and a larger toolbox of solutions using established risk science endorsed by the Federal EPA and successfully employed in other eastern States. Summaries from workable programs in New Jersey and Massachusetts are included as Appendices.

The soils workgroup was divided into 3 working subgroups: DEC, PMC, Sediments. This report summarizes the work group's recommendations for each of these groups.

DIRECT EXPOSURE CRITERIA SUBGROUP SUMMARY

I. Background

The current RSRs offer two useful sets of default numerical cleanup standards for residential and industrial/commercial settings. The assumptions do not apply to all sites, and flexibility to alter the exposure assumptions is not available as a self-implementing option. Industrial/commercial standards can only be used if an institutional control (Environmental Land Use Restriction- or ELUR) is in place. The ELUR in effect alerts property owners through a deed notice, granting the DEEP certain rights and requiring subordination with interest holders, such as lenders and utility companies. The RSRs also allow for exemptions from the standards if soil is covered to prevent exposure (with specified soil thickness, engineered barriers or buildings), again with an ELUR in place. In our experience, the default assumptions may not apply to the Site, the soil cover requirements may not be physically attainable at the Site, and ELUR procedures, particularly subordination agreements, may not be attainable at reasonable cost, if at all.

II. Discussion and Recommendations

This DEC sub-group offers five main recommendations:

Recommendation 1 – Review Assumptions for the default Industrial/commercial DEC

The default exposure assumptions for the industrial/commercial DEC are very conservative and should be re-evaluated. The assumed worker exposure to soil is 250 days per year for 25 years. In New England the ground will be frozen and often under snow cover for about 3-4 months per year and such intensity and duration of contact to a Site worker is highly unlikely. Therefore we recommend the DEEP re-visit their exposure frequency and duration assumptions to consider a more likely and reasonable 150 days per year and to revise the resulting criteria. This is consistent with other state programs, (e.g. Mass MCP). Accordingly, it may be prudent to evaluate industrial and commercial exposures separately.

Recommendation 2 – Moderate Exposure Approach

We recommend that DEEP (in conjunction with the Connecticut Department of Public Health) develop a third set of default numerical standards, “Moderate Exposure” standards, to cover the potential for moderate (and not worst case i.e., residential) scenario exposure. Recreational use is an example of a land use and set of exposure assumptions that fit between the conservative residential and less conservative industrial/commercial scenarios.

Variables such as exposure pathways, frequency and duration of exposure for various activities (sports playing, trail walking, etc.) would be used to calculate the new criteria. Such variables are delineated in EPA’s Exposure Factor Handbook (2011). Initially published in 1989, the Handbook provides statistical data on the various human factors used in assessing exposure. The most recent edition published EPA’s recommended values based on its interpretations of the most up-to-date data. An example from the State of New Jersey, outlining active and passive recreational soil exposure assumptions where consideration is specifically given to the inhalation pathway, is attached in Appendix A.

Recommendation 3 – Self-Implementing Risk Characterization

The current residential DEC was based on very conservative assumptions (e.g. exposure durations of 365 days per year for 30 years). The industrial/commercial DEC was based on less conservative exposure durations (250 days per year for 25 years). Numerous Sites have risk profiles that do not match these assumptions, and often have different zones with different land uses, making the current default standards overly stringent and too conservative in many instances.

We recommend a self-implementing risk characterization option be available that takes into account site-specific variables such as exposure pathway, dosage of the contaminant, the frequency and duration of exposure and body weight of the exposed population.

As an example, the MCP “Method III” risk characterization procedure summarized in Appendix A, has had success in Massachusetts. The fundamental concept of “no significant risk” allows sites with low potential for exposure, or exposure pathways interrupted by soil cover, engineered barriers or buildings to exit the regulatory system despite having subsurface soil contaminants that exceed the numerical standards. The “No Significant Risk” concept has gained public acceptance, has allowed non-productive sites to be redeveloped safely and offers an early “off-ramp” from the regulations.

The following risk assessment topics need additional evaluation and refinement:

- A guidance document will be needed that outlines the recommended risk assessment procedures;
- DEEP Staff, land professionals (consultants, developers, attorneys, lenders, utility companies) and citizens need to become comfortable with the concept of “no significant risk” as an outcome. This concept signifies that there is an absence of likely harm to human or ecological receptors, based on toxicological assessment, and in some cases, incomplete exposure pathways, even if cleanup standards are exceeded; and,
- DEEP Staff will need training to be comfortable reviewing site-specific risk assessments.

Recommendation 4 – Deed Notice

Control of exposure is an important tool for managing environmental contamination and the associated risk. Under current Connecticut Law, an Environmental Land Use Restriction (ELUR) must accompany exposure control where access is limited. The purpose of the ELUR is to minimize the risk of human exposure to pollutants and environmental hazards by preventing specific uses or activities at a property or portion of a property.

The Connecticut ELUR (*See Conn. Gen. Stat. § 22a-133q-1*), comprises a formal State-granted easement, subordination of other interests in the property, and final approval by the State. The express language of the ELUR requires that the restrictions run with the land and that the restrictions are binding and enforceable against the property owner and any of its successors in interest.

The working group understands that this robust and highly protective approach is appropriate in certain higher-risk situations, such as preventing human exposure to soil that exceeds the direct exposure criteria, preventing infiltration of water through soils which exceed the pollutant mobility criteria, preventing the disturbance of engineered controls, and/or preventing the construction of a building over groundwater that exceeds the volatilization criteria (*See Conn. Gen. Stats. §§ 22a-133k-2(b)(3), 22a-133k-2(c)(4)(B), 22a-133k-2(f)(2)(B)*),

22a-133k-3(c)(5)(A)). However, an ELUR in some cases may not be necessary to gain adequate protection.

We recommend that in certain circumstances a deed notice be allowed instead of an ELUR. Deed notices may need to be accompanied by one or more of the following: Health & Safety plan; soil management plan; inspections; erosion control; no-build restriction; limits on any other activity that exposes underlying soil, all depending on future use.

The working group recommends that a self-implementing streamlined process of deed notice is considered for lower-risk situations and is consistent with current and foreseeable use restrictions. For example Mass MCP uses Activity and Use Limitations (AUL). These restrictions would limit the property to industrial or commercial use on property either currently characterized under that use and/or in an area zoned for that use. (See Conn. Gen. Stats. §§ 22a-133k-2(b)(2)(A), 22a-133k-2(b)(2)(B), 22a-133k-3(c)(2), 22a-133k-3(c)(3)(A), 22a-133k-3(e)(2)(C)). This approach would also apply to the Moderate Exposure Standard. This type of deed notice ensures that the activity and use limitation information remains on property title. A filing with the municipal building or planning department to ensure that there is no change in zoning or building without consideration of the restrictions may also be an appropriate measure.

Recommendation 5 – Flexibility for Cover Thickness and Materials

The workgroup recommends a change to the way inaccessible soils are currently defined in the RSRs where DEC is exceeded. In addition, the use of an ELUR is overly protective and an alternative deed restriction is proposed.

Currently the RSRs define inaccessible soil as:

- where there is no physical barrier to the soil (next 2 bullets), there needs to be 4 ft of soil that does not exceed DEC over the soil that exceeds DEC (see Figure 1);
- where there are 3 inches of pavement at the surface, there needs to be 2 feet of soil (that does not exceed DEC) over the top of contaminated soil that exceeds DEC (See Figure 1);
- where there is a building or other structure approved by the commissioner that overlies the soil that exceeds DEC, no clean soil is needed (see Figure 1); and,
- the aforementioned all require an ELUR to restrict the disturbance of any of the above barriers in the future.

The workgroup recommends that the definition of “inaccessible soil” be changed as follows (presented in the same order as above) and illustrated in Figure 2:

- For soils that exceed DEC, allow less than 4 feet of clean fill over the contaminated soil.

- In cases where, for example PAHs, lead or arsenic, exceed DEC in soil due to asphalt fragments or the presence of urban fill (urban fill needs to be clearly defined: definition may use a multiple of the default RSR criteria [e.g. haz. not. criteria] and/or could be tied to a release or non-release by the responsible party), and the solution is to install or use an existing barrier of asphalt to prevent direct contact, then that asphalt barrier can be installed or allowed to remain directly over the DEC soil without requiring the use of 2 feet of clean fill over the PAH soil.
- For the building or structure exception, allow the use of “other structures” as a barrier to contaminated soil to be self-implementing, i.e. would not require the approval from the commissioner. Design elements of “other structures” should be defined, similar to an engineered cap. Instead of listing acceptable structures in the regs, there could be a description of what an acceptable structure must do, e.g. acceptable structures must prevent direct exposure without maintenance for a period of xx years; must withstand any use from the intended future users of the property and guidance on financial assurance for the structure. . Examples of other structures are: rip rap, wood chips, stone dust, asphalt, (choice depends on future use).

POLLUTANT MOBILITY CRITERIA SUBGROUP SUMMARY

I. Background and Recommendations

The PMC Subgroup reviewed the current basis on which the PMC are applied and the available self-implementing options. The workgroup identified two basic areas for re-evaluation during the transformation process: review of the derivation process for the PMC and expansion of the self-implementing options. Specifically, the workgroup recommends the following items:

1. Review of the methods and science behind development of the PMC. Is the method used by Connecticut scientifically based and defensible?
2. Review and consider the Anti-degradation policy to expand groundwater quality classifications or PMC categories, such as
 - a. Create new categories of the PMC tied to GW Classification or land use, such as GA-current and potential drinking water, GA-non-potential drinking water, and Non-drinking water (GB) uses.
 - b. Expand groundwater classifications to include a GC-type class, where a contaminant source has been removed/controlled, and a defined plume is undergoing a DEEP-approved monitored natural attenuation (MNA) program.

3. Modify “environmentally isolated” soil provision to include contaminated soil beneath a concrete slab (building demolished) and asphalt pavement. Further streamline the process by replacing the ELUR with a deed notice as appropriate.
4. Modify existing current self-implementing options
 - a. Make site specific circumstance for GA and GB areas more consistent
 - b. Implement changes to 95% UCL procedures as recommended by 95% UCL Workgroup (Attached)
5. Incorporate the self-implementing options provided in the proposed 2008 and 2012 RSR changes
6. Add a self-implementing option that allows the use of additional EPA-approved leaching tests to demonstrate compliance based on site-specific factors.

II. Discussion

PMC Derivation

The PMCs were derived in the 1990s using water potability standards, which were then conservatively applied to soil based on a comparison of various leaching tests. The Synthetic Precipitation Leaching Procedure (SPLP) extraction and analysis process was determined to provide a conservative estimate of leachability. Organic compounds were assumed to leach from soil at 100 percent of measured mass and therefore a 20 to 50-fold dilution factor was applied based on the dilution inherent in the SPLP extraction method. Because metals are naturally occurring it was determined that the hypothetical 20-fold leaching equivalent was not appropriate and a specific SPLP leaching values were used instead. The PMCs derived for GA areas were then multiplied by a factor of 10 to derive GB PMCs for most cases. The resulting PMCs were conservative and may not actually yield a groundwater plume in a natural setting at Connecticut sites. Site specific leaching studies may be completed under the current RSR but are costly and not self implementing. The PMC derivations and Criteria values need to be revisited and updated based on more recent available science. Other states such as New York and New Jersey use a soil partitioning based approach to develop a similar soil leaching based standard.

PMC Exemption Urban Fill

Historically, urban fill has been seen as the elephant in the room in respect to cleanups. Because of its wide variability and unknown composition, ranging from natural soils to a hazardous mixture of chemicals and metals, it has been left as the orphan in regards to regulation. The workgroup supports a review of this complex issue and, if possible, the

incorporation of a definition into the regulations, concurrent with an approach that deals with the relative risk. We understand that there is another work group that is addressing this issue and we support continuing their efforts. See Appendix B for a draft definition of Urban Fill, proposed by this work group.

New PMC Groundwater Categories

The PMC are currently broken down by the two primary groundwater classification categories: GA and GB. However, many GA areas are densely populated areas that are served by public water. In addition, although the GA groundwater classification is used to indicate that groundwater may be used as potable water without treatment, many areas classified as GA do not have a sufficient level of permeability to yield sufficient water for a public or private water supply. Therefore, additional categories for PMC applicability should be added, including:

- GA-current and potential drinking water;
- GA-non-potential drinking water; and,
- Non-drinking water (GB) uses.

In addition, for sites where a contaminant source has been removed/controlled and a defined plume is undergoing an MNA program, a “GA-impaired” class is recommended to designate a zone where a plume is present, groundwater is not currently or likely to be used for drinking water and where groundwater quality is improving with time. All of the above would need to be implemented with the anti-degradation policy in mind such that additional contamination is prevented and existing impacts to the State’s groundwater are improved with time.

Environmentally Isolated Soil and ELURs

The RSRs currently only allow soil in excess of the PMC to remain in place if it is “Environmentally Isolated” under a building or other permanent structure approved by the Commissioner or under a Commissioner-approved Engineered Control of Polluted Soil that incorporates a low permeability layer. These each need to be “locked in” with an ELUR. Many sites where soil may be isolated under a building are in underutilized abandoned industrial properties (brownfield properties). Where a developer may be interested in such properties, if an ELUR is in place requiring that building to remain, the cost of renovating or maintaining the buildings may be prohibitive. However, the underlying soil may be sufficiently protected from infiltration by the building slabs if left in place following demolition, especially given the conservative assumptions built into the current RSRs. Therefore consideration should be given to including additional categories of Environmentally Isolated soils such as building slabs left in place or low permeability bituminous pavement in a non-drinking water area. Also, a more easily implementable property restriction, such as a deed notice, that doesn’t require subordination agreements or the signature of the Commissioner, should be available. These would be particularly useful in a scenario such as that described above where a property is in a

non drinking water-classified area and no potable water sources or nearby environmental receptors such as streams or wetlands are present.

Site Specific Circumstances

The RSRs currently allow modification of the PMC under certain site specific circumstances. These circumstances vary depending upon the groundwater classification and the site's setting. In a GA area, for example, the GA PMC may be multiplied by 10 if a public water supply distribution system is available within 200 feet, the ground water within the areal extent of such ground-water plume is not used for drinking water, no public or private water supply wells exist within 500 feet of the subject release area, and the ground water affected by the subject release area is not a potential public water supply resource. However, in a GB area, TCLP or SPLP analyses can be compared to the groundwater protection criterion times either a factor of ten, or a factor derived from the sum of the upgradient and downgradient areas to the footprint of the release area (up to 500). There are other circumstances as well. These circumstances provide some self-implementing flexibility to the current RSR; however, they should be reviewed and revised to give greater consistency in approach and more options for self-implementing alternatives.

2012 and 2008 RSR Changes

The currently proposed changes to the RSRs include a limited self-implementing performance evaluation to exclude the applicability of the PMC to soil, which was also included in the draft 2008 RSR revisions that were never formally proposed. This performance evaluation is based on whether or not a groundwater plume exists on a site where at least 80% of a release area has been subject to infiltration for a minimum of five years. This proposal is a useful and pragmatic and promotes the PMC's purpose to remediate a release to soil above the water table if it is the source of a plume in the underlying groundwater. If there is no plume, remediation to meet the PMC is a needless and costly exercise. The RSR proposal does allow this test of groundwater to be applied to other circumstances but only subject to Commissioner approval. Consideration should be given to expanding the applicability of this concept to a broader spectrum of sites on a self-implementing basis.

Pesticides/Herbicides used in Agricultural Applications

Large tracts of farmland, orchards, tobacco beds and other agricultural properties became contaminated through the application, in conformance with labeling directions, of pesticides and herbicides. Some of these pesticides and herbicides are now banned, with constituents including arsenic, lead, dieldrin, Endrin, lindane, 1,2-dichloropropane, and other persistent pollutants. The application of these materials has resulted in the presence in soils of residual concentrations of constituents in exceedence of the PMC criteria of the RSRs.

In instances where these properties have come under environmental scrutiny due to redevelopment, large quantities of soil have been disposed of as wastes, due to exceedences of the PMC criteria, or have resulted in prolonged oversight within the regulatory programs.

The workgroup recommends a self-implementing exemption from PMC for these properties, in instances where the groundwater meets the applicable RSRs criteria and/or demonstrates protection of all potentially affected drinking water receptors and sensitive environmental receptors. The exemption would specifically exclude spills, mixing, and storage areas, where concentrations of the contaminants exceed those which would typically be present as a result of appropriate application practices or guidelines.

PMC 95% UCL

DEEP created a separate workgroup (95% UCL Workgroup) to develop guidance regarding the use of a 95% UCL calculation on soil and groundwater data sets to demonstrate compliance with certain RSR criteria. The 95% UCL Workgroup submitted written comment on October 25, 2012 in response to the public hearing for the currently proposed changes to the RSR (Attached). The 95% UCL Workgroup recommended a number of changes that directly affect the PMC. The PMC Subgroup concurs with the recommendations of the 95% UCL workgroup, which include but are not limited to:

1. Removal of the two times limit on the data.
2. Change to language regarding the minimum number of soil samples from “...not less than 20 samples of soil collected above the water table” to “...not less than 10 samples of soil collected above the water table” within the release area.
3. Removal of the excavation restriction for use of the 95% UCL for the PMC.

The complete recommendations of the 95% UCL are attached for reference.

Additional EPA Approved Leaching Tests

The workgroup supports the self-implementing use of alternate PMC models that are EPA-approved and applicable for use based upon site specific criteria.

The RSRs currently only specify the SPLP and TCLP methods as self-implementing methods for evaluating the site-specific capacity of soil to leach contaminants. In 1997, the DEEP partially funded an evaluation of TCLP and SPLP batch leaching procedures as alternative methods for evaluating pollutant mobility in glaciated soils over more time consuming, complex and expensive flow through column tests¹. The study indicated that metals extraction through the SPLP method provided more realistic estimates of mobility metals in glaciated soils than TCLP.

¹ Lackovic, J.A. , N.P. Nikolaidis, C. Pradeep, R. J. Carley and E. Patton. 1997. *Evaluation of Batch Leaching Procedures for Estimating Metal Mobility in Glaciated Soils*. Ground Water Monitoring and Remediation 231-240.

TCLP was designed to evaluate mobility in solid waste landfills where anaerobic degradation of wastes produce carboxylic acids, such as acetic acid, these conditions are not normally found in the vadose zone of industrial sites. However, the authors concluded that the SPLP tests overestimated the mobility of most metals in the soils studied and more closely represents a conservative scenario of sandy glaciated soils with low organic content. This has been observed empirically where metals in glaciated soils at concentrations within the range of the site background have leached by SPLP extraction at concentrations in excess of PMC numeric criteria.

EPA is currently evaluating additional methods for approval that may provide more effective means for evaluating leachability. For example, EPA is currently evaluating Method 1316 which provides an evaluation of the liquid/solid partitioning of various inorganic and non-volatile organic constituents at the natural pH of the solid material at conditions that approach the true liquid-solid equilibrium. Extracted constituent concentrations in materials with low liquid-to-solid ratios can provide better estimates of pore solution concentrations, either in a granular bed or in the pore space of low-permeability material like a compacted granular fill. In addition, analysis of extracts of dissolved organic carbon and of the remaining solid phase allow for estimation of the impact of organic carbon release and the potential influence of dissolved organic carbon on the liquid/solid partitioning of inorganic constituents. The availability of additional self-implementing test methods under a revised regulation will enhance the ability of the regulated community to advance sites towards risk reduction and closure.

SEDIMENT SUBGROUP SUMMARY

I. Background

For sites that meet the applicability requirements, the RSRs clearly require remediation of soil and groundwater contamination and provides default numeric cleanup criteria to address these affected environmental media. When the RSRs were promulgated, the requirement to perform sediment remediation was not included as it was envisioned that the remediation of sediment would be managed on a case-by-case basis and would be overseen by DEEP. The uncertainty with this approach has resulted in a lack of clarity for the regulated community, environmental professional and DEEP and a patch work of divergent policies.

At the present, this case by case approach has resulted in many sites becoming “stuck” in the regulatory/remedial process and requiring significant DEEP resources. Given the regulated community’s and DEEP’s interests and commitment to the ongoing process of remedial program transformation, there is an opportunity to enhance the process for addressing sediment investigation and remediation. This can result in reducing uncertainty and backlog and optimizing the important use of DEEP resources.

II. Recommendations

The workgroup believes it is critically important for the DEEP to modify the RSRs to clearly define when sediment remediation is required and to develop a program that is a combination of regulation and guidance to address characterization and remediation of sediment. The recommended approach:

1. Has been shown to be successful in numerous other states;
2. Is supported by the Federal Regulatory community;
3. Would allow for the LEPs to guide the investigation and remediation of lower risk sites;
and
4. Would result in the allocation of DEEP resources, appropriately, to only those higher risk sites.

Based on the potential complexity of sediment toxicology and respectful of the importance of site setting in sediment assessment, the workgroup recommends that the RSRs clearly indicate the need to address sediment as part of an appropriate standard of care for site investigation and remediation in Connecticut. Likewise, the workgroup believes that, the most efficient manner for establishing the ways and means related to this standard of care, is through well thought out guidance that can be utilized by the regulated community and environmental professionals.

This being said, the Workgroup envisions a tiered program that incorporates the current LEP program that would be directed by a guidance document, with DEEP approved regulatory program exit strategies and checklists. The tiered program would define exit points from the regulations and would rely upon an environmental professional to perform and document the necessary evaluations.

In keeping with the current system of checks and balances and picking up on what some other States have incorporated, documentation of the decision making process on forms prescribed by the Commissioner should be incorporated and selected decision of early exit could be the subject of Department audit, as needed to insure the integrity of the process. We envision that the submittals at each audit selected exit point would be reviewed by DEEP, as under specific time frames (i.e. 30 or 60 days from receipt) after which the submittal would be considered approved by DEEP and allow the exit process would be complete. Consistent with other programs, DEEP would have the option to request more information during the time frame which would extend the approval deadline by a specified amount of time. This approach of exit options and select DEEP audit would provide certainty of process and standard of care.

III. Discussion

The workgroup envisions four tiers, or levels of assessment, for sediments at a site. Tiers would be followed sequentially and should a submittal under higher level tiers be selected, a demonstration that the requirements of the lower tiers have been met should be part of the higher tier submittal. The following tiers (consistent with Federal guidance) are envisioned:

Tier 1 - Scoping Level Assessment

This tier would be utilized when it can be demonstrated that there is no complete exposure pathway. The intent of a Tier 1 demonstration is to substantiate a “no unacceptable risk to human health or the environment” outcome due to the release of interest without the need for testing of sediment or only limited testing in certain settings. The Workgroup would recommend a simple check box form that would demonstrate this level of sediment assessment and provide the credentials of the environmental professional making the assessment. As an example, the Pennsylvania regulations (see appendix) for sediment provide an example of an early exit format for sites with no or a low potential for sediment contamination issues.

Tier 2 – Screening Level Assessment

Assessment at this tier would be performed when the environmental professional has made a determination that sediments are present and a complete exposure pathway may exist that would result in sediments being impacted by a site release. This level of assessment would determine, by sediment testing, whether selected screening criteria can be met; indicating a site does not pose an unacceptable risk to human health or the environment due to sediment contamination. The MacDonald screening criteria (consensus based numeric criteria for sediment quality) cited by the Wisconsin regulatory program (see appendix) were used as an example of this approach. Since the publication of the MacDonald screening criteria, additional information may be available and the Workgroup recommends that the information is reviewed for further elaboration and availability of options applicable to this level.

DEEP must address “Special case contaminants” such as PAHs as well as background and unrelated sources and setting (fill, urban fill, parking lot, etc.) to facilitate efficient Tier 2 level assessments.

Tier 3 – LEP Risk Assessment

The next tier in the sediment assessment process would be for sites where concentrations of constituents exceed the screening criteria established under Tier 2. Regulatory requirements at this level would allow LEPs (and possibly other credentialed environmental professionals such as degreed ecologists and certain biologists) to conduct site specific risk assessments (see Pennsylvania regulatory program information in Appendix).

If the site specific risk assessment shows no unacceptable risk to human health and the environment (based on analytical testing and professional assessment of conditions and pathways), the site may exit with a final report that meets the requirements of a regulatory assessment checklist designed specifically for this tier.

If the risk assessment shows no unacceptable risk to human health and the environment if specific institutional controls are used (the institutional controls selected by the environmental professional would have to meet proscribed, DEEP approved criteria and be documented), then the LEP could supervise the institutional controls and the site could exit at this point (may require a monitoring program and/or some form of recorded use limitation).

If the site requires remediation, the LEP should be allowed to implement the remediation and then exit the program, once the remediation is complete, documented and approved (with a default approval time line of 60 to 90 days) by DEEP.

Tier 4 – DEEP Risk Assessment

The last tier of sediment assessment would be for those sites where certain factors may not allow for an LEP to achieve one of the earlier exit points described for Tier 1 to Tier 3. Examples would include: technical impracticability; a proposed remedy that would result in the destruction of the wetland containing the impacted sediment; and/or sites where broader public policy issues are involved such as settings where an entire watershed could be affected by the contamination or the proposed remedy. The DEEP would direct the proposed remediation of sediments under more complicated Tier 4 situations and DEEP would work with the LEP to develop an appropriate remedial and/or monitoring program.

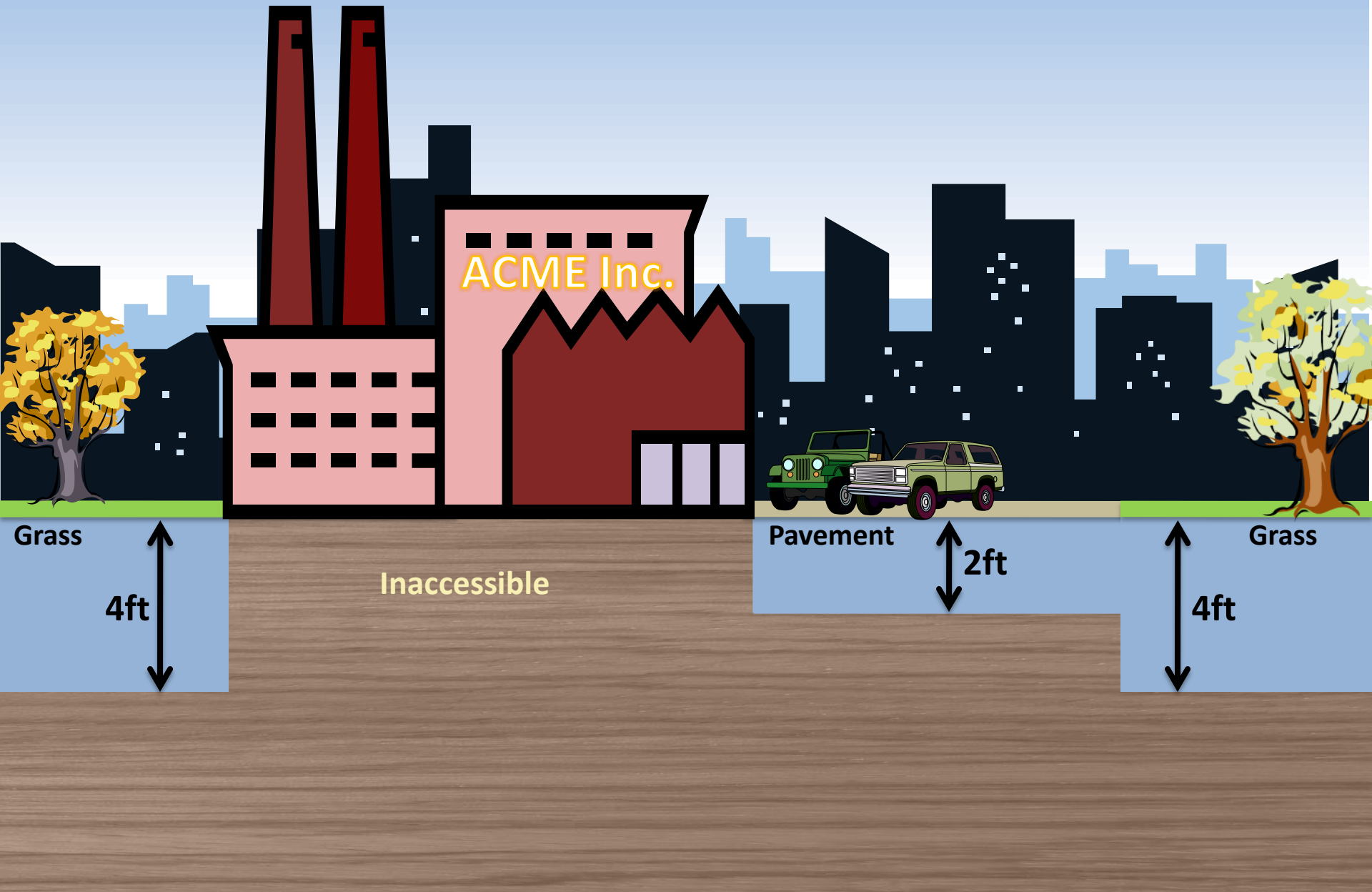
Additional Recommendations

The guidance documents for sediment characterization and remediation should include description in detail of the qualifications environmental professionals should have to complete sediment assessment. The guidance should include a description of appropriate level of expertise for environmental professionals that perform initial characterization and evaluation of exposure pathways, design of sediment sampling programs, design of controls and design of remedial measures. The guidance should make allowances for lower level qualifications for sediment sample collection and data gathering as long as it is supervised by an environmental professional with higher level qualifications. Certification by an environmental professional for certain steps in the process may be warranted.

Information from PA, NJ, WI (see appendix) and current CT draft regulations should be referred to when preparing DEEP guidance that would govern the sediment regulatory process.

Figures

Figure 1 - Direct Exposure Criteria: Current RSRs



Not to Scale

Figure 2 - Direct Exposure Criteria: Proposed Changes



Grass
↑
Less
↓
than 4ft

Pavement

Grass
↑
Less
↓
than 4ft

Inaccessible Soil

Not to Scale

Appendix A
Direct Exposure Criteria Supporting Documentation

APPENDIX A

New Jersey's Example Risk Assessment Approach

Submitted by Andrea Boissevain (Stratford Health Department)

New Jersey employs several different sets of alternative standards under NJAC 7:26D that include two categories of recreational land use:

- Active (e.g. sports playing fields & playgrounds); and
- Passive (e.g. bike riding and trail walking)

All the variables that are used in assessing a particular [human] receptor's exposure would come into play along—again using science-based exposure variables, allowing even more specificity with regard to site use, receptors, duration, frequency, etc.

New Jersey also regulates inhalation of particles if that exposure pathway is a concern. We recommend that DEEP consider this pathway and allow the flexibility for site-specific risk assessment, because there are sites in Connecticut where asbestos-impacted soils at or near the ground surface are a concern for the fugitive dust exposure pathway.

Summary of New Jersey Soil Remediation Standards & Process

1. **Minimum soil remediation standards** Under NJAC 7:26D, NJDEP calculates for carcinogens and non-carcinogens for residents and non-residential outdoor worker.
 - a. Classic Superfund risk assessment methodology incorporates an algorithm that derives a combined ingestion & dermal criterion as well as an inhalation criteria. The resultant Direct Contact Soil Remediation Standard appears to be the more stringent of the two.
2. **Alternative Remediation Standards (ARS)** Options are also available. Some require NJDEP approval (guidance is available to determine what requires approval).
 - a. Soil-Direct Contact ARS Option 1 is lead-contaminated sites.
 - i. Residential: IEUBK model may be used with site-specific soil and dust concentrations-no straying from default parameters
 - ii. Non-residential lead sites: worker sites; recreational land use sites may use input parameters specified EPA Technical Review Workgroup documents.
 - iii. No remediation of a discharge if below "natural" background level. Appears to require investigation per NJAC 7:26E3.8.

- b. Soil ARS Option 2 – Recreational Land use
 - i. Active (e.g. sports playing fields & playgrounds)
 - ii. Passive (e.g. walking or biking trails)
 - iii. BOTH require proper institutional controls

- c. Inhalation Pathway ARS Option 1 Volatile Phase Contaminants
 - i. Depth Range of Contamination
 - ii. Soil organic carbon content (sampling required)
 - iii. Site size

- d. Particulate Phase ARS Option 2
 - i. Residential – vegetative cover calculated & calculate particulate emission factor.
 - ii. Land use restriction required.

- e. Inhalation Pathway ARS Option 3– Recreational Land use
 - i. Active (e.g. sports playing fields & playgrounds)
 - ii. Passive (e.g. walking or biking trails)
 - iii. BOTH require proper institutional control

APPENDIX A

Massachusetts Example Risk Assessment Approach

Submitted by Valerie Tillinghast, LSP and Paul Tanner, LEP, O'Reilly Talbot & Okun Associates, Inc.

Massachusetts employs three sets of risk-based soil standards which are generated based on different land use assumptions. LSPs may calculate additional standards for constituents not included on the standard list. Additionally, site-specific risk characterization may be completed for any site at the LSP's discretion. These three methods of risk characterization are identified as Method 1 through Method 3 in the MCP.

Method 1: Default Standards

Default (Method 1) soil standards are available for approximately 120 contaminants. The default exposure assumptions used to develop the Method 1 standards are:

- S-1 standards assume residential children and adults may be exposed to impacted soil 153 days per year (every day from May to September);
- S-2 standards assume passive recreational users or site workers may contact impacted soil five days a week April through October;
- S-3 standards assume construction workers may contact impacted soil five days per work during a three month period of earthwork.

Site concentrations below applicable Method 1 standards are considered to pose No Significant Risk.

Method 2: Standards for Additional Compounds

Method 1 standards are available only for about 120 contaminants, but comparable standards may be calculated by the LSP for additional contaminants detected at their sites. Calculation of these Method 2 standards uses the default exposure assumptions identified in Method 1, combined with chemical-specific toxicity factors, generally obtained from USEPA sources.

As with Method 1 standards, site concentrations below Method 2 standards indicates a condition of No Significant Risk for exposure to the impacted soil.

Method 3: Site Specific Risk Characterization

Method 3 risk characterization may be completed for any site at the LSPs discretion. When vapor intrusion is a concern, the use of Method 3 is obligatory, because Method 1 standards do not include a vapor intrusion component.

Method 3 involves calculating risks for each contaminant via each potential exposure pathway at the site (soil contact, dust inhalation, vapor inhalation, groundwater contact, groundwater ingestion, etc.). Pathways that are incomplete (such as due to soil cover and an ELUR) are not quantitatively assessed.

Exposure assumptions, such as body weight and exposure duration, are based on the receptors who may be present. For consistency of approach, assumptions are often adopted from MassDEP's Method 1 equations.

Risks are calculated separately for carcinogenic and noncarcinogenic effects. The risk estimates are summed and compared to MassDEP risk management criteria (1 for noncancer, 1×10^{-5} , or one-in-one-hundred thousand for cancer). If risk totals are below these limits, a condition of No Significant Risk is concluded.

Appendix B
Pollutant Mobility Criteria Subgroup
Supporting Documentation

WHAT IS CONSIDERED URBAN SOIL?

Urban Soil includes material deposited on a parcel that contains a mixture of one or more of the following: soil, coal ash, coal fragments, wood ash, asphalt paving fragments, clinkers, brick, concrete, glass, ceramics, metal fragments and incidental amounts of other construction and land-clearing debris



WHAT IS CONSIDERED URBAN SOIL?

Provided that:

- Contaminants present above RSR criteria in the material are not the result of a release;
- Deposition not prohibited at time of placement;
- Urban soil would not include materials such as foundry slag, casting sand or coal tar; and
- Urban Soil would be exempt from the pollutant mobility criteria under RSRs 22a-133k-2(f)



OVERVIEW

“Clean Fill” not included

Not limited to “fill” brought in from off-site

Not limited to inner city sites

Contaminants in Urban Soils considered to be a “release”

RSR exceedances require remediation

Standardized EC designs provided



October 25, 2012

Mr. Rob Bell
Remediation Division, 2nd Floor
Bureau of Water Protection and Land Reuse
Department of Energy and Environmental Protection
79 Elm Street
Hartford, CT 06106

Dear Mr. Bell:

This letter provides comments on the August 21, 2012 proposed amendments to the Remediation Standard Regulations (RSRs), Sections 22a-133k-1 through 22a-133k-3, inclusive, and 22a-133q-1, of the Regulations of Connecticut State Agencies (RCSA), submitted for public comment by the Connecticut Department of Energy and Environmental Protection (DEEP). The comments submitted herein were prepared by the 95% Upper Confidence Limit (UCL) Workgroup (herein referred to as the 95% UCL Workgroup). The 95% UCL Workgroup was established in May 2012 by the DEEP Remediation Roundtable as a part of the DEEP Remediation Program Transformation Process. The workgroup consists of two DEEP staff members, one Department of Public Health staff member, and seven members of the regulated community (composed predominately of licensed environmental professionals). The objective of the 95% UCL Workgroup is to develop guidance for the regulated community regarding the use of the 95% UCL calculation on soil and groundwater sample datasets to demonstrate compliance with certain RSR criteria. As of the date of this letter, these criteria include, the Direct Exposure Criteria (DEC) per Section 22a-133k-2(e)(1), the Pollutant Mobility Criteria (PMC) per Section 22a-133k-2(e)(2)(A), the Groundwater Protection Criteria (GWPC) per Section 22a-133k-3(f)(1), and the Volatilization Criteria (VC) per Section 22a-133k-3(f)(3).

During the course of the last several months, the 95% UCL Workgroup has conducted research associated with the formulation and application of the 95% UCL calculation, including a review of federal and state literature and guidance. Based on our research and findings, the 95% UCL Workgroup has prepared the following comments on the proposed amendments to the RSRs as they relate to the 95% UCL concept:

1. The 95% UCL Workgroup recommends the removal of the two times multiplier limit on the data and associated proposed change “unless an alternative multiplier has been approved by the Commissioner in writing”. The 95% UCL calculation is self-limiting as to how high any single concentration in the dataset can be while still calculating a 95% UCL that is below criteria. For example, inclusion of concentrations more than two times the criteria may result in a 95% UCL below criteria; however, the inclusion of concentrations more than five or ten times the criteria may not. Our recommendation is supported by our research where EPA and several other states do not include a restriction on the maximum concentration allowed in the dataset.

If the two times multiplier restriction is not removed, the 95% UCL Workgroup concurs with the inclusion of “unless an alternative multiplier has been approved by the Commissioner in writing” as it currently stands in the proposed amendments and added to Section 22a-133k3(f)(1)(B) for the GWPC (if that language remains in place).

2. The 95% UCL Workgroup recommends a change of the language in proposed Section 22a-133k-2(e)(2)(A)(i) for the PMC, which states the use of “...not less than 20 samples of soil located above the water table” to “not less than 10 samples located above the water table or another dataset size approved in writing by the Commissioner”. This is supported by the research conducted by the 95% UCL Workgroup, which indicates dataset sizes between 10 and 20 samples may also be adequate for calculating the 95% UCL based on the data distribution. Alternatively, the language “unless an alternative number of samples has been approved by the Commissioner in writing” should be added to the proposed amendments. Sections 22a-133k-2(e)(2)(B) will also require revisions to be consistent with the changes in 22a-133k-2(e)(2)(A).
3. The 95% UCL Workgroup concurs with the removal of the excavation restriction for the use of the 95% UCL for PMC. However, it appears that not all of the language associated with the excavation restriction was removed, specifically in 22a-133k-2(e)(2)(B)(ii) and (C). These sections should be deleted to remove all references to the excavation restriction.
4. With regard to all instances in which the 95% UCL is referenced, the language should be changed to “the ninety-five percent, or higher, upper confidence level of the arithmetic mean ...” to allow for the use of the 97.5% or 99% UCLs if deemed more appropriate based on the data distribution.
5. The 95% UCL Workgroup recommends that the option of demonstrating compliance with the GWPC using the 95% UCL remain in the RSRs (original Section 22a-133k-3(f)(1)). There has been no evidence to support that averaging groundwater concentrations at a single well location is not protective of human health and the environment, and as such there would be no need to remove this provision. The 95% UCL Workgroup also recommends that the dataset restrictions in the original language be modified so that samples may be collected at a frequency greater than monthly and over a longer time period. For example, a minimum of ten samples representative of seasonal variation collected over a three-year period.
6. It is recommended that the language in proposed Section 22a-133k-3(g)(2)(C)(i) be clarified with respect to “all sample results of laboratory analyses”. The dataset to be used for the 95% UCL calculation should contain a minimum of ten samples, be representative of the current conditions of the groundwater plume, and account for seasonal variations.

Mr. Bell
October 25, 2012
Page 3 of 3

We appreciate the opportunity to provide these comments as part of the public participation process. If you have any questions or require clarification, we request that you contact Kathy Lehnus, LEP at 860-948-1628 ex 7118 or kathy.lehnus@stantec.com.

Sincerely,



Kathy Lehnus, LEP on behalf of
The 95% UCL Workgroup¹:

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¹Note: DEEP personnel participating on the 95% UCL Work Group have been omitted per state requirement

Appendix C

Sediment Subgroup Supporting Documentation

State of New Jersey

State of Pennsylvania

State of Wisconsin



**New Jersey Department of Environmental Protection
Site Remediation Program
GUIDANCE FOR SEDIMENT QUALITY EVALUATIONS**

Table of Contents

	<u>Page No.</u>
1.0 Introduction	2
2.0 Sampling Plan Design	3
2.1 Sediment Sampling Plan	3
2.2 Special Considerations for Sampling in Tidally-Influenced Areas	4
2.3 Chemical Characterization of Upgradient and/or Off-Site Reference Conditions	4
2.4 Surface Water Quality Investigations and Criteria	5
3.0 Sediment Screening Values for Use in the Baseline Ecological Evaluation	6
3.1 Inorganics, Semivolatile Organics, Pesticides/PCBs	6
3.2 Volatile Organics	6
3.3 Total Petroleum Hydrocarbons (TPHC)	6
3.4 Comparison of Site-Related Data to Sediment Screening Values	7
4.0 Biological Evaluations for Use in the Ecological Risk Assessment	8
4.1 Sediment Toxicity Testing	8
4.2 Benthic Macroinvertebrate Surveys	10
4.3 Tissue Residue Analysis	11

List of Tables

The Tables in the 1998 edition are outdated. For current information, see the Ecological Screening Criteria Table at <http://www.nj.gov/dep/srp/guidance/ecoscreening/>

List of Figures

Figure 1. Sketch map of river showing stratified regions and sampling points	3
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1.0 INTRODUCTION

The purpose of this document is to establish practical guidance for the evaluation of sediment quality to be used in the ecological risk assessment process associated with contaminated sites under the jurisdiction of the Site Remediation Program (SRP) in the New Jersey Department of Environmental Protection. Presented are procedures and references that form a framework for qualitative and quantitative determinations of actual or potential adverse ecological effects and provide the basis for remedial decision-making and evaluation of injury to natural resources in sediment media. The information presented in this document is based on State and Federal regulations and guidances, in particular *Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessments* (EPA 540-R-97-006) and *Risk Assessment Guidance for Superfund, Volume II, Environmental Evaluation Manual* (EPA/540/1-89/001). It is intended to be consistent with, and supplementary to, the *Technical Requirements for Site Remediation, N.J.A.C. 7:26E*. References are presented at the end of each major section for ease of use.

In accordance with *N.J.A.C. 7:26E-3.8 (b)*, the collection of sediment samples is required when it is evident that a discharge to a surface water body has occurred pursuant to *N.J.A.C. 7:26E-3.8 (a)*. Successful evaluation and risk management of contaminated sediments requires knowledge of the nature, concentration and areal extent of contamination, as well as site-specific variables that affect the expression of environmental impacts. There are three components of a complete assessment of sediment quality:

- (1) measurement of contaminant concentration, via standard or special analytical laboratory procedures;
- (2) measurement of toxicity and bioavailability, via tissue analysis, sediment toxicity testing, etc.; and
- (3) assessment of resident biota, via community bioassessment/survey procedures.

These three components, measured at potentially site-impacted and reference locations, provide complementary data, because no single component can be used to predict the measurement of the other components. For example, sediment chemistry provides information on the identification and extent of contamination but not on biological effects. Sediment toxicity testing provides direct evidence of sediment toxicity but cannot discriminate among contaminants nor predict actual in-situ responses. In-situ responses of resident biota, measured by in-fauna community surveys can provide direct evidence of contaminant-related effects, but only if confounding effects unrelated to contamination can be excluded, such as differences in habitat quality. Thus, a sediment evaluation program must be based on this "triad" approach to provide a weight of evidence for determining if adverse effects are occurring, and if so, whether they are due to the site in question.

For sediment quality evaluations at SRP sites, this "triad" investigation is accomplished pursuant to the tiered approach described in *N.J.A.C. 7:26E-3.11 and 4.7*. In the Baseline Ecological Evaluation (BEE), the site is examined for the co-occurrence of chemicals of potential ecological concern, environmentally sensitive areas, and complete chemical migration pathways, to assess the potential for ecological risk. If this initial evaluation indicates the potential for adverse ecological effects, a subsequent, more rigorous evaluation will be required for the full Ecological Risk Assessment (ERA) to further characterize risk.

REFERENCES

N.J.A.C. 7:26E. Technical Requirements for Site Remediation.

U.S. Environmental Protection Agency. March, 1989. *Risk assessment guidance for Superfund, volume II, environmental evaluation manual*. EPA/540/1-89/001. Office of Emergency and Remedial Response. Washington.

U.S. Environmental Protection Agency. June, 1997. *Ecological risk assessment guidance for Superfund, process for designing and conducting ecological risk assessments*. EPA 540-R-97-006. Office of Solid Waste and Emergency Response. Washington.

2.0 SAMPLING PLAN DESIGN

2.1 SEDIMENT SAMPLING PLAN

Generally, the goals of a sediment sampling program include preliminary and definitive determination of the nature and areal extent of contamination, and identification of areas of highest contamination. Data may also be gathered in support of ecological risk assessments, long-term monitoring, or for sediment transport and deposition modeling. The sediment sampling plan shall be a component of the Site Investigation or Remedial Investigation Work Plan, and shall be prepared pursuant to *N.J.A.C. 7:26E* and the *NJDEP Field Sampling Procedures Manual (FSPM, May 1992 or most recent version)*. Department approval may be required, pursuant to the oversight document (for privately funded projects) or contract in effect. Site-specific details regarding the study objectives, data quality objectives, sampling methodology, location, and depth of samples must be specified, as well as field and laboratory quality control/quality assurance procedures. Guidance and special considerations for designing a sediment sampling scheme are provided herein to supplement and highlight the regulatory requirements and FSPM guidance; the reader is referred to these documents for a comprehensive treatment of the subject.

1. Number of Samples

The reader is referred to USEPA's *Sediment Sampling Quality Assurance User's Guide* (USEPA, 1985) and the NJDEP FSPM for guidance on statistically determining the appropriate number of sediment samples.

2. Location

In aquatic systems, the areas of greatest contamination will generally occur in depositional areas, thus these must be specifically targeted by the sampling plan. Such depositional areas are generally characterized by slow moving water where fine sediments tend to accumulate (e.g., pool areas, river bends, etc.). Sediment samples collected for chemical analysis, toxicity testing and benthic community surveys must be spatially and temporally co-located.

a. Stream/River/Tidal Creeks Systems

An idealized approach to locating sediments samples is as follows: The stream location adjacent to the contaminated site most likely to receive contaminant input via the chemical migration pathway is considered the initial sample point. The study region is divided into linear segments and sample transects located systematically within each segment; the length of the segments and distance between transects increases with increasing distance downstream. This is depicted in Figure 1, a diagram of a sampling plan indicating 15 sediment samples per segment region. In this example, the first segment is from 0 to 1 km, the second from 1 to 3 km, and third from 3 to 7 km. The sampling transects are located at $\frac{1}{4}$, $\frac{1}{2}$, and $\frac{3}{4}$ the distance along each segment. Sample points are located along the transects at $\frac{1}{6}$, $\frac{1}{3}$, $\frac{1}{2}$, $\frac{2}{3}$, and $\frac{5}{6}$ the distance bank to bank (USEPA, 1985). In tidal creeks, the distance from bank to bank is measured from the high water mark. **Note that upgradient sediment samples must be collected (refer to Section 2.3), thus similar sampling transects should be located upstream of the initial sampling point.**

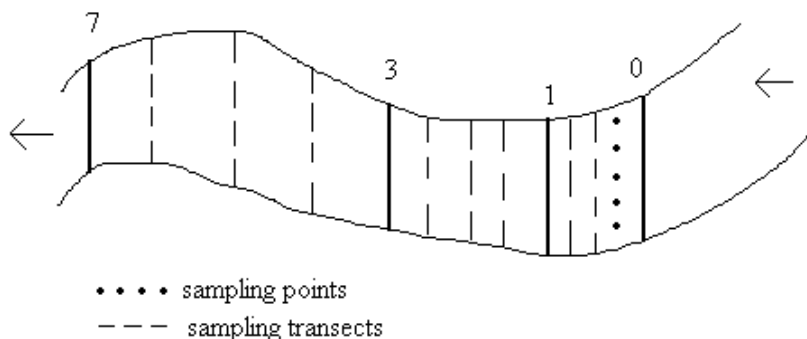


Figure 1. Sketch map of river showing stratified regions and sampling points.

The actual number and location of sample points will be decided on a case-by-case basis, based on the study objectives, water body dimensions, flow conditions, substrate conditions, availability of previous data, etc.

b. Lakes/Lagoons/Pond Areas

Sediment samples must be biased toward inflow/outflow areas and topographically low/deep areas where sediments may be expected to accumulate. If there is no basis for biasing, then random sampling of these areas is required, pursuant to *N.J.A.C. 7:26E-3.9(f)*.

3. Sample Depth

Pursuant to *N.J.A.C. 7:26E-4.1*, surface and subsurface sediment samples are required for contaminant delineation and to assess the potential for resuspension of contaminated sediments during flood/current-based scouring events, dredging operations, or other disturbances. Surface sediment samples must be taken at the 0-6" interval, generally considered the biotic zone in sediments. Subsurface core samples, 6-12" or deeper (actual depth based on site-specific conditions), are appropriate in areas of known discharge of contaminated groundwater to surface water (refer to 2.1.5. below) or where known historic discharges have become overlain with newer sediment.

4. Analytical Protocol and Additional Measurements

In addition to bulk chemistry analysis pursuant to *N.J.A.C. 7:26E-2*, a sediment quality evaluation may include additional physical measurements, including but not limited to river depth, flow rate, suspended solids, bed load, pH, and temperature. Total organic carbon (TOC) and particle grain size must be included as indicators of contaminant bioavailability and the depositional nature of the sediments. TOC is necessary for the determination of certain sample-specific sediment quality guidelines (refer to Section 3.0).

5. Volatile Organic Contamination

The most prevalent scenario requiring the collection of sediment samples when volatile organics are of potential concern is when contaminated groundwater is known/suspected to discharge to a surface water body. When this pathway is being investigated, the sediment samples shall be collected from the 6-12" interval. It should be noted that non-aqueous samples to be analyzed for volatile organics shall be sampled using a methanol extraction/preservation method acceptable to the NJDEP pursuant to *N.J.A.C. 7:26E-2.1 (a)4*.

2.2 SPECIAL CONSIDERATION FOR SAMPLING IN TIDALLY-INFLUENCED AREAS

Salinity and tides can be strong factors in the distribution of contaminants. The delineation of the point at which these effects are most pronounced, and the distribution of the highly contaminated sediments, might be confounded by these factors. For example, as contaminated water moves downstream, an abrupt increase in salinity can cause a sudden change in contaminant solubility. When less soluble, a contaminant may precipitate and appear in the sediment at substantially higher concentrations than the previous (i.e., upstream) location. These factors should be taken into consideration and assessed when making decisions regarding the selection of sample locations and relation of contaminants to the site.

Sediment sampling must be conducted during consistent tidal conditions. Either an ebb tide or flood tide interval is appropriate and shall be decided on a case-by-case basis. The tidal stage must be recorded. Samples must be collected from depositional areas (e.g., intertidal areas along the shoreline, which are often marked by emergent vegetation and muddy or organic bottoms, as well as mudflats, etc.).

2.3 CHEMICAL CHARACTERIZATION OF UPGRADIENT AND/OR OFFSITE REFERENCE CONDITIONS

When investigating sediment contamination in order to determine if it is linked to site operations, it is important to establish the chemical composition of upgradient sediments. These data also aid in the assessment of the site's contamination relative to the regional quality of the water body being investigated and in the development of remedial goals. The SRP recognizes that many of the State's water bodies, especially in urban/industrial settings, have become contaminated by historic point and non-point discharges, resulting in the diffuse, anthropogenic contamination of sediments at concentrations greater than natural background. Additionally, upgradient sediments can be contaminated by the site

because of tidal influences. While it is difficult to distinguish between site and non site-related contamination at these settings, it is the policy of NJDEP as well as USEPA Region II to make a reasonable attempt to do so. If potential sources of contamination are present upstream of the site, and it is believed that these sources have contributed to the contamination detected on-site, these upgradient areas should be sampled, and professional judgment should dictate how these data are to be interpreted/utilized (refer to Section 3.0). Note that these results will not be considered representative of true reference (i.e., natural background) conditions.

Certain site-specific conditions or study objectives may warrant the sampling of an offsite local reference location. The need for such data shall be determined on a case-by-case basis in consultation with BEERA/ETRA.

For upgradient and offsite reference locations, SRP recommends the collection of a minimum of three (3) to five (5) samples to establish a range of reference location contaminant concentrations (the larger number of samples is recommended due to sediment heterogeneity). Samples shall be collected from areas outside the site's potential influence. The samples must not be collected from locations directly influenced by or in close proximity to other obvious sources of contamination (i.e., other hazardous waste sites, sewer/storm water outfalls, tributaries, other point and non-point source discharges, etc.). If a local reference site is included in the sampling plan, it must be of comparable habitat to the study area. Upstream areas influenced by tides shall be sampled at locations determined to be within the mixing zone to delineate upstream migration of contaminants as well as upstream of any mixing zone in order to assess local ambient conditions. At a minimum, upgradient and local reference samples shall receive the same chemical analyses as site-related samples. Additional determinations, such as benthic community structure, may be required on a case-by-case basis.

2.4 SURFACE WATER QUALITY INVESTIGATIONS AND CRITERIA

Pursuant to *N.J.A.C. 7:26E-3.8 and 4.5*, a surface water investigation is required when there is evidence that surface water may have been impacted by site-related contamination. Additionally, since the release of contaminants from sediments may play a substantial role in surface water contamination, especially in quiescent aquatic systems such as lakes, wetlands, ponds and intermittent or slow moving streams, it is appropriate to include surface water samples in the overall assessment of sediment quality. Surface water quality data also serve as a tool for the interpretation of related biological test data.

Details for surface water sampling plan design, field sampling methodology, and analytical requirements are found in *N.J.A.C. 7:26E* and the *NJDEP FSPM*. As a general guide, surface water samples should be collected near banks/depositional areas where water current is slower and there is greater retention time for the surface water to accumulate contaminants from sediment. Since contaminated groundwater and surface water can serve as sources of sediment contamination, obvious surface-runoff channels, leachate seeps, groundwater discharge areas, etc., should be targeted. Determination of the number and location of samples should be made after all surface water migration pathways and discharge points have been identified; the potential for upstream contaminant migration in tidal water bodies must be addressed.

Surface water samples must be collocated spatially and temporally with sediment samples. In addition to bulk chemical analysis, measurements for salinity (in estuarine systems), pH, dissolved oxygen, and total hardness (as mg/1 CaCO₃) are required.

Surface water risks to aquatic receptors are evaluated based on comparison of measured concentrations with acute and chronic *Surface Water Quality Standards (N.J.A.C. 7:9B)* and surface water screening criteria where Surface Water Quality Standards (SWQS) do not exist. The SWQS and surface water screening criteria can be viewed at <http://www.nj.gov/dep/srp/guidance/ecoscreening/>. The most recent version of the list entitled *Surface Water Quality Criteria Applicable to New Jersey* can be obtained from the Standards Assessment and Modeling Unit, Office of Environmental Planning, at 609-633-7020. Those criteria that require a hardness value to derive the applicable criterion must employ a station-specific hardness value, not an average value.

For inorganic contaminants, it is recommended by the USEPA Region II Biological Technical Assistance Group (BTAG) and the SRP that both dissolved and total recoverable metals be measured. Most aquatic water quality criteria are based on the dissolved (filtered) form of the metal; however, the total recoverable (unfiltered) inorganic value is more indicative of total contaminant exposure and should be used for risk-management decision-making. Additionally, USEPA Office of

Water recommends that Superfund ecological risk assessments consider inorganics on a total recoverable basis to conservatively avoid underestimation of bioavailable metals. (USEPA, 1993). Together, the two sets of measurements are used to judge regulatory compliance as well as potential adverse ecological impact.

REFERENCES

N.J.A.C. 7:9B. Surface Water Quality Standards

N.J.A.C. 7:26E. Technical Requirements for Site Remediation

N. J. Department of Environmental Protection. 1992. *Field sampling procedures manual*. Trenton, NJ.

U. S. Environmental Protection Agency. July, 1985. *Sediment sampling quality assurance user's guide*. EPA/600/4-85/048. PB85-233542. Environmental Monitoring Systems Laboratory. Las Vegas, NV.

U.S. Environmental Protection Agency. October 1, 1993. Office of water policy and technical guidance on interpretation and implementation of aquatic life metals criteria. Office of Water. Washington.

3.0 SEDIMENT SCREENING VALUES FOR USE IN THE BASELINE ECOLOGICAL EVALUATION

To aid in the identification of contaminants of potential ecological concern, site-related sediment data are compared to established screening level criteria in the Baseline Ecological Evaluation (BEE). SRP's Bureau of Environmental Evaluation and Risk Assessment, Environmental Toxicology and Risk Assessment Unit (BEERA/ETRA) recommends the use of the sediment screening values on the three (3) attached tables for the purpose of identifying sediment contaminants of concern for a BEE. These values supersede those provided in *Guidance for Sediment Quality Evaluations, Final Draft for Internal Use Only, March 1991* and are applicable to traditional sediments and to wetland sediments if a benthic community is supported.

3.1 INORGANICS, SEMIVOLATILE ORGANICS, PESTICIDES/PCBs

The values presented in the Ecological Screening Criteria Table at <http://www.nj.gov/dep/srp/guidance/ecoscreening/> are extracted from references cited in N.J.A.C. 7:26E-3.11 and are used by USEPA Region II BTAG for EPA Screening Level Ecological Risk Assessments [and other published sources]. Freshwater sediment screening values used for the BEE are the Ontario Lowest Effects Levels (LEL) (Persaud et al., 1993), and marine/estuarine sediment screening values used for the BEE are the Effects Range-Low (ER-L) values (Long et al., 1995). Additional screening values to be used in the BEE are listed in the LEL and ER-L columns with sources cited.

3.2 VOLATILE ORGANICS

The values indicated in the Ecological Screening Criteria Table at <http://www.nj.gov/dep/srp/guidance/ecoscreening/> are to be used as sediment screening criteria. The values were obtained from Environment Canada's The Development of Canadian Marine Environmental Quality Guidelines (MacDonald et al., 1992) and other sources as cited.

3.3 TOTAL PETROLEUM HYDROCARBONS (TPHC)

There is currently no sediment screening value for TPHC, therefore TPHC-contaminated sediment should be analyzed for volatile and semivolatile organics and resultant data evaluated on a chemical-specific basis. If chemical analyses produce low or nondetectable levels of the expected organic compounds, but petroleum product is observable, the product is likely to cause adverse ecological effects (physical impairment of biota, loss of available substrate, etc.). A benthic macroinvertebrate survey (Section 4.2) in the affected area and in an appropriate reference location can be conducted to guide remedial decision-making. In general, sediments with TPHC contamination are managed on a case-by-case basis in consultation with BEERA/ETRA.

3.4 COMPARISON OF SITE-RELATED DATA TO SEDIMENT SCREENING VALUES

The following should be considered when comparing data from potentially impacted samples to sediment screening values:

1. In the BEE, maximum and mean concentrations of site-related and reference sample data are compared to the sediment screening values. No contaminants can be excluded from the evaluation without adequate justification. Contaminants may not be excluded from consideration based on comparison with background/reference location data until completion of the BEE because an evaluation of total site risk is appropriate at this stage.
2. The Long et al. marine/estuarine **ER-L** (Effects Range-Low) screens represent a concentration at which adverse benthic impacts are found in approximately 10% of studies. A level greater than the **ER-M** (Effects Range-Median) indicates a greater than 50% incidence of adverse effects to sensitive species and/or life stages. A concentration between the ER-L and ER-M therefore indicates an expected impact frequency between 10% and 50%.

Ontario's freshwater **LEL** (Lowest Effects Level) screen is generally comparable to Long et al., ER-Ls. Ontario has no ER-M, but does provide an **SEL** (Severe Effect Level) indicating **severe benthic impacts** in 95% of studies. For non-polar organics, the SEL is calculated via site-specific total organic carbon (TOC). See Table 1 footnotes for details on SEL calculation.

The ER-L and LEL screens were developed based on benthic community studies and do not directly address **biomagnification (food chain toxicity)** to water column species (fishes), birds, and mammals. However, values found to be protective of the food chain are generally similar (within an order of magnitude) to ER-L/LEL values. When **PCBs, organochlorine pesticides and mercury (Hg)** are found in sediments at or above these screens, potential wildlife risks exist and case-by-case evaluation is warranted. Other known biomagnifiers without Ontario or Long et al. screening numbers that warrant case-by-case evaluation are **dioxins, furans, other chlorinated organics, and selenium (Se)**.

3. The attached **ER-L** and **LEL** values are not cleanup standards but screening guidelines for use in the BEE. An exceedence indicates a **potential** risk (adverse impact) to the benthic community and need for further investigations, which would reduce uncertainty and better characterize risk and natural resource injury. Such investigations include toxicity testing, macroinvertebrate community surveys, and tissue bioassays. The determination for more rigorous studies should be made on a case-by-case basis in consultation with BEERA/ETRA.

Further remedial investigations/actions need not be triggered by BEE screening exceedences if sediments proximal to the site display contaminant concentration ranges similar to upgradient sediments, which may be impacted by other sources, diffuse anthropogenic contamination, etc. However, upgradient sediment data must not be used to eliminate contaminants of concern until the BEE has been completed. At that point, the determination of chemicals of concern retained for further evaluation will be addressed through the risk management process in consultation with the case team. Justification for no further action must be provided in the BEE for Department review and must contain site-specific upgradient data (refer to Section 2.3).

Risk assessment and **risk management** should be clearly distinguished. Local reference contaminant levels comparable to site levels do not indicate absence of site risk, but do indicate reference area and site risks that are similar. A risk management decision to forego further action is based on no observable additional site-generated risk.

4. A number of screening values for Polynuclear Aromatic Hydrocarbons (PAHs) are below Practical Quantitation Limits (PQLs) and Contract Required Detection Limits (CRQLs). To screen site data that are below the CRQL, the estimated values (indicated by a "J" data qualifier) are to be compared with the screening criteria.
5. Generally, sediments containing ppb-levels of non-persistent ($\log_{10} K_{OW} < 3$), photodegradable, non-polar **volatile organics** are not of ecological concern and further remedial investigation or remediation would not be warranted. However, this approach is conditioned upon no observable acute or chronic toxicity in the sediments, source removal, and compliance with associated Surface Water Quality Standards.
6. Where analytical detection limits are higher than screening criteria, contaminants must be retained as contaminants of concern. For this reason, detection limits for all analytes, including undetected contaminants, must be provided with all data summaries.

7. **Particle/grain size, pH, and TOC** analyses are required for all sediment investigations. These data confirm whether samples were collected in depositional zones, as indicated by relatively higher TOC values and a higher percentage of fine-grained particles, and provide a qualitative indication of bioavailability. Depositional zones are areas of highest potential contamination and must be targeted during sampling events.

TOC results may be used to interpret borderline screening exceedences in a weight of evidence /professional judgement decision, or to generate site specific screening values via an Equilibrium Partitioning (EP) approach (non-polar organics only, e.g., PCBs, PAHs, organochlorine pesticides). Some EPA sediment screening numbers, and some Ontario SELs, are generated via this approach; however, BEERA/ETRA and the USEPA Region II BTAG no longer use the EP approach for general screening purposes due to uncertainties regarding some of the assumptions used. Please consult **BEERA/ETRA (609-633-1348)** if a No Further Action (NFA) remedial decision is based on an EP approach or an EP approach is considered to have site-specific utility.

References for TOC (Kahn, 1988) and particle/grain size (ASTM, 1992) analyses are provided below. At a minimum, particle size analysis results must provide the percent clay, silt, sand and gravel.

8. If contaminant levels are marginally higher than screens or background, consult BEERA/ETRA prior to requiring additional studies, as a weight of evidence"/professional judgment approach may preclude the need for the studies.
9. If a screening value is not provided for a specific contaminant, it must be retained as a contaminant of concern. It is also recommended that BEERA/ETRA be contacted prior to conducting a literature search, since ETRA may be able to determine if a screening value is presently available. Published sediment screening values other than those cited in this guidance may be used on a case-by-case basis following consultation with ETRA .

REFERENCES

American Society for Testing and Materials (ASTM). 1992. *Standard guide for selection of methods of particle size analysis of fluvial sediments (manual methods)*. Method D4822-88. American Society for Testing and Materials. 1916 Race Street, Philadelphia, PA. Volume 11.02, pg. 622-624.

Kahn, L. 1988. *Determination of total organic carbon in sediment*. U.S. Environmental Protection Agency, Region II. Environmental Services Division, Monitoring Management Branch, Edison, NJ.

Long, E.R., MacDonald, D.D., Smith, S.L., and Calder, F.D. 1995. *Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments*. Environmental Management Vol.19, No.1. pp. 81-97.

MacDonald, D.D., Smith, S.L., Wong, M.P. and Murdoch, P. Environment Canada. 1992. *The development of Canadian marine environmental quality guidelines. Marine environmental quality series no. 1*. Ecosystem Sciences and Evaluation Directorate. Eco-Health Branch. Ottawa, Ontario. 121 pp.

Persaud, D., Jaagumagi, R., and Hayton, A. 1993. *Guidelines for the protection and management of aquatic sediment quality in Ontario*. ISBN 0-7729-9248-7. Ontario Ministry of the Environment, Ottawa, Ontario. 23p.

4.0 BIOLOGICAL EVALUATIONS FOR USE IN ECOLOGICAL RISK ASSESSMENT

4.1 SEDIMENT TOXICITY TESTING

Toxicity tests are used to expose test organisms to a medium (i.e., sediment) and to evaluate the effects of contamination on the survival, growth, reproduction, behavior and/or other attributes of these organisms. They provide important information that cannot be derived solely from chemical analysis nor from community surveys. The data assimilated by sediment toxicity tests can be used to: a) demonstrate the bioavailability of sediments contaminants, b) evaluate the aggregate toxic effects of all contaminants in a medium, c) evaluate the toxicity of substances whose biological effects may not have been well characterized, d) characterize the nature of a toxic effect, e) characterize the distribution of toxicity at a site, f) develop remedial goals, g) monitor the effectiveness of remedial actions, and h) determine a site's post-remedial potential to support a viable ecological community (USEPA, 1994).

When designing a toxicity assessment, one must consider the study objective, test site, reference site, medium analyzed, test organisms, test methodology, and quality assurance/quality control requirements. All of the above elements must be tailored to meet the site specific needs/goals of the investigation. The specific type and technique of sediment toxicity test appropriate in a particular situation will be determined by a variety of site-specific factors. These include, but are not limited to, type and salinity of water body present, nature and extent of contamination, local biota, and site-specific informational needs. Numerous studies have shown that different testing regimes with the same sediment and organism can result in different bioassay responses. Additionally, bioassays with different organisms conducted on the same sediment do not always give similar results. For these reasons, it is imperative that a sediment bioassay program not rely on a single species endpoint. No single test is adequate to allow a detection of an impact among the various toxicants or stresses present at hazardous waste sites.

At a minimum, a sediment toxicity test shall incorporate the following:

1. Both **acute** (i.e., survival) and **sub-chronic** (i.e., growth, reproductive capacity) endpoint measurements.
2. The use of two (2) test organisms, preferably representing two different ecological niches (e.g., one infaunal and one epifaunal species).
3. Each sediment sample collected and slated for sediment toxicity testing shall also be analyzed for the chemical contaminants of concern associated with the site. The sample shall be obtained directly from the bulk sediment intended to be used for the sediment toxicity test.
4. Sediment samples must be maintained in the dark at 4°C prior to beginning toxicity testing.
5. A control sediment sample should be tested in addition to the reference sample, and is usually supplied along with the cultured organisms.
6. Five (5) test replicates per sample.
7. Two (2) weeks is the maximum allowable holding time for sediments used in toxicity tests.
8. For work conducted under SRP oversight, the source of the reference sediments and overlay water, intended procedures for endpoint measurement, and statistical analyses for results, etc., should be provided to the SRP via a work plan prior to commencement.

As previously stated, the particular tests that are selected will be determined by site-specific characteristics and needs. The following list of references can serve as a starting point in the selection of appropriate tests but should not be considered as all inclusive. It is highly recommended that BEERA/ETRA is consulted prior to the selection and implementation of a sediment toxicity test.

REFERENCES

American Society for Testing and Materials (ASTM). 1992. *Standard guide for conducting sediment toxicity tests with freshwater invertebrates*. American Society for Testing and Materials, Philadelphia, PA. 23 pp.

American Society for Testing and Materials (ASTM). 1992. *Standard guide for conducting 10-day static sediment toxicity tests with marine and estuarine amphipods*. American Society for Testing and Materials, Philadelphia, PA. 24 pp.

U.S. Environmental Protection Agency. June 1994. *Methods for assessing the toxicity of sediment-associated contaminants with estuarine and marine amphipods*. EPA/600/R-94/025. Office of Research and Development, Narragansett, RI.

U.S. Environmental Protection Agency. June 1994. *Methods for assessing the toxicity of and bioaccumulation of sediment-associated contaminants with freshwater invertebrates*. EPA/600/R-94/024. Office of Research and Development, Narragansett, RI.

U.S. Environmental Protection Agency. September 1994. *Using toxicity tests in ecological risk assessment*. ECO Update. Publication 9345.0-051. EPA 540-F-94-012. PB94-963303. Intermittent Bulletin, Volume 2, Number 1.

U.S. Environmental Protection Agency. September 1994. *Catalogue of standard toxicity tests in ecological risk assessment*. Publication 9345.0-051. EPA 540-F-94-013. PB94-963304. Intermittent Bulletin, Volume 2, Number 2.

4.2 BENTHIC MACROINVERTEBRATE SURVEYS

Benthic macroinvertebrate surveys have been performed historically by USEPA and state regulatory agencies to evaluate the ecological integrity of aquatic systems as mandated by specific sections of the Clean Water Act. Recently, such evaluations have been used, in conjunction with other methodologies (i.e. sediment toxicity tests, sediment chemistry data), to assess the health of aquatic systems associated with the investigation of hazardous waste sites.

Assessments of benthic macroinvertebrate community structure and function are used extensively to provide direct evidence of contaminant-related effects in the environment. Benthic macroinvertebrates are relatively sedentary organisms that inhabit or depend upon the sediment environment for their various life functions. They are sensitive to both long term and short-term changes in sediment and water quality. Benthic macroinvertebrates are frequently used as environmental indicators of biological integrity because they are found in most aquatic habitats, are of a size permitting ease of collection, and can be used to describe water quality conditions or health of ecosystem components, and to identify causes of impaired conditions (USEPA, 1990). A wide variety of procedures have been developed to evaluate how changes in environmental quality affect benthic communities. A complete description of these methods is beyond the scope of this document. However, these procedures can be divided into those that measure community structure and those that measure community function. Community structure is the measurement of biotic characteristics (i.e., species abundance, diversity, and composition) at a point in time, whereas community function is the measurement of rate processes (i.e., species colonization rates) of the ecosystem. The use of biological communities in environmental monitoring is normally done from a structural perspective because structural studies usually take less time, are more conventional, and facilitate comparisons with data from other studies. It must be kept in mind, however, that contamination is not the only factor capable of changing community structure. Changes in salinity, temperature, dissolved oxygen, pH, Eh, sediment texture, and shading can all effect community structure.

The specifics on sampling strategy, collection, identification, data reduction, and interpretation of results will depend upon site-specific conditions and requirements. It is important that benthic macroinvertebrate studies be carefully designed as confounding effects not related to pollution (e.g., natural temporal and spatial variability, competition, predation, sediment type, salinity, sample depth, season of sampling, sediment pH) can profoundly influence study results. At a minimum, it is essential that all locations selected for macroinvertebrate surveys also undergo sediment chemistry analyses. The sediment used for the chemical analyses shall be obtained at the same location and time of the macroinvertebrate survey.

It is recommended that the guidance documents listed below be consulted for work plan development. As previously stated, the particular type of survey selected will be determined by site-specific characteristics and data needs. As the decisions regarding the selection of procedures and methodologies to be used in the macroinvertebrate survey are often complex, it is recommended that the macroinvertebrate survey work plan be discussed with BEERA/ETRA prior to implementation.

REFERENCES

U.S. Environmental Protection Agency. May 1989. *Rapid bioassessment protocols for use in streams and rivers - benthic macroinvertebrates and fish*. EPA/440/4-89/001. Assessment and Watershed Protection Division, Washington.

U.S. Environmental Protection Agency. November 1990. *Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface waters*. EPA/600/4-90/030. Environmental Monitoring Systems Laboratory, Cincinnati, OH.

U.S. Environmental Protection Agency. 1997. *Field and laboratory methods for macroinvertebrates and habitat assessment of low gradient nontidal streams*. Mid-Atlantic Coastal Streams Workgroup, Environmental Services Division, Region 3, Wheeling, WV.

4.3 TISSUE RESIDUE ANALYSIS

Many contaminants found at hazardous waste sites are capable of being transferred from the sediment, water, and diet to biota. These contaminants can accumulate within tissues of organisms to levels that greatly exceed ambient concentrations. Bioaccumulation can result in acute and chronic effects (including adverse effects on reproduction) on individual organisms and also expose predators to toxic doses of contaminants. Biomagnification is the total process by which tissue concentrations of bioaccumulated compounds increase as compounds are transferred up the food chain.

During ecological/sediment quality investigations, the purpose of tissue residue analysis is to measure whole body contaminant concentrations in prey species consumed by a predatory species of concern. This will provide a usable estimate of the exposure dose to the species of concern and allow comparison with literature-based No Observed Adverse Effect Levels (NOAEL) and/or Lowest Observed Adverse Effect Levels (LOAEL) for the purpose of estimating risk. Also, a protective sediment clean-up number based on the NOAEL/LOAEL can be estimated knowing (1) the concentration of a given contaminant in fish tissue corresponding to the LOAEL/NOAEL for adverse effects to a species of concern and (2) the relationship between the contaminant levels in sediments and in the forage species (site-specific bioaccumulation factor).

Considerations for a tissue analysis study include, but are not limited to, the following:

1. Species Selection - the aquatic species selected for sampling will depend on site-specific data requirements and ecologic characteristics. The organisms should ideally have a small home range and forage within the study area, overlapping areas of maximum contamination. The species selected must be sufficiently abundant that adequate numbers of individuals can be collected to achieve the necessary sample mass required for analysis. Predatory species of concern, feeding guilds of interest, lipid content, etc. should all be considered. "Back-up" species should be selected in the event that the recommended target species are not able to fulfill the study's objectives.

- a. Fish

Fish are useful tools in monitoring biological uptake and have proven to be good indicators of both inorganic and organic contamination. Fish species are used in various environmental monitoring capacities creating an extensive database for background levels of many compounds. Care must be taken in choosing among fish species to be sampled, as many fish species have a large home range and/or are migratory, thus would not be entirely indicative of local conditions. When appropriate, fish species should be selected that are present year round. If measurement of maximum accumulation is desired, the species should be high in lipid content, long-lived, and closely associated with the sediment. Two fresh water species that meet these criteria and are commonly used in sediment monitoring programs are the common carp (*Cyprinus carpio*) and brown bullhead catfish (*Ictalurus nebulosus*). Mummichogs (*Fundulus heteroclitus*) are a marine/estuarine species that has been used successfully at several SRP sites. If only fin fish species are to be collected for tissue residue analysis, two different trophic levels should be represented.

- b. Mollusks/Crustaceans

Mollusks and crustaceans have been successfully used to monitor biological uptake of sediment contaminants. The behavior of these species, which places them in direct contact with sediment, make them particularly useful in measuring the potential for biological uptake of sediment contaminants. Species that have been used in biological sampling programs in the SRP include blue claw crab (*Callinectes sapidus*), grass shrimp (*Palaemonetes* spp.), soft shell clam (*Mya arenaria*), fiddler crab (*Uca minax*), and bent-nose macoma clam (*Macoma nasuta*).

2. Seasonality

The season during which biological samples are collected for tissue analysis is an important consideration. The spawning and breeding season should be avoided whenever possible because aquatic species are often stressed at this time, having different feeding habits, fat content, and respiration rates, which can influence pollution uptake and clearance. Generally, the most appropriate sampling period is from late summer to early fall (i.e., August through October), when the lipid content of many species is generally highest after a full, active season of consumption and contaminant accumulation. Also, fresh water levels are typically lower during this time, facilitating sample collection.

3. Sample Compositing

Because a sample mass of 20g to 50g is typically required for analysis, individuals are routinely composited. Individual organisms used in composite samples must be of the same species because bioaccumulation potential is species-specific. Accurate taxonomic identification is essential to prevent the compositing of closely related species. The sample must be a whole-body, soft tissue composite, assuming the whole organism is consumed.

Sample composites must be segregated based on age and sex. BEERA/ETRA generally recommends sampling adults, which will have had a greater opportunity for contaminant accumulation. The sampler should be aware of situations which could introduce bias into results. For example, samples containing high ratios of gravid females could dramatically increase concentrations of contaminants known to biomagnify. As another example, the large claw and muscle tissue of the mature male fiddler crab generally have lower levels of contaminants than more lipid-rich digestive and reproductive organs; results from a composite sample containing a greater proportion of mature males would likely be biased low due to sex differences rather than from site conditions.

BEERA/ETRA generally recommends three (3) to five (5) replicate composite tissue samples of each target species at each sample location.

It is highly recommended that the references cited below be consulted for further information on tissue sample collection, sample preparation, and analytical methods.

REFERENCES

U.S. Department of Commerce. 1993. *Sampling and analytical methods of the National Status and Trends Program, National Benthic Surveillance and Mussel Watch Projects, 1984 - 1992. Volume IV. Comprehensive descriptions of trace organic analytical methods.* NOAA Technical Memorandum NOS ORCA 71. National Oceanic and Atmospheric Administration, Coastal Monitoring and Bioeffects Assessment Division, Ocean Resources Conservation and Assessment, National Ocean Service, Silver Spring, MD.

U.S. Environmental Protection Agency. 1993. *Guidance for assessing chemical contaminant data for use in fish advisories. Volume 1: Fish sampling and analysis.* EPA 823-R-93-0 Office of Science and Technology, Office of Water, Washington.

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	Soil (mg/kg)			
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria				EcoSSLs ²⁰			
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian
		Acute	Chronic		Acute	Chronic											
Acenaphthene	83-32-9		38 ^b	670(h)			990(h)	See Saline Criteria ³ 0.00671 ^b		0.016	0.500	20 ⁹					
Acenaphthylene	208-96-8		4840 ^b					See Saline Criteria ³ 0.00587 ^b		0.044	0.640	682 ^b					
Acrolein	107-02-8		0.19 ^b	6.1(h)			9.3(h)	0.0000152 ^b				5.27 ^b					
Acrylonitrile	107-13-1		66 ^b	0.051(hc)			0.25(hc)	0.0012 ^b				0.0239 ^b					
Aldrin	309-00-2	3	0.017 ^b	0.000049(hc)	1.3		0.000050(hc)	0.002	8	See Freshwater Criteria ⁶		0.00332 ^b					
Aluminum	7429-90-5							2.55% ¹⁵			1.8% ¹⁵		50				
Ammonia, un-ionized	7664-41-7	See N.J.A.C. 7:9B-1.14(e)			See N.J.A.C. 7:9B-												
Anthracene	120-12-7		0.035 ^b	8,300(h)			40,000(h)	0.0572 ^b	370	0.085	1.1	1,480 ^b					
Antimony	7440-36-0		80 ^b	5.6(h)(T)			640(h)(T)		3 ¹⁵		9.3 ¹⁵	5 ⁹	5		78		0.27
Arsenic	7440-38-2	340(d)(s)	150(d)(s)	0.017(hc)(T)	69(d)(s)	36(d)(s)	0.061(hc)(T)	6 9.9790 ^b	33	8.2	70	9.9 ^{9,10}	10	18		43	46
Asbestos	1332-21-4			7x10 ⁶ fibers/L >10um(h)													
Barium	7440-39-3		220 ^b	2,000(h)(T)							48 ¹⁵	283 ¹¹	500		330		2,000
Benz(a)anthracene	56-55-3		0.025 ^b	0.038(hc)			0.18(hc)	0.320 0.108 ^b	1,480	0.261	1.6	5.21 ^b					
Benzene	71-43-2		114 ^b 824 ¹⁶	0.15(hc)			3.3(hc)	See Saline Criteria ³ 0.142 ^b		0.34 ⁷		0.255 ^b					
Benzidine	92-87-5			0.000086(hc)			0.00020(hc)										
3,4-Benzofluoranthene (Benzo(b)fluoranthene)	205-99-2		9.07 ^b	0.038(hc)			0.18(hc)	10.4 ^b			1.800 ¹⁵	59.8 ^b					
Benzo(k)fluoranthene	207-08-9			0.38(hc)			1.8(hc)	0.240	1,340	See Freshwater Criteria ⁶		148 ^b					
Benzo(g,h,i)perylene	191-24-2		7.64 ^b					0.170 0.37	320	See Freshwater Criteria ⁶		119 ^b					
Benzo(a)pyrene (BaP)	50-32-8		0.014 ^b	0.0038(hc)			0.018(hc)	0.150 ^b	1,440	0.430	1.6	1.52 ^b					
Beryllium	7440-41-7		3.6 ^b	6.0(h)(T)			42(h)(T)					10 ⁹	10		40		21
BHC (Benzohexachloride)								0.003	12	See Freshwater Criteria ⁶							
alpha-BHC (alpha-HCH)	319-84-6		12.4 ^b	0.0026(hc)			0.0049(hc)	0.006	10			0.0994 ^b					
beta-BHC (beta-HCH)	319-85-7		0.495 ^b	0.0091(hc)			0.017(hc)	0.005	21			0.00398 ^b					
gamma-BHC (gamma- HCH/Lindane)	58-89-9	0.95	0.026 ^b	0.98(h)	0.16		1.8(h)	0.003	1			0.00500 ^b					
Biphenyl	92-52-4											60 ⁹					
Bis(2-chloroethyl) ether	111-44-4		1900 ^b	0.030(hc)			0.53(hc)	3.520 ^b				23.7 ^b					
Bis(2-chloroisopropyl) ether	108-60-1			1,400(h)			65,000(h)					19.9 ^b					
Bis(2-ethylhexyl) phthalate	117-81-7		0.3 ^b	1.2(hc)			2.2(hc)	0.182 ^b	0.750 ¹⁵	0.18216 ¹⁵	2.64651 ¹⁵	0.925 ^b					
Boron	7440-42-8											0.5 ⁹	0.5				
Bromine	7726-95-6											10 ⁹	10				
Bromodichloromethane (Dichlorobromomethane)	75-27-4			0.55(hc)			17(hc)					0.540 ^b					

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Soil (mg/kg)						
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria		Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	EcoSSLs ²⁰				
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian	
		Acute	Chronic		Acute	Chronic												
Bromoform	75-25-2		230 ^b	4.3(hc)			140(hc)	0.492 ^b				15.9 ^b						
Butyl benzyl phthalate	85-68-7		23 ^b	150(h)			190(h)	1.970 ^b			0.063 ¹⁵	0.239 ^b						
Cadmium	7440-43-9	(a)	(a)	3.4(h)(T)	40(d)(s)	8.8(d)(s)	16(h)(T)	0.990 ^b	10	1.2	9.6	4 ^{9,11}	4	32	140	0.77	0.36	
Carbon tetrachloride	56-23-5		240 ^b	0.33(hc)			2.3(hc)	1.450 ^b				2.98 ^b						
Chlordane	57-74-9	2.4	0.0043	0.00010(hc)	0.09	0.004	0.00011(hc)	0.00324 ^b	6	See Freshwater Criteria ⁶		0.224 ^b						
Chloride	6	860,000	230,000	250,000														
Chlorine Produced Oxidants (CPO)	7782-50-5	19	11		13	7.5												
3-Chloroaniline	108-42-9											20 ⁹	20					
Chlorobenzene	108-90-7		47 ^b	210(h)			2,500(h)	0.291 ^b				40 ¹²						
Chloroform	67-66-3		140 ^b	68(h)			2,100(h)	0.121 ^b				13.1 ^b						
2-Chloronaphthalene	91-58-7		0.396 ^b	1,000(h)			1,600(h)	0.417 ^b				1.19 ^b						
2-Chlorophenol	95-57-8		24 ^b	81(h)			150(h)	0.0319 ^b			0.008 ¹⁵	0.0122 ^b						
3-Chlorophenol	108-43-0											0.243 ^b						
Chlorpyrifos	2921-88-2	0.083	0.041		0.011	0.0056						7 ¹²	7					
Chromium	7440-47-3		42 ^b	92(h)(T)			750(h)(T)	26				43.4 ^b	110	81	370	0.4 ¹²	1	
Chromium+3	1	(a)	(a)														26	34
Chromium+6	9	15(d)(s)	10(d)(s)		1,100(d)(s)	50(d)(s)												130
Chrysene	218-01-9			3.8(hc)			18(hc)	0.34				0.166 ^b	460	0.384	2.8	4.73 ^b		
Cobalt	7440-48-4		24 ^b					50 ^b				20 ⁹						
Copper	7440-50-8	(a)	(a)	1,300(h)(T)	4.8(d)(s)	3.1(d)(s)		16	110	34	270	0.14 ^b	20	13		120	230	
Cyanide (Total)	57-12-5	22(fc)	5.2(fc)	140(h)	1.0(fc)	1.0(fc)	140(h)	0.0001 ^b				60 ¹²						
4,4'-DDD (p,p'-TDE)	72-54-8			0.00031(hc)			0.00031(hc)	0.00488 ^b	6	0.002 ¹⁵	0.02 ¹⁵	5.4 ^b	100	70	80	28	49	
4,4'-DDE	72-55-9		0.00000000451 ^b	0.00022(hc)			0.00022(hc)	0.005				1.33 ^b						
4,4'-DDT	50-29-3	1.1	0.001	0.00022(hc)	0.13	0.001	0.00022(hc)	0.008				0.596 ^b						
DDT (Total)								0.00416 ^b	71	0.001 ¹⁵	0.007 ¹⁵	0.0035 ^b						
Demeton	8065-48-3		0.1			0.1		0.007	12	0.0016	0.046						0.093 ²¹	0.021 ²¹
Dibenz(a,h)anthracene	53-70-3			0.0038(hc)			0.018(hc)	0.06				18.4 ^b						
Dibromochloromethane (Chlorodibromomethane)	124-48-1			0.40(hc)			13(hc)	0.033 ^b	130	0.063	0.26	2.05 ^b						
Di-n-butyl phthalate	84-74-2		9.7 ^b	2,000(h)			4,500(h)	1.114 ^b	0.110 ¹⁵		0.058 ¹⁵	200 ⁹						
1,2-Dichlorobenzene	95-50-1		14 ^b	2,000(h)			6,200(h)	0.294 ^b				0.15 ^b						
1,3-Dichlorobenzene	541-73-1		38 ^b	2,200(h)			8,300(h)	1.315 ^b				2.96 ^b						
1,4-Dichlorobenzene	106-46-7		9.4 ^b	550(h)			2,200(h)	0.318 ^b			0.110 ¹⁵	37.7 ^b						
3,3'-Dichlorobenzidine	91-94-1		4.5 ^b	0.021(hc)			0.028(hc)	0.127 ^b				20 ¹²						
1,2-Dichloroethane	107-06-2		910 ^b	0.29(hc)			28(hc)	0.260 ^b				0.546 ^b						
												21.2 ^b						

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Soil (mg/kg)					
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria		Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	EcoSSLs ²⁰			
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian
		Acute	Chronic		Acute	Chronic											
1,1-Dichloroethylene	75-35-4		65 ^B	4.7(h)			100(h)	0.0194 ^B				8.28 ^B					
trans-1,2-Dichloroethylene	156-60-5		970 ^B	590(h)			43,000(h)	0.654 ^B				0.784 ^B					
2,4-Dichlorophenol	120-83-2		11 ^B	77(h)			290(h)	0.0817 ^B			0.005 ¹⁵	87.5 ^B					
3,4-Dichlorophenol	95-77-2											20 ^{9,12}	20				
1,2-Dichloropropane	78-87-5		360 ^B	0.50(hc)			15(hc)	0.333 ^B				32.7 ^B					
1,3-Dichloropropene (cis and trans)	542-75-6			0.34(hc)			21(hc)										
Dieldrin	60-57-1	0.24	0.056	0.000052(hc)	0.71	0.0019	0.000054(hc)	0.0019 ^B	91	See Freshwater Criteria ⁶		0.00238 ^B				0.022	0.0049
Diethyl phthalate	84-66-2		110 ^B	17,000(h)			44,000(h)	0.295 ^B			0.006 ¹⁵	100 ⁹					
2,4-Dimethyl phenol	105-67-9		100 ^B	380(h)			850(h)	0.304 ^B				24.8 ^B					
4,6-Dinitro-o-cresol	534-52-1			13(h)			280(h)					0.010 ^B					
2,4-Dinitrophenol	51-28-5		19 ^B	69(h)			5,300(h)	0.00621 ^B				20 ⁹					
2,4-Dinitrotoluene	121-14-2		44 ^B	0.11(hc)			3.4(hc)	0.0144 ^B				0.0609 ^B	20				
1,2-Diphenylhydrazine	122-66-7			0.036(hc)			0.20(hc)					1.28 ^B					
Endosulfans (alpha and beta)	115-29-7	0.22	0.056	62(h)	0.034	0.0087	89(h)										
Endosulfan sulfate	1031-07-8		2.22 ^B	62(h)			89(h)	0.0346 ^B				0.0358 ^B					
Endrin	72-20-8	0.086	0.036	0.059(h)	0.037	0.0023	0.060(h)	0.00222 ^B	130	See Freshwater Criteria ⁶		0.0101 ^B					
Endrin aldehyde	7421-93-4		0.15 ^B	0.059(h)			0.060(h)	0.480 ^B				0.0105 ^B					
Ethylbenzene	100-41-4		14 ^B 81 ¹⁶	530(h)			2,100(h)	See Saline Criteria ³ 0.175 ^B 0.75		1.4 ⁷		5.16 ^B					
Fluoranthene	206-44-0		1.9 ^B	130(h)			140(h)	0.423 ^B 0.19	1,020	0.600	5.1	122 ^B					
Fluorene	86-73-7		19 ^B	1,100(h)			5,300(h)	0.0774 ^B	160	0.019	0.54	122 ^B					
Fluorine	7782-41-4											200 ⁹	200				
Furan	110-00-9											600 ⁹					
Guthion	86-50-0		0.01			0.01											
Heptachlor	76-44-8	0.52	0.0038	0.000079(hc)	0.053	0.0036	0.000079(hc)	0.0006 ^B 0.005	0.010 ¹⁵		0.0003 ¹⁵	0.00598 ^B					
Heptachlor epoxide	1024-57-3	0.52	0.0038	0.000039(hc)	0.053	0.0036	0.000039(hc)	0.00247 ^B	5	See Freshwater Criteria ⁶		0.152 ^B					
Hexachlorobenzene	118-74-1		0.0003 ^B	0.00028(hc)			0.00029(hc)	0.020	24	See Freshwater Criteria ⁶		0.199 ^B					
Hexachlorobutadiene	87-68-3		0.053 ^B	0.44(hc)			18(hc)	0.0265 ^B			0.0013 ¹⁵	0.0398 ^B					
Hexachlorocyclopentadiene	77-47-4		77 ^B	40(h)			1,100(h)	0.901 ^B				10 ⁹					
Hexachloroethane	67-72-1		8 ^B	1.4(hc)			3.3(hc)	0.584 ^B			0.073 ¹⁵	0.755 ^B					
Indeno(1,2,3-cd)pyrene	193-39-5		4.31 ^B	0.038(hc)			0.18(hc)	0.200	320	See Freshwater Criteria ⁶		109 ^B					
Iodine	7553-56-2											4 ⁹	4				
Isophorone	78-59-1		920 ^B	35(hc)			960(hc)	0.432 ^B				139 ^B					
Lead	7439-92-1	38(d)(s)	5.4(d)(s)	5.0(h)(T)	210(d)(s)	24(d)(s)		31 35.8 ^B	250	47	218	40.5 ¹¹ 0.0537 ^B	50	120	1,700	11	56
Lithium	7439-93-2											2 ⁹	2				
Malathion	121-75-5		0.1			0.1											

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Soil (mg/kg)					
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria		Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	EcoSSLs ²⁰			
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian
		Acute	Chronic		Acute	Chronic											
Manganese	7439-96-5						100(h)(T)	630 ¹⁵	1,100 ¹⁵				500	220	450	4,300	4,000
Mercury	7439-97-6	1.4(d)(s)	0.77(d)(s)	0.050(h)(T)	1.8(d)(s)	0.94(d)(s)	0.051(h)(T)	0.2	2	0.15	0.71	0.00051 ¹¹	0.3				
Methoxychlor	72-43-5		0.03	40(h)		0.03		0.174 ⁸				0.1 ⁸					
Methyl bromide (bromomethane)	74-83-9			47(h)			1,500(h)	0.0136 ⁸				0.0199 ⁸					
Methyl t-butyl ether (MTBE)	1634-04-4	151,000 ¹⁷	51,450 ¹⁶	70(h)	53,000 ¹⁷	18,000 ¹⁷											
Methylene chloride	75-09-2		940 ⁸	2.5(hc)			310(hc)	0.00137 ⁸				0.235 ⁸					
2-Methylnaphthalene	91-57-6		330 ⁸					See Saline Criteria ³		0.070	0.67	4.05 ⁸					
Mirex	2385-85-5		0.001			0.001		0.0202 ⁸		See Freshwater Criteria ⁵		3.24 ⁸					
Molybdenum	7439-98-7											2 ⁹	2				
Naphthalene	91-20-3		13 ⁸					See Saline Criteria ³				0.0994 ⁸					
Nickel	7440-02-0	(a)	(a)	500(h)(T)	64(d)(s)	22(d)(s)	1,700(h)(T)	0.176 ⁸		0.16	2.1	30 ⁹	30	38	280	210	130
Nitrate (as N)	8			10,000(h)				16	75	21	52	13.6 ⁸					
Nitrobenzene	98-95-3		220 ⁸	17(h)			690(h)	22.7 ⁸				7 ¹²					
4-Nitrophenol	100-02-7		60 ⁸					0.145 ⁸				5.12 ⁸					
N-Nitrosodi-n-butylamine	924-16-3			0.0063(hc)			0.22(hc)	0.0133 ⁸									
N-Nitrosodiethylamine	55-18-5		768 ⁸	0.00023(hc)			0.13(hc)	16				0.0693 ⁸					
N-Nitrosodimethylamine	62-75-9			0.00069(hc)			3.0(hc)	0.0228 ⁸				0.0000321 ⁸					
N-Nitrosodiphenylamine	86-30-6			3.3(hc)			6.0(hc)					0.545 ⁸					
N-Nitrosodi-n-propylamine (Di-n-propylnitrosamine)	621-64-7			0.0050(hc)			0.51(hc)										
N-Nitrosopyrrolidine	930-55-2			0.016(hc)			34(hc)					0.0126 ⁸					
Parathion	56-38-2	0.065	0.013					0.000757 ⁸				0.00034 ⁸					
Pentachlorobenzene	608-93-5		0.019 ⁸	1.4(h)			1.5(h)					20 ¹²					
Pentachlorophenol	87-86-5	(b)	(b)	0.27(hc)	13	7.9	3.0(hc)	0.024 ⁸				0.497 ⁸	3	5.0	31	2.1	2.8
Phenanthrene	85-01-8		3.6 ⁸					23 ⁸	950	0.240	1.5	0.119 ⁸					
Phenol	108-95-2		180 ⁸	10,000(h)			860,000(h)	0.56				45.7 ⁸					
Phosphorous (yellow)	7723-14-0					0.1		0.204 ⁸				30 ¹²					
PCB Aroclor 1016								0.0491 ⁸	0.048 ¹⁵			120 ⁸					
PCB Aroclor 1248								0.007	53	See Freshwater Criteria ⁶							
PCB Aroclor 1254								0.030	150	See Freshwater Criteria ⁶							
PCB Aroclor 1260								0.060	34	See Freshwater Criteria ⁶							
PCB Aroclor 1260								0.005	24	See Freshwater Criteria ⁶							

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Soil (mg/kg)					
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria		Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	EcoSSLs ²⁰			
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian
		Acute	Chronic		Acute	Chronic											
Polychlorinated biphenyls (PCBs)	1336-36-3		0.014	0.000064(hc)		0.03	0.000064(hc)	0.07 0.0598 ⁸	530	0.023	0.180	0.371 ¹⁰ 0.000332 ⁸	40				
Pyrene	129-00-0		0.30 ⁸	830(h)			4,000(h)	0.490 0.195 ⁸	850	0.665	2.6	78.5 ⁹ 0.21 ¹³					
Selenium	7782-49-2	20(s)	5.0(s)	170(h)(T)	290(d)(s)	71(d)(s)	4,200(h)(T)					1 ¹⁵	0.52	4.1	1.2	0.63	
Silver	7440-22-4	(a)	0.12 ⁸	170(h)(T)	1.9(d)(s)		40,000(h)(T)	See Saline Criteria ³ 0.5 ⁸		1.0	3.7	2 ⁹ 4.04 ⁸ 300 ⁹ 4.69 ⁸	2	560		4.2	14
Styrene	100-42-5		32 ⁸					0.254 ⁸					300				
Sulfide-hydrogen sulfide (undissociated)	7783-06-4		2			2											
TCDF												0.00084 ¹⁴					
Technetium	7440-26-8											0.2 ⁹	0.2				
tert-Butyl alcohol (TBA)	75-65-0		355,000 ¹⁶														
2,3,5,6-Tetrachloroaniline	3481-20-7											20 ⁹	20				
1,2,3,4-Tetrachlorobenzene	634-66-2											10 ⁹					
1,2,4,5-Tetrachlorobenzene	95-94-3		3 ⁸	0.97(h)			1.1(h)	1.252 ⁸				2.02 ⁸					
2,3,7,8-Tetrachlorodibenzo-p-dioxin	1746-01-6		0.000000003 ⁸	0.0000000050(hc)			0.0000000051(hc)	0.00000012 ⁸	0.0000088 ¹⁵			0.00000315 ¹⁰					
1,1,2,2-Tetrachloroethane	79-34-5		380 ⁸	4.7(h)			110(h)	0.850 ⁸				0.127 ⁸					
Tetrachloroethylene	127-18-4		45 ⁸	0.34(hc)			1.6(hc)	See Saline Criteria ³ 0.990 ⁸		0.45 ⁷		9.92 ⁸					
2,3,4,5-Tetrachlorophenol	4901-51-3											20 ¹²					
Thallium	7440-28-0		10 ⁸	0.24(h)(T)			0.47(h)(T)					1 ⁹	1				
Tin	7440-31-5		180 ⁸									>3.4 ¹⁵	50 ⁹	50			
Toluene	108-88-3		253 ⁸ 822 ¹⁶	1,300(h)			15,000(h)	See Saline Criteria ³ 1.220 ⁸		2.5 ⁷		200 ⁹	200				
Toxaphene	8001-35-2	0.73	0.0002	0.00028(hc)	0.21	0.0002	0.00028(hc)	0.000077 ⁸				0.119 ⁸					
2,4,5-Trichloroaniline	636-30-6											20 ⁹	20				
1,2,3-Trichlorobenzene	87-61-6											20 ¹²					
1,2,4-Trichlorobenzene	120-82-1		30 ⁸	21(h)			42(h)	5.062 ⁸				>0.0048 ¹⁵	20 ¹²				
1,1,1-Trichloroethane	71-55-6		76 ⁸	120(h)			2,600(h)	0.213 ⁸				29.8 ⁸					
1,1,2-Trichloroethane	79-00-5		500 ⁸	13(h)			350(h)	0.518 ⁸				28.6 ⁸					
Trichloroethylene	79-01-6		47 ⁸	1.0(hc)			12(hc)	See Saline Criteria ³ 0.112 ⁸		1.6 ⁷		12.4 ⁸					
2,4,5-Trichlorophenol	95-95-4			1,800(h)			3,600(h)					9 ¹²	4				
2,4,6-Trichlorophenol	88-06-2		4.9 ⁸	0.58(hc)			1.0(hc)	0.208 ⁸				0.006 ¹⁵	4 ⁹				
Uranium	7440-61-1											5 ⁹	5				
Vanadium	7440-62-2		12 ⁸									57 ¹⁵	2 ⁹			7.8	280
Vinyl chloride	75-01-4		930 ⁸	0.082(hc)			8.1(hc)	0.202 ⁸				0.646 ⁸					
Xylene	1330-20-7		27 ⁸ 296 ¹⁶					See Saline Criteria ³ 0.433 ⁸				>0.12 ⁷	10 ⁸				

NJDEP Ecological Screening Criteria

Toxic Substance	CAS Number	Surface Water (ug/L)						Sediment (mg/kg)				Soil (mg/kg)					
		Fresh Water (FW2) Criteria			Saline Water (SE & SC) Criteria			Fresh Water Criteria		Saline Water Criteria		Wildlife PRGs (flora and fauna)	Terrestrial Plant Tox Benchmarks	EcoSSLs ²⁰			
		Aquatic		Human Health	Aquatic		Human Health	Lowest Effects Level (LEL) ¹	Severe Effects Level (SEL) ²	Effects Range Low (ER-L) ⁴	Effects Range Medium (ER-M) ⁵			Plants	Soil Invertebrates	Avian	Mammalian
		Acute	Chronic		Acute	Chronic											
Zinc	7440-66-6	(a)	(a)	7,400(h)(T)	90(d)(s)	81(d)(s)	26,000(h)(T)	120 121 ⁸	820	150	410	8.5 ¹¹ 6.62 ⁸	50	160	120	46	79
Low Molecular Weight PAHs ¹⁸															29		100
High Molecular Weight PAHs ¹⁹															18		1.1
Total PAHs								4.0	10,000	4.0	45.0						

(a) Criteria as listed at (f)3 below as formula

(b) Criteria as listed at (f)4 below as formula

(d) Criterion is expressed as a function of the Water Effect Ratio (WER). For criterion in the table, WER equates to the default value of 1.0.

(c) Criteria expressed as free cyanide (as CN)/L

(h) Human health noncarcinogen

(hc) Human health carcinogen

(s) Dissolved criterion

(T) Total recoverable criterion

1. Lowest Effects Levels (LELs) indicate concentrations at which adverse benthic impact may begin to occur (level tolerated by most benthic organisms). Water column species and wildlife are at potential risk via bio-magnification (food chain toxicity) if site-related sediment concentrations of PCBs, organochlorine pesticides, or mercury are at or above the LEL. Other known biomagnifiers without ESC warrant case-by-case evaluation.

2. Severe Effects Levels (SELs) are also provided, but the SEL is not a BEE screening value. Contamination at this level indicates severe impacts to the benthic community in most cases studied. For non-polar organics (PAHs, organochlorine pesticides, PCBs), the SEL is calculated from a site-specific TOC level. Since the table SEL is based on 100% organic carbon, the calculated site-specific number is lower.

3. Refer to Estuarine/Marine Screening Criteria when a freshwater parameter has no corresponding value. Since the biological activity of non-polar organics is not expected to differ greatly in the estuarine/marine environment, these screens can be used as surrogates. While uncertainty associated with the use of estuarine/marine metal screens as freshwater surrogates is greater than with non-polar organics, one surrogate metal (silver) is provided.

4. Effects Range-Low (ER-L) represents a concentration at which adverse benthic impacts are found in approximately 10% of studies. Water column species and wildlife are at potential risk via biomagnification (food chain toxicity) if site-related sediment concentrations of PCBs, organochlorine pesticides, or mercury are at or above the ER-L. Other known biomagnifiers without NOAA screening numbers (dioxins, furans, other chlorinated organics, and selenium) warrant case-by-case evaluation.

5. The Effects Range-Median (ER-M) is also provided. The ER-M is not a BEE screening value. Contamination greater than the ER-M value indicates adverse benthic impacts in more than 50% of cases studied.

6. Refer to Freshwater Sediment Screening Criteria when a Estuarine/Marine parameter has no corresponding value and for individual Aroclor values. Since the biological activity of non-polar organics is not expected to differ greatly in the fresh water environment, freshwater screens can be used as surrogates.

7. Screening values were developed for the protection of marine receptors; however, for the purpose of this document they are considered surrogates for freshwater systems.

8. USEPA Region 5, RCRA Ecological Screening Levels (ESLs) represent a protective benchmark (e.g., water quality criteria, sediment quality guidelines/ criteria, and chronic no adverse effect levels) for 223 contaminants and are not intended to serve as cleanup levels, but are intended to function as screening levels. <http://www.epa.gov/reg5rcra/ca/ESL.pdf>

9. Wildlife Preliminary Remediation Goal based on plant study.

10. Wildlife Preliminary Remediation Goal based on shrew study.

11. Wildlife Preliminary Remediation Goal based on woodcock study.

12. Wildlife Preliminary Remediation Goal based on earthworm study.

13. Wildlife Preliminary Remediation Goal based on mouse study.

14. Wildlife Preliminary Remediation Goal based on hawk study.

15. Sediment value from NOAA Screening Quick Reference Tables (SQuiRTs).

16. Westhollow Technical Center Levels were developed by Shell Oil for surface water and were approved for use by NJDEP with the following conditions: 1) the source area is removed, 2) these levels are on the fringe of the contamination area, and 3) active remediation is occurring. These levels are applicable to surface water and wetland areas.

17. USEPA Ambient Water Quality Criteria Update for Methyl Tertiary-Butyl Ether (MTBE) <http://www.epa.gov/waterscience/criteria/mtbe-fs.html>

18. Low Molecular Weight PAHs are defined as compounds composed of fewer than four rings.

19. High Molecular Weight PAHs are defined as compounds composed of four or more rings.

20. Guidance for Developing Ecological Soil Screening Levels, OSWER Directive 9285.7-55, U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, 1200 Pennsylvania Avenue, N.W., Washington, DC 20460, November 2003, Revised February 2005.

http://www.epa.gov/ecotox/ecossl/pdf/ecossl_guidance_chapters.pdf

21. Value applies to DDT and metabolites.

NOTE: See Page 777 (SW Calculations tab) for Surface Water Calculator for metals.

Ecological Screening Process

Note: The CSSAB Risk Assessment Subcommittee asked that a Department contractor (Ogden Environmental Services) assist in developing the rationale for the ecological screen. This Attachment is based upon the work done by Ogden and presented in their report, but has been modified as necessary to conform to changes in the screening process made after the report was written.

Ecological Health Evaluation - Screening Procedure for Sites in Pennsylvania

Introduction

To ensure that any substantial present or probable future risk to the environment is eliminated, both human health and ecological risk evaluations are necessary. The objective of the proposed procedure is to quickly evaluate whether surface soils or sediments at a site have the potential to pose substantial ecological impact. The site screening procedure defines substantial impact as the potential for constituents detected on-site to cause a greater than 20% change in abundance of species of concern compared to an appropriate reference area, or a greater than 50% change in the composition or diversity of a habitat of concern compared to an appropriate reference area (Suter, 1993; Suter et al., 1995; U.S. EPA, 1989). The goal of the screening procedure is to minimize, to the extent practicable, the number of sites which require detailed ecological risk assessment, while remaining protective of the environment. This goal can be accomplished by recognizing:

- the sources of regulated substances in the environment will be controlled;
- natural physical and chemical attenuation mechanisms act on the released regulated compounds, resulting in degradation or sequestration and consequent reduced bioavailability of remaining chemical residuals;
- that at many sites, risks to unmanaged habitats (*e.g.*, areas that are not landscaped) are likely to be low because of human activity/use (such as residential, commercial or industrial), which may preclude the existence of unmanaged habitats;
- the substantial acclimation capacity of natural populations¹ to exposure to low or moderate concentrations of chemical residuals;
- that most remedial actions cause substantial injury to areas of concern beyond the toxicological impacts, as well as impacts to previously unimpacted areas along the perimeter of the remediation area; and,
- that natural systems are self-organizing, and attempts to manage these processes to produce a particular result require long-term management, and even then can result in less than desirable results.

The site ecological screening procedure has been divided into eight discrete steps, as discussed below and illustrated in Figure 1, Ecological Screening Flowchart. Upon completion of this screening procedure, a site will be identified as either:

- not the source of substantial ecological impact and therefore not requiring further ecological evaluation, or
- having the potential to cause a substantial ecological impact and therefore requiring further ecological evaluation.

The key elements of the screening procedure include the following: the presence of light petroleum-product constituents only; the size of the site; the presence or absence of Constituents of Potential Ecological Concern (CPECs) on the site; the presence or absence of species of concern or habitats of concern; and, the presence or absence of completed exposure pathways, taking into account the current or planned future use of the site.

¹Population is defined as an aggregate of individuals of a species within a specified location in space and time (U.S. EPA, 1994a).

Regardless of the outcome of the ecological screening, the results are documented in a written report. It is important to note that if the impacted area of surface soil is equal to or greater than 2 acres, or if the impacted area of sediments is greater than or equal to 1000 square feet, completion of the site ecological screening process requires a site walk. Using a streamlined set of guidelines, this site walk is a critical component of the means of identifying those sites that may pose substantial ecological impacts and, of documenting the lack of ecological impacts at other sites. Without such a site visit, a weight of evidence-based evaluation cannot be achieved, as required by EPA guidance (e.g., EPA's *Framework for Ecological Risk Assessment*; 1992) and ASTM standards (ASTM Designation: E1706-95). In addition, this screening procedure is consistent with the initial steps of EPA's ecological risk assessment guidelines for contaminated sites (U.S. EPA, 1994a). The remainder of this paper discusses each of the eight steps of the ecological screening procedure in more detail.

Step 1: Presence of Light Petroleum-Product Constituents

The first step in the site ecological screening process is to determine whether gasoline, jet fuel A, kerosene, or #2 fuel oil/diesel fuel, which have relatively low PAH content (ASTM Designation: E1739-95), are present. If light petroleum-product constituents (including BTEX) are the only constituents detected on-site, then the screening process moves to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report. If constituents in addition to, or other than, light petroleum-product constituents are present the screening process continues to Step 2 (Site Size).

The purpose of this step is to eliminate from further evaluation those sites at which the only detected constituents are residual compounds from a release of light petroleum products. In general, remediation of light petroleum-product release sites to prevent substantial ecological impacts is not probable based on:

- their rapid attenuation (through multiple fate and transport mechanisms) in surface soils and sediments such that prolonged exposure of species of concern to elevated concentrations is unlikely;
- the likelihood that potential human exposures and risks (through consumption of groundwater (BTEX) and ingestion of soil (PAH)) are greater than potential ecological impacts and, as such, remediation at such sites would be driven by protection of human health; and,
- elevated concentrations of petroleum constituents will be remediated for protection of human health.

Step 2: Site Size

The second step in the site ecological screening process is a comparison of the area of the site² to pre-specified minimum areas of exposed and contaminated surface soil (excluding areas covered by pavement, buildings, or other structures) and sediments that are of potential ecological concern. The minimum areas are: greater than or equal to 2 acres of exposed and contaminated surface soil, and greater than or equal to 1000 square feet of contaminated sediment.

If a site exceeds these specified minimum areas, then the screening process continues to Step 3 (Obvious pathway elimination). If the area of the site is smaller than the specified

² The site is defined as the extent of contamination originating within the property boundaries and all areas in close proximity (Act 2 of 1995, the Land Recycling and Environmental Remediation Standards Act).

minimum areas, then the screening process moves to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report.

For ecological impact to be considered substantial, the minimum size thresholds must be exceeded. Considerations in setting the 2 acre and 1,000 ft² size thresholds were:

- sources of regulated substances will be removed and natural attenuation/acclimation processes in relatively small areas will mitigate impacts naturally to the point that they cannot be regarded as substantial;
- compliance with waste management regulations and human health standards would require remediation of elevated levels of constituents that pose a risk to human health. Such compliance would also protect ecological health to a certain degree;
- regardless of size, the requirements of the Clean Streams Law and the Endangered Species Act must be met; and,
- the smaller the area of contamination, the greater the proportionate risk associated with remedial action posed to adjacent areas unimpacted by chemical residuals, such that the environmental injury caused by remedial actions will be greater than the impacts attributable to the residual regulated substances (U.S. EPA, 1991b).

Incorporation of a minimum size criterion in the ecological screening process is based on the two general types of adverse effects that are of concern when conducting ecological risk assessments: (1) direct toxic effects to the receptors contacting an environmental medium (soil or sediment in this case) or (2) indirect effects (*i.e.*, effects that manifest themselves through the food chain) to higher trophic level receptors that consume organisms living on the site that are directly contacting either surface soils or sediments. Sites smaller than the minimum areas listed above are assumed to not pose a substantial risk through the food chain, even if CPECs are present, because higher trophic level receptors that are likely to be of concern at most sites have feeding ranges substantially greater than the minimum areas used in the site ecological screening process. For example, the red tail hawk or bald eagle has a home range of several hundred to over 2,500 acres (U.S. EPA, 1993a). Consequently, the average concentration of CPECs, if any, in the diet of such species is not assumed to pose a substantial risk. The size threshold for sediment areas of concern is smaller than for surface soils, based on the propensity for constituents to concentrate as a result of differential particle size transport and sorting processes, the sedentary nature of the species making up the benthic community, and the generally greater sensitivity of many aquatic species to constituents (*e.g.*, health-based soil screening criteria for a small mammal, calculated in accordance with standard EPA protocol, are generally higher than effects-based ER-Ls for sediment).

“Sediment” is defined as those mineral and organic materials situated beneath an aqueous layer for durations sufficient to permit development of benthic assemblages. Indicators of benthic assemblages would include macroscopic algae, aquatic invertebrates, or aquatic plants. The aqueous layer may be static, as in lakes, ponds, or other water covered surface depressions greater than or equal to 1,000 square feet but necessarily contiguous (excluding permitted open water management units), or flowing, as in rivers and streams located on a site. This recommended definition of sediment is a combination of definitions from two U.S. EPA documents (U.S. EPA, 1993b; U.S. EPA, 1991a), with modification to accommodate recommendations of the CSSAB.

CPECs in soils and sediments at sites smaller than the critical minimum area are assumed not to pose substantial impact to populations of lower trophic level species. Moreover, any

possible localized changes to populations of lower trophic level species are not likely to impact higher trophic level species of concern because these species forage in areas greater than 2 acres.

Step 3: Obvious pathway elimination

The third step in the site ecological screening process is the consideration of exposure pathways. If all current and potential exposure pathways are eliminated, then the screening process proceeds to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report. If all current and potential pathways are not eliminated, the screen moves on to Step 4 (Presence of Constituents of Potential Ecological Concern).

This step in the screen identifies those sites, particularly those in heavily industrialized or developed areas, where complete pathways to ecological receptors do not exist as the result of factors other than the release(s) associated with the property being remediated.

Step 4: Presence of Constituents of Potential Ecological Concern (CPECs)

The fourth step in the site ecological screening process is the determination of whether any of the constituents detected at the site are considered to be constituents of potential ecological concern (CPECs). The Risk Assessment Subcommittee notes that, as with human health risk assessment, all chemicals are potentially toxic to some component of any ecosystem, given certain conditions. However, it is not practical to evaluate in detail the potential ecological threats posed by all regulated chemicals present at any particular site. Therefore, the Risk Assessment Subcommittee has included a step in the site ecological screening process to identify the presence of constituents of potential ecological concern associated with a release under Act 2 that may substantially alter the structure or function of the ecosystem, and determine whether further evaluation is warranted.

In this and the following step, available site information would be reviewed to determine if CPECs are likely to have been released into the environment. If CPECs are not detected at the site, then the screening process continues to Step 5 (Preliminary Onsite Evaluation). If one or more CPECs are detected at the site, then the screening process moves to Step 6 (Detailed Onsite Evaluation).

For the list of CPECs, the Risk Assessment Subcommittee has selected the 67 chemicals for which U.S. EPA has identified Ecotox Thresholds as CPECs (U.S. EPA, 1996). There are several advantages to using this list, including the following:

- the chemicals on the list are those that typically account for most of the ecological impact at contaminated sites (U.S. EPA, 1994b);
- it would be uncommon for a site to pose substantial ecological impact and not contain some of these chemicals;
- the listed chemicals are those chemicals for which the potential ecological impacts are generally better understood; adding chemicals will increase the frequency of more intensive and academic study, because the necessary toxicity data for quantitative evaluation are not available; and,

- the list contains a reasonable and workable subset of the substances regulated under Act 2.

In addition to this list of 67 constituents, the Risk Assessment Subcommittee has added 4 pesticides (aldrin, chlordane, kepone, and mirex) because of their potential to pose substantial impact to species and habitats of concern, either because of their inherent toxicity or their potential to biomagnify in the food chain. Thus, the final list of CPECs includes 71 constituents, as shown in Table 8 of the regulations.

The ecological evaluation process that has been developed includes additional evaluation criteria for sites where CPECs are not found. Step 5 (Preliminary Onsite Evaluation) is an evaluation of adverse chemical effects that may result from regulated substances other than CPECs and as such, reduces the probability that substantive adverse environmental impacts will go undetected. Also, surface water regulations and standards will remain applicable to those sites, adding to the overall protection of the environment at any site, as will other regulations applicable to ecological receptors, such as the Endangered Species Act.

Step 5: Preliminary Onsite Evaluation

The fifth step of the site ecological screening process is a preliminary onsite evaluation, to be conducted by a qualified environmental scientist (minimum of a bachelor's degree in an environmental science field and 5 years of experience in an environmental field), using written criteria presented in this step. If, after conducting the preliminary site evaluation, the qualified environmental scientist determines that substantial ecological impacts are not probable or evident based on the weight of evidence available for the site, the screening process moves to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report. If after conducting the preliminary site evaluation, the qualified environmental scientist determines that substantial ecological impacts are or may be present, the screening process continues to Step 6 (Formal Onsite Evaluation).

The objective of the ecological evaluation conducted during the preliminary site evaluation is to ensure that substantial ecological impacts resulting from non-CPECs are detected. The preliminary site evaluation involves three steps:

- Review of readily available site background information including: operational history; chemicals used, and probable sources of releases of regulated substances; and, environmental setting with emphasis on physical, chemical and biological factors that would influence the nature and extent of contamination.
- A preliminary site visit to identify physical and habitat features of the area and to identify nearby reference areas³ (if available) that are outside of the probable site (area of contamination associated with a particular release) The following should be noted during the site visit:
 1. signs of stressed or dead vegetation (*e.g.*, chlorotic vegetation);
 2. discolored soil, sediment or water (*i.e.*, a sheen);

³Reference area defined as an area not contaminated by regulated substances originating on the site and used for comparison to the site (U.S. EPA, 1994a). In addition, a reference area should be near the site and have similar geochemical, physical, and biological conditions, but be uncontaminated with regulated substances from the subject site (*i.e.*, unimpacted by the site).

3. presence of non-native materials in sediments resulting from seeps or other discharges emanating from the subject site; and,
 4. presence of deformed organisms (if encountered).
- Preparation of a brief written summary of findings including sketches of the suspected area of contamination and reference areas. To the extent practicable, differences of greater than 50% in the density of species of concern or in the diversity and extent of habitats of concern shall be regarded as potentially substantive (Suter, et al., 1995; U.S. EPA, 1989). However, any differences in the abundance of endangered and threatened species would trigger further evaluation, in addition to requirements under the Endangered Species Act.

Based on all of the information collected as part of the preliminary site evaluation, the investigator makes a determination as to whether substantial ecological impacts exist or are probable even though CPECs were not detected on the site. The conclusion, which documents the weight of evidence from the site evaluation, is summarized in bulleted format.

The Risk Assessment Subcommittee recognizes there are limitations in the ability of such a preliminary site evaluation to detect impacts which result in sub-lethal effects or subtle changes in species density or diversity from non-CPEC constituents. However, the Risk Assessment Subcommittee also recognizes that most ecological restoration efforts result in considerable ecological injury in the area of contamination and surrounding areas which are unimpacted by chemical effects, but which would be injured by restoration activities in the adjacent contaminated area. Consequently, the Risk Assessment Subcommittee is confident that, as long as sources of environmental releases are controlled, the completion of the CPEC screening and preliminary onsite evaluation will identify those situations where adverse toxicological effects to ecological receptors from exposure to regulated substances would result in substantial ecological impacts. The subsequent steps of the screening process determine if the ecological evaluation process should go forward. The environmental effects from less substantive releases will naturally attenuate or be remediated for protection of human health, consistent with the no further action alternative, which would most probably be selected if a more detailed evaluation were to be conducted.

Step 6: Detailed Onsite Evaluation

The sixth step in the site ecological screening process is a detailed onsite evaluation and a determination of whether species or habitats of concern exist on the site or, for endangered and threatened species, if those species exist on the site or within a 2,500-foot radius of the site in its current or intended use. If, during the detailed onsite evaluation, no species or habitats of concern are identified on the site and no threatened and endangered species exist within a 2,500 ft. radius of the site, the screening process moves to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report. If species or habitats of concern are identified on the site, the screening process continues to Step 7 (Identification of Completed Exposure Pathways).

Identification of species and habitats of concern requires a detailed onsite evaluation by a certified ecologist or a trained environmental biologist or ecologist. At a minimum, the

person conducting the detailed onsite evaluation must be a certified ecologist or hold a college degree in ecology or environmental science and at least 5 years of experience conducting ecological field work and risk assessments.

The objective of the detailed onsite evaluation is to identify species or habitats of concern and to make observations that will permit a determination of whether complete exposure pathways are present at the site, as required by Step 7 of the site ecological screening process. The detailed onsite evaluation is to be conducted by a qualified ecologist, as described above, and has the following components:

- Review of readily available site background information including:
 1. operational history, chemicals used, and probable sources of releases of CPECs;
 2. environmental setting with emphasis on physical, chemical and biological factors that would influence the nature and extent of contamination; and,
 3. readily available literature and other relevant documents related to recognition of species and habitats of concern, including endangered and threatened species. (·)The qualified investigator shall conduct the following evaluation:
 4. complete a preliminary walk of the site to identify physical and habitat features of the area; then identify nearby reference areas (if available) which are outside of the probable site (area of contamination associated with a particular site);
- qualitatively evaluate whether species or habitats of concern are present at the site and in the reference area; and,
- in comparison to reference areas (if available), the qualified investigator shall evaluate the following to the extent that they can be readily evaluated at a site:
 1. signs of stressed or dead vegetation (*e.g.*, chlorotic vegetation);
 2. discolored soil, sediment or water;
 3. presence of non-native materials in sediments resulting from seeps or other discharges emanating from the subject property;
 4. community composition differences readily distinguished by U.S. EPA protocols such as the Rapid Bioassessment procedures (U.S. EPA, 1989);
 5. absence of biota (especially keystone species and ecological dominants) compared with similar areas of the same system;
 6. presence of non-native or exotic species compared with reference areas (*e.g.*, *Phragmites*);
 7. presence of deformed organisms (if encountered); and,
 8. potential for residual contamination to habitats of concern and areas utilized by species of concern.
- A brief written summary of findings including sketches of the suspected area of contamination and reference areas. Differences of greater than 20% in the density of species of concern or greater than 50% in the diversity or the extent of habitats of concern shall be regarded as potentially substantive (Suter, 1993; Suter, et al., 1995; U.S. EPA, 1989). However, any differences in the abundance of endangered and

threatened species would trigger further evaluation, in addition to requirements under the Endangered Species Act.

Species of concern are species that have been designated as of special concern, rare, endangered, threatened or candidate by the Pennsylvania Game Commission, Pennsylvania Fish & Boat Commission, and the Bureau of Forestry, if the species has not also been designated as threatened or endangered by the Federal government. A list of such species is presented in Attachment V.I to this Manual. Note that species on the list may be deleted and new species added; therefore an updated list of endangered, threatened and candidate species should be used for the detailed onsite evaluation. Contact the Pennsylvania Game Commission or Pennsylvania Fish & Boat Commission for the most recent listing;

Habitats of concern are:

- wetlands and wetland transition areas;
- breeding areas for species of concern;
- migratory stopover areas for some species of concern (*e.g.*, migrant shorebirds, raptors or passerines);
- wintering areas for species of concern;
- habitat for State endangered plant and animal species;
- Federal, State, and local parks and wilderness areas;
- areas designated⁴ as wild, scenic, recreational; and,
- areas otherwise designated as critical or of concern by the Pennsylvania Game Commission, Pennsylvania Fish & Boat Commission, or the Department of Conservation and Natural Resources.

Step 7: Existence of Completed Exposure Pathways

The seventh step in the site ecological screening process is a determination of whether a completed exposure pathway from CPECs to species or habitats of concern exists at the site in its current or intended use. The existence of a completed exposure pathway⁵ is determined during the detailed site evaluation, as described above for Step 6. Note that the CPECs in soil beneath a paved parking lot or below the root zone (top two feet) are not accessible to most species and habitats of concern and therefore, this pathway is classified as incomplete. If a completed pathway exists at the site, then the screening process moves to Step 8 (Document Attainment of a Standard). If no complete exposure pathways are identified during the formal site walk, then the screening process continues to Step 9 (No Further Ecological Evaluation Required), and the results are documented in the final report.

⁴as defined by guidance.

⁵Exposure pathway - the course a regulated substance(s) takes from the source area(s) to an exposed organism of a species of concern including absorption or intake into the organism. Each complete exposure pathway must include a source or release from a source, a point of exposure, and an exposure route into the organism. The mere presence of a regulated substance in the proximity of a receptor does not constitute a completed pathway. The receptor of concern must be capable of contacting the regulated substance in such a way that there is high probability that the chemical is absorbed into the organism (ASTM. E1739-95; modified to accommodate provisions of Act 2).

Step 8: Document Attainment of a Standard

The eighth step of the site ecological screening process requires that a report be written describing each of the steps in the site ecological screening process and the findings from each step. The report shall provide:

- some indication of the magnitude of potential adverse ecological effects expected, given current and intended future site use;
- the types of species and habitats that potentially may be affected;
- the possible changes or adverse impacts that might result if natural attenuation were the only mitigation process;
- the results of the formal site evaluation;
- the types of ecological impacts should be characterized as direct or indirect, permanent or reversible, and immediate or delayed; and,
- recommendations based on a weight of evidence evaluation. The recommendations may include but are not limited to: no further action; further evaluation of exposure pathways; exposure reduction/elimination plans; consideration of ecological risk reduction due to human health-based remedial actions; completion of a quantitative ecological risk assessment; collection of additional data and completion of a quantitative ecological risk assessment; and immediate response followed by additional study. In addition, if the adverse effects from remedial actions are likely to have a higher probability of causing substantial environmental injury than no further action, then no further action can be warranted at this stage of the screening process.

Step 9: Final Report: No Further Ecological Evaluation Required

The ninth step of the site ecological screening process requires that a brief report be written documenting the findings of the completed steps of the screening process, and the basis for the conclusion that a substantial ecological impact is unlikely and that further ecological evaluation is not required. The conclusion that substantial ecological impact is unlikely is based on one of the following:

- the presence of light petroleum-related constituents only (findings from Step 1);
- the area of impacted surface soil or sediment is less than the minimum size criterion (findings from Step 2);
- all pathways are obviously eliminated because of site features such as paving (from Step 3);
- no CPECs are present on-site and the preliminary onsite evaluation indicates that substantial ecological impacts have not been overlooked (findings from Steps 4 and 5);
- no species or habitats of concern were identified on the site during the formal onsite evaluation (findings from Step 6); or,
- no completed exposure pathways from CPECs to species or habitats of concern were identified during the formal site evaluation (findings from Step 7).

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**State of Connecticut
Department of Energy and Environmental Protection
Transformation Workgroup 5
Soil and Sediment**

**Overview – Wisconsin
Approach to Sediment Remediation**

Wisconsin

- Requirement to investigate and remediate from Wisconsin statutes and administrative code¹. Investigation and cleanup actions that are required after a discharge of a hazardous substance occurs or is discovered. In general, a person responsible for a discharge is required to report, investigate and clean up the contamination.
 - Under 292 All investigation and remediation is implemented by the Department but a “voluntary party” may implement investigation and remediation so long as:
 - The environmental investigation of the property is conducted that is approved by the department.
 - The environment is restored to the extent practicable with respect to the discharges....
 - The voluntary party obtains a certificate of completion from the department that the environment has been satisfactorily restored to the extent practicable....
 - If the voluntary party owns or controls the property, the voluntary party maintains and monitors the property....
 - The voluntary party does not engage in activities that are inconsistent with the maintenance of the property.
 - The voluntary party has not obtained the certificate of completion by fraud or misrepresentation.....
 - Does not appear to preclude anyone from investigating their own property

- Administrative programs - Chapter NR 347 – Sediment Sampling and Analysis, Monitoring protocol and Disposal Criteria for Dredging Projects²
 - Purpose and policy.
 - Applicability.
 - Definitions.
 - Permits, approvals and reviews required.
 - Preliminary application and analytical requirements.
 - Sampling and analysis.
 - Review procedures and review criteria.
 - Monitoring, reporting and enforcement

¹ <http://docs.legis.wisconsin.gov/statutes/statutes/292.pdf>

² http://docs.legis.wisconsin.gov/code/admin_code/nr/347

**State of Connecticut
Department of Energy and Environmental Protection
Transformation Workgroup 5
Soil and Sediment**

**Overview – Wisconsin
Approach to Sediment Remediation**

- Administrative programs – Chapter 700 – Environmental Protection – Investigation And Remediation ³
 - Makes clear the obligation to investigate sediment
 - "Sediment" means particles in surface waters or wetlands that are derived from the erosion of rock, minerals, soils and biological materials, as well as chemical precipitation from the water column. Sediment particles are transported by, suspended in or deposited by water

- Created *Consensus-Based Sediment Quality Guidelines Recommendations for Use & Application* published as final interim guidance in 2003⁴
 - Contains prescriptive elements
 - Recommend that the consensus-based SQGs developed by MacDonald et al. (2000a) be utilized in appropriate situations by all Department programs for screening sediment quality data to help estimate the likelihood of toxicity
 - Provides "additional considerations" for PAHs, Cyanide, Dioxins and Furans
 - Contemplates the use of "Background" or "Reference Site Concentration Considerations" in evaluation process
 - Considers application zone – i.e. defines the biologically active zone (20 to 40 cm and up to 100 cm) in bed sediment but also considers contaminants at depth can be transported up to biologically active zone
 - Contains prescriptive standards based on MacDonald TECs and PECs – includes derivation of a MEC – midpoint effect concentration

³ <http://docs.legis.wisconsin.gov/code/toc/nr>

⁴ http://dnr.wi.gov/topic/Brownfields/documents/cbsqg_interim_final.pdf



Consensus-Based Sediment Quality Guidelines

Recommendations for Use & Application

Interim Guidance

Developed by the
Contaminated Sediment Standing Team

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Guidance Status

This guidance will be updated as needed. Comments and concerns may be sent to "Guidance Revisions" – RR/3, WDNR, P.O. Box 7921, Madison, WI, 53707

Notice

This document is intended solely as guidance and does not contain any mandatory requirements except where requirements found in statute or administrative rule are referenced. This guidance does not establish legal rights or obligations and is not finally determinative of any of the issues addressed. This guidance does not create any rights enforceable by any party in litigation with the State of Wisconsin or the Department of Natural Resources. Any regulatory decisions made by the Department of Natural Resources will be made by applying the governing statutes and administrative rules to the relevant facts.

Consensus - Based Sediment Quality Guidelines; Recommendations for Use & Application

Table of Contents

1. Overview	1
2. Introduction.....	3
3. Recommendations on the Type of Sediment Quality Guidelines to be Used.....	4
4. The Uses of Sediment Quality Guidelines.....	4
5. Considerations and Advantages of Using Consensus-Based Sediment Quality Guidelines.....	6
6. Interpreting Guidelines Concentrations That Fall Between the Lower TEC and Upper PEC Consensus-Based Effect Guideline Values.....	8
7. Recommended Guidelines and Values to be Used in Sediment Quality Assessments	9
8. Additional Considerations for Some Contaminants.....	10
9. Background or Reference Site Concentration Considerations in Using the Effect-Based Sediment Quality Guidelines.....	11
9.1 Metals and Silt/Clay Fraction Relationships.....	13
9.2 Nonpolar Organic Compound and Total Organic Carbon Relationships.....	14
10. Point of Application of the CBSQGs in the Bed Sediment.....	15
11. Other Approaches Being Used to Develop SQGs	16

Tables

Table 1. Recommended Sediment Quality Guideline Values for Metals and Associated Levels of Concern to be Used in Doing Assessments of Sediment Quality.....	17
---	----

Table 2. Recommended Sediment Quality Guideline Values for Polycyclic Aromatic Hydrocarbons (PAHs) and Associated Levels of Concern to be Used in Doing Assessments of Sediment Quality.....	18
--	----

Table 3. Recommended Sediment Quality Guideline Values for Polychlorinated Biphenyls (PCBs) and Chlorinated and Other Pesticides and Associated Levels of Concern to be Used in Doing Assessments of Sediment Quality.....	19
--	----

Table 4. Recommended Sediment Quality Guideline Values for Assorted Contaminants and Associated Levels of Concern to be Used in Doing Assessments of Sediment Quality.....	20
--	----

References	21
------------------	----

Appendixes

Appendix A. Recommended Procedure for Calculating Mean Probable Effect Quotients (PEC Quotients) for Mixtures of Chemicals found at Contaminated Sediment Sites and Their Reliability of Predicting the Presence or Absence of Toxicity.....	25
Appendix B. Recommended Procedure for Calculating the Maximum Probable Background Concentration (MPBC) For a Metal or Organic Compound at Reference or Background Sites.....	29
Appendix C. Notes On Dioxins and Furans.....	32
Appendix D. Calculation Table. Dry Weight Sediment Concentrations of Organic Compounds Normalized to 1% TOC for Comparison with CBSQGs and Grain Size Normalizations of Metals for Site-to-Site Comparisons	33
Appendix E. Identification of Contamination that Leads to Adverse Effects.....	35

Consensus - Based Sediment Quality Guidelines; Recommendations for Use & Application

1. Overview

- Wisconsin DNR needs effects-based (i.e., empirical) sediment quality guidelines (SQGs) for commonly found, in place contaminants to serve as benchmark values for making comparisons to the concentrations of contaminant levels in sediments at sites under evaluation for various reasons (e.g., NR 347 dredging projects, degree and extent studies, screening level ecological risk assessments). There is a need for these values on lower assessment tiers and on a screening level basis and for other objectives during different phases of a site assessment.
- In the last few years, a number of entities have generated effects-based SQGs for some of the more widely measured contaminant metal and organic chemical compounds. Most of the guidelines have focused on effects to benthic-dwelling species. Watershed program staff have used some of the guidelines for evaluating sediment quality at initial or lower tiers in the assessment process for the sediment quality at sites.
- The most recent development in sediment quality guidelines is where the effect-level concentrations from several guidelines of similar narrative intent are combined through averaging to yield consensus-based lower and upper effect values for contaminants of concern (e.g., MacDonald *et al.* 2000a). The consensus-based values have been evaluated for their reliability in predicting toxicity in sediments by using matching sediment chemistry and toxicity data from field studies. The results of the reliability evaluation showed that most of the consensus-based values for individual contaminants provide an accurate basis for predicting the presence or absence of toxicity (MacDonald *et al.* 2000a). To predict the toxicity for mixtures of various contaminants in sediments, the concentration of each contaminant is divided by its corresponding probable effect concentration (PEC). The resulting values are called PEC-Quotients (PEC-Q). The individual PEC-Qs are summed and divided by the number of PEC-Qs to yield a mean PEC-Q. Using relationships derived from existing databases, the mean PEC-Q value can be used to predict the toxicity of a mixture of contaminants in a sediment sample. The appendix provides further explanation and examples of calculating and combining PEC-Q values.
- The CBSQGs as developed only involve effects to benthic macroinvertebrate species. A large amount of databases from toxicological research have established the cause and effect or correlations of sediment contaminants to benthic organism and benthic community assessment endpoints. The guidelines do not consider the potential for bioaccumulation in aquatic organisms and subsequent food chain transfers and effects to humans or wildlife that consume the upper food chain organisms. For the most part where noncarcinogenic or nonbioaccumulative organic chemicals are involved, the guidelines should be protective of human health and wildlife concerns. Where bioaccumulative compounds such as PCBs and methyl mercury are involved, protection of human health or wildlife-based endpoints could result in more restrictive sediment concentrations than contained in the CBSQGs. Where these bioaccumulative compounds are involved, the CBSQGs need to be used in conjunction with other tools, such as human health and ecological risk assessments, bioaccumulation-based guidelines, bioaccumulation studies, and

tissue residue guidelines to evaluate the direct toxicity and upper food chain effects of these compounds. Food chain models will need to be used to estimate safe levels of contaminants in sediments that will not result in accumulated levels in upper food chain organisms that exceed toxicity and tissue reference values.

- There are a number of program needs and uses for sediment quality guidelines during a tiered assessment process for a site under investigation related to further investigative and management decisions. For consistency sake, we recommend that the consensus-based SQGs (CBSQGs) as currently developed by MacDonald et al. (2000a) be utilized in appropriate situations by all Department programs for screening sediment quality data to help estimate the likelihood of toxicity, as staff evaluate the available information in order to make case-by-case investigative and management decisions for a site. For chemicals for which CBSQGs are not available, we recommend utilizing the most reliable of other effects-based freshwater SQGs that have been published in the scientific literature or developed by WDNR or other regulatory entities. In the SQG tables that follow, these latter values are included and identified as to source. In most cases, the guidelines will need to be backed by additional sampling and field studies at sites under investigation to support the guideline-predicted biological effects.
- The MacDonald et al. (2000a) CBSQGs have a lower (threshold effect concentration - TEC) and upper (probable effect concentration - PEC) effect level at which toxicity to benthic-dwelling organisms are predicted to be unlikely and probable, respectively. There is an incremental increase in toxicity as the contaminant concentrations increase between the TEC and PEC concentrations, although specific numerical values relating to the degree of toxicity can't be derived. Based on the ranges of concentration related to the TEC and PEC values, we have developed a qualitative descriptor system to be used to provide a common basis of expressing relative levels of concern with increasing contaminant concentrations. The resulting levels of concern can be used to rank and prioritize sites for additional investigation phases. The midpoint effect concentration (MEC) is a concentration midway between the TEC and PEC concentrations.

Level of Concern	Threshold Effect Concentration (TEC)	Level of Concern	Midpoint Effect Concentration (MEC)	Level of Concern	Probable Effect Concentration (PEC)	Level of Concern
Level 1	From CBSQGs	Level 2	TEC + PEC / 2	Level 3	From CBSQGs	Level 4
≤ TEC		> TEC ≤ MEC	= MEC	> MEC ≤ PEC		> PEC

- Development of sediment quality guidelines is an evolving science. As additional SQGs with applicability to Wisconsin sites and reliability in predicting toxicity are developed, they in turn should be evaluated for possible replacement of the CBSQGs as appropriate. There is a need to continually reexamine the appropriate use of SQGs as management tools and to refine uses of SQGs to better predict toxicity and/or biological community impairment (Fairey et al. 2001). Given the 1) variable environmental and site-specific factors that control the sequestering, release, and bioavailability of contaminants in sediments, 2) the effects of varying mixtures of sediment contaminants, and 3) the variable sensitivities and exposure and uptake routes of benthic macroinvertebrates to contaminants, there is a continued need for guidelines to be supported by site-specific field studies. Along with numerical guidelines, biological criteria based on specific toxicity tests and identified endpoints (e.g., mortality, growth, and reproduction to the test organisms) and benthic community study metrics should be established and used, as

appropriate, in evaluating sediment quality. Levels of acceptable reductions in the endpoints (e.g., no more than 20% reduction [$p < 0.05$] in endpoint response compared to the reference site or control site results in toxicity tests) that can be extrapolated to have ecological relevance for the survival of populations in the field should be established (Lawrence, 1999; Michelsen, 1999; Chapman *et al.* 1997; Suter, 1996; and Suter and Tsao, 1996) and used in the evaluation and management decisions for a contaminated sediment site.

2. Introduction

Over the past several years, different entities including several states, Canadian provinces, U.S. EPA, and various researchers have each developed sets of effects-based SQGs. The guidelines were generally developed using empirical approaches that established databases that related a range of effects (e.g. reduced survival, growth, or reproduction of benthic macroinvertebrate organisms) to a range of increasing concentrations of individual sediment-associated contaminants. The guidelines generally established two concentration levels based on effects - a lower effect level at which no or minimal effects are predicted and an upper effect concentration level at which adverse effects are highly probable or will frequently be seen. The focus for all the sets of guidelines was primarily on developing concentrations that would be protective of the majority of bottom dwelling species that reside on or in the sediments and sediment pore water. The developed guidelines generally do not consider the food chain aspects of such bioaccumulative compounds as methyl mercury and the nonpolar organic compounds (e.g., PCBs) in terms of effects to humans or wildlife.

During the early-1990's, the sediment staff within the Water Quality Standards Section of the Bureau of Watershed Management had initially used effects-based guidelines developed by the province of Ontario in Canada (Persaud *et al.* 1993) and NOAA (1991) in doing screening level assessments of sediment quality for various sediment projects (e.g., NR 347 assessments and in relationship to site investigations conducted at a number of sites). In 1996, based on the studies of contaminated sediments in the Great Lakes, U.S. EPA (Ingersoll *et al.* 1996a, 1996b) produced a set of sediment quality guidelines that Water Program staff incorporated into doing assessments along with the above two sets of guidelines. The Ontario and U.S. EPA guidelines are relevant because they were developed based on databases from studies involving benthic macroinvertebrate species and sites from the Great Lakes region. Since the U.S. EPA guidelines were published, several other sets of guidelines have been developed and published (MacDonald and MacFarlane, 1999 and CCME, 1999).

The most recent development in SQGs is the consensus-based SQGs (CBSQGs) in which the geometric mean of several sets of SQGs of similar narrative intent have been integrated to yield "consensus based" lower (threshold effect concentration - TEC) and upper (probable effect concentration - PEC) effect levels (MacDonald *et al.* 2000a, 2000b ; Swartz, 1999). The CBSQGs of MacDonald *et al.* (2000a) have been adopted for use as sediment quality targets in the St. Louis River Area of concern (Crane *et al.* 2000). Prior to publication of the above consensus-based guidelines in the literature, Water Program staff used the consensus-based approach to develop sediment quality guidelines for a number of metals based on averaging the effect levels from several sets of guidelines. The latter sediment quality objectives are now being superseded by our recommendation that the CBSQGs of MacDonald *et al.* (2000a) be used for all future sediment quality assessments.

3. Recommendations On the Type of Sediment Quality Guidelines To Be Used

For the sake of consistency on a statewide basis in doing initial screenings of sediment quality in the lower tiers of a site assessment and for other uses, it is recommended that:

- 1) The CBSQGs as developed by MacDonald et al. (2000a) for the protection of benthic organisms should be considered for use by all evaluators;
- 2) Reliable effect-based freshwater sediment quality guidelines published in the scientific literature or in Water Quality Standards Section development memos should be used for contaminants for which CBSQGs are not available; and
- 3) Because points 1 and 2 above principally involve protective levels for benthic organisms, other approaches such as food chain modeling and back calculating from acceptable fish tissue levels should be used to establish protective levels of bioaccumulative contaminants in sediments for ecological receptors and humans. Water Quality Standards Section staff tentatively plan to develop a separate technical paper that lists the approaches available and calculation methods of each approach to derive concentrations of contaminants in sediments that would be protective of humans and ecological receptors such as birds and wildlife.

4. The Uses of Sediment Quality Guidelines

As discussed above, there is a need for effects-based sediment SQGs for commonly found contaminants in order to compare to the concentrations that may be in the sediments of a site under study. There is a need for these values on a screening level basis and for other needs during different phases of a site assessment. The uses for CBSQGs include:

- 1) To assess the quality of prospective dredged materials (NR 347 dredging projects) related to potential effects both in place, during removal activities, and at the completion of removal activities. The possible impacts of residual contaminant levels left exposed at the project depth and/or in the side walls at the project boundaries also need to be evaluated.
- 2) To screen study site contaminant concentrations to evaluate the relative degree of potential risks and impacts to sediment dwelling species.
- 3) To identify and to help prioritize sites for additional studies based on the relative degree and extent of contamination, size of contaminated deposits, and potential risks to benthic receptors. These steps can allow for a systematic basis for prioritizing sites for allocation of available funding and resources for further monitoring.
- 4) To evaluate the need to collect additional sediment chemistry data, based on initial screening results, and determine the need to do a concurrent collection of biological data (e.g., toxicity testing and macroinvertebrate community studies) in a second study phase to more adequately characterize the degree and extent of contamination. The biological studies would attempt to validate if the CBSQGs are accurate predictors of toxicity and impacts to the benthic community related to the contaminant concentrations found at a site.
- 5) As toxicity benchmarks in the staged processes associated with screening level ecological risk assessments and the problem formulation stage of baseline ecological risk assessments (Crane *et al.* 2000; Ingersoll *et al.* 1997; U.S. EPA, 1997; WDNR, 1992). Use of the CBSQGs

as benchmarks for toxicity screening serves to 1) estimate the likelihood that a particular ecological risk exists, 2) helps identify the need for site-specific data collection efforts, and 3) helps to focus site-specific baseline ecological risk assessments.

- 6) As one line of evidence where multiple lines of evidence are used to support decision-making activities for a site in a weight-of-evidence approach. No single line of evidence would be used to drive decision-making. Each line of evidence should be evaluated for the 1) adequacy and quality of the data, 2) degree and type of uncertainty associated with the evidence, and 3) relationship of the evidence to the potential degree of impact being estimated. All of the lines of evidence will be integrated to characterize risk based on: 1) concurrence of all line of evidence results 2) preponderance, 3) magnitude, 4) extent, and 5) strength of relationships between the exposure and the effects data.
- 7) The process for assessing sediment quality as it relates to identifying surface water issues will be based on the tiered assessment framework established by the Department's Contaminated Sediment Standing Team (WDNR, 2001). The tiered framework utilizes numerical CBSQGs in the lower tiers and moves to more comprehensive, structured risk-based assessments in the higher tiers. The diversity of different types of sediment assessments and objectives calls for the need for a flexible framework with options for assessing sediment quality. More information is developed in successive tiers until it can be determined that enough information is available to adequately assess the sediment quality related to biological effects. Reasons for conducting risk-based studies at higher assessment tiers may include 1) the complexity of the interactions of the aquatic ecosystem and the contaminant stressors, 2) diverse mixtures of contaminants may be present at a site, 3) outstanding exposure issues where a risk assessment will allow realistic use of information about the natural history of a species such as foraging areas, breeding times, and migration patterns (Moore et al. 1998), and/or 4) there are unresolved issues with regard to potential human or ecological exposures. A formal risk assessment is not something that needs to be conducted at every sediment site under assessment. The appropriate risk-based studies may need to be designed and carried out at higher assessment tiers. As needed, site-specific studies can progress to effects-based testing and risk-based studies of various designs and scope. Guidance for carrying out such risk-based studies are contained in WDNR guidance documents (1992a; 1992b) and a number of U.S. EPA guidance documents (e.g. U.S. EPA, 1998).
- 8) The CBSQGs should not be used on a stand-alone basis to establish cleanup levels or for sediment management decision making. However, in certain situations, with the agreement of all parties involved in overseeing remediation and those responsible for remediating a contaminated sediment site, the CBSQG values deemed to be protective of the site receptors can be used as the remediation objective for a site (at or approaching the lower effect or threshold effect levels for the contaminant of concern). An example of the latter application was at Gruber's Grove Bay on the Wisconsin River, which was contaminated by discharges containing metals from the Badger Army Ammunition Plant. The Army agreed to clean up the sediments based on the greater of the CBSQG TEC for mercury or the background concentration, in lieu of doing any additional biological assessments or studies for the site. Since the background concentration for mercury was found to be greater than the TEC value, background was used as the remediation objective. Using CBSQGs to drive cleanup of some sites may be preferable under certain conditions (based on considerations of size of site and defined boundaries of contamination) rather than spending a large amount of time and

resources for additional studies and risk assessments that may lead to considerable costs with little benefit. At larger, more complex sites, the costs associated with detailed studies may be warranted to reduce uncertainties and focus resources on the remedial actions that provide the greatest benefits (MacDonald et al. 1999).

- 9) It should be noted that there may be contaminated sediment sites and situations where a numerical chemical concentration related to effects may not be the primary driver in a sediment cleanup. Based on a number of balancing factors (e.g., technical feasibility of remediation methods, considerations of natural attenuation factors specific to the site, remedial implementability, human health and ecological risks, stakeholder input, and costs) performance-based standards based on the removal of an established mass of contaminant or removal of visual contamination (applicable to coal tars and petroleum oils) from a site may be the remediation action objective rather than a numerical concentration. There may be situations where the above balancing factors will also be considered to derive a factored cleanup concentration that will not initially achieve the science-based protective sediment concentration but may after an established time period (e.g., when factors such as natural attenuation are considered).

5. Considerations and Advantages of Using Consensus-Based Sediment Quality Guidelines

Given the number of guidelines available, selection of any one as the most appropriate and most reliable for ability to predict toxicity and impacts to benthic species at a study site is difficult. Each guideline set was generally developed using a different methodology (e.g. Ontario [Persaud *et al.* 1993] used the screening level concentration approach and Ingersoll *et al.*[1996a] used the effect level approach). Each approach for developing guidelines has inherent advantages, limitations, levels of acceptance, different extent of field validation, and differing degree of environmental applicability (EPA, 1992). Selecting one set of guidelines is further complicated by uncertainties regarding the bioavailability of contaminants in sediments, the effects of co-varying chemicals and chemical mixtures, the ecological relevance of the guidelines, and correlative versus causal relations between chemistry and biological effects (MacDonald *et al.* 2000a). Given these problems, much discussion has taken place over the use of guidelines as a tool for use in doing sediment quality assessments (Peddicord *et al.* 1998). Cautions are often placed on the use of any one set of guidelines as stand alone decision tools in the assessment and remediation decision making process without additional supporting data from toxicity testing and in-field studies. However, recent evaluations based on combining several sets of guidelines into one to yield "consensus-based" guidelines have shown that such guidelines can substantially increase the reliability, predictive ability, and level of confidence in using and applying the guidelines (Crane *et al.* 2000; MacDonald *et al.* 2000 a, 2000 b; Ingersoll *et al.* 2000). The agreement of guidelines derived from a variety of theoretical and empirical approaches helps to establish the validity of the consensus-based values. Use of values from multiple guidelines that are similar for a contaminant provides a weight-of-evidence for relating to actual biological effects.

A series of papers were produced (Swartz, 1999; Macdonald *et al.* 2000a, 2000b;) that addressed some of the difficulties associated with the assessment of sediment quality conditions using various numerical sediment quality guidelines. The results of these investigations demonstrated that combining and integrating the effect levels from several sets of guidelines to result in consensus-based sediment quality guidelines provide a unifying synthesis of the existing guidelines, reflect causal rather than correlative effects, and can account for the effects of contaminant mixtures in

sediment (Swartz, 1999). Additionally, MacDonald *et al.* (2000a) have evaluated the consensus-based effect levels for reliability in predicting toxicity in sediments by using matching sediment chemistry and toxicity data from field studies conducted throughout the United States. The results of their evaluation showed that most of the consensus-based threshold effect concentrations (TEC - lower effect level) and probable effect concentrations (PEC - upper effect level) for individual contaminants provide an accurate basis for predicting the absence or presence, respectively, of sediment toxicity.

Ingersoll *et al.* (2000, 2001), MacDonald *et al.* (2000a), and Fairey *et al.* (2001) evaluated the reliability of using mean quotient concentration-related values to predict the toxicity in sediments of a mixture of different contaminants. For example, mean PEC quotients were calculated to evaluate the combined effects of multiple contaminants in sediments (Ingersoll *et al.* 2000, 2001; MacDonald *et al.* 2000a). A PEC quotient is calculated for each contaminant in each sample by dividing the concentration of a contaminant in sediment by the PEC concentration for that chemical. A mean quotient was calculated for each sample by summing the individual quotient for each contaminant and then dividing this sum by the number of PECs evaluated. Dividing by the number of PEC quotients normalizes the value to provide comparable indices of contamination among samples for which different numbers of contaminants were analyzed. Results of the evaluation showed that the mean PEC quotients that represent mixtures of contaminants were highly correlated to the incidences of toxicity in the same sediments. See Appendix A for calculation methods and ranges of PEC quotient values that are potentially associated with toxicity.

Based on MacDonald *et al.* (2000a), the consensus-based SQGs can be used for or considered for the following:

- To provide a reliable basis for assessing sediment quality conditions in freshwater ecosystems.
- To identify hot spots with respect to sediment contamination.
- To determine the potential for and spatial extent of injury to sediment-dwelling organisms.
- To evaluate the need for sediment remediation.
- To support the development of monitoring programs to further assess the extent of contamination and the effects of contaminated sediment on sediment-dwelling organisms.

The above applications are strengthened when the consensus-based values are used in combination with other sediment quality assessment tools including effects-based testing (i.e., sediment toxicity tests, bioaccumulation assessments, benthic invertebrate community assessments, and more comprehensive designed risk-based studies).

The consensus-based SQGs as developed only involve effects to benthic macroinvertebrate species. The guidelines do not consider the potential for bioaccumulation in aquatic organisms and subsequent food chain transfers to humans or wildlife. Where bioaccumulative compounds are involved, the consensus-based SQGs need to be used in conjunction with other tools, such as bioaccumulation-based guidelines, bioaccumulation studies, food chain modeling, and tissue residue guidelines to evaluate the direct toxicity and upper food chain effects of these compounds.

The MacDonald *et al.* (2000a) consensus-based sediment quality guidelines have been adopted by the Minnesota Pollution Control Agency (Crane *et al.* 2000) for use as sediment quality targets in the St. Louis River Area of Concern (AOC) on Lake Superior. Following the recommendation in this guidance for the use of the MacDonald *et al.* (2000a) consensus-based SQGs, which would involve their use on the Wisconsin side of the AOC, would be somewhat consistent with their planned use by Minnesota for making assessment and management decisions for contaminated sediment sites on the Duluth side of the AOC.

6. Interpreting Sediment Concentrations That Fall Between the Lower TEC and Upper PEC Consensus-Based Effect Guideline Concentrations

The greatest certainty in predicting the absence or presence of sediment toxicity occurs at sediment contaminant concentrations that are lower than the TEC or greater than the PEC values, respectively. The development of consensus-based SQGs does not include determining the predictability of toxicity related to specific contaminant concentrations in the gradient between the TEC and PEC values. Generally, a consensus-based value for a contaminant cannot be set within the range between the TEC and PEC that would have a low frequency of both false negatives and false positives (Swartz, 1999). Toxicity does occur at contaminant concentrations between the TEC and PEC values with the amount of toxicity dependent on the particular contaminant and with the incidence of toxicity greater than that which occurs at the TEC concentration but less than that which occurs at the PEC concentration (MacDonald *et al.* 2000a). The TEC and PEC concentrations in the consensus-based SQGs define three ranges of concentrations for each contaminant (i.e. < TEC ; > TEC but < PEC ; and > PEC). In assessing the degree of concordance that exists between the chemical concentrations in the three ranges and the incidence of toxicity, it has been demonstrated that for most reliable consensus-based SQG contaminants, there is a consistent and incremental increase in the incidence of toxicity to sediment-dwelling organisms with increasing chemical concentrations (MacDonald *et al.* 2000a, 2000b).

The databases for some individual sets of guidelines, such as the Ontario guidelines (Persaud *et al.* 1993) that have been combined with other guidelines to produce the consensus-based SQGs can be interpolated to yield predictions of the percent of benthic species that may be affected at specific concentrations between the lower and upper effect levels. A somewhat conservative but still realistic interpretation that can be applied to contaminant concentrations that fall in the gradient of concentrations between the consensus-based TEC and PEC concentrations is that as the concentrations of a contaminant increase, toxicity and effects to benthic macroinvertebrate species related to reductions in survival, reproduction, and growth, bioaccumulation, and benthic community alterations correspondingly increase and/or are increasingly more probable. An identified limitation of this relationship is that the threshold and nature of this trend can be controlled by factors in specific sediments due to their characteristics (Peddicord *et al.* 1998). Site specific effects-based testing can be performed to determine the reliability of the prediction of adverse effects based on the use of the CBSQGs on the lower tiers of the assessment.

It is recommended that for the purposes of interpreting the potential impacts of concentrations of contaminants between the TEC and PEC values of the CBSQGs or other guidelines, that a midpoint effect concentration (MEC) be derived and qualitative descriptors be applied to the four possible ranges of concentration that will be created. The qualitative descriptors would be termed "Concern Levels" and would be used as a relative gauge of the potential impacts to the benthic species at that level of contaminant and could be used to prioritize sites for additional studies. A prioritization scheme

for ranking sites will, in most cases, depend on professional judgment of staff given the fact that sampling data for sites will generally be variable for the number of samples and the number of parameters analyzed for. The descriptive “Concern Level” scheme is shown in the following table for arsenic concentrations and is applied below in Tables 1 – 4 of the CBSQGs for the various grouped contaminants.

Level of Concern	Threshold Effect Concentration (TEC)	Level of Concern	Midpoint Effect Concentration (MEC)	Level of Concern	Probable Effect Concentration (PEC)	Level of Concern
Level 1	CBSQG Value	Level 2	TEC + PEC / 2 = MEC	Level 3	CBSQG Value	Level 4
≤ TEC		> TEC ≤ MEC		> MEC ≤ PEC		> PEC
Example For CBSQG Values for Arsenic (mg/kg)						
≤ 9.8	9.8	> 9.8 ≤ 21.4	21.4	> 21.4 ≤ 33	33	> 33

7. Recommended Guidelines and Values to be Used in Sediment Quality Assessments

The consensus-based SQG parameters and related effect concentrations in the tables below are from MacDonald *et al.* (2000a) and are indicated in the source column as CBSQGs. Effect-based sediment quality guideline values for some contaminants from other published sources for which CBSQGs were not available are also included in the following tables and identified as such in the source column. These values also represent useful tools for assessing sediment quality. However, their ability to predict toxicity and reliability may not be as great as that for the CBSQGs for a number of reasons including incomplete validation from field testing. This uncertainty has to be weighed in using the values in the assessment process. In cases where more than one set of guidelines have effect-based concentrations for contaminants for which CBSQGs are not available, the effect-based values from that set of guidelines that were the lowest were generally used in the guideline tables that follow. The narrative terminology for effect levels for the latter guidelines may be different from the TEC and PEC terminology from the CBSQGs but the narrative intent is generally the same in establishing a lower and a higher effect level. Also, the emphasis is on those guidelines developed from studies done in freshwater rather than marine or estuarine habitats.

The individual sets of guidelines that were combined and integrated by MacDonald *et al.* (2000a) to yield the CBSQGs are as follows:

Type of SQG	Acronym	Approach	Reference
Derivation of Threshold Effect Concentration (TEC) CBSQG by MacDonald et al. (2000a) from the following			
Lowest Effect Level	LEL	Screening Level Concentration Approach	Persaud <i>et al.</i> 1993
Threshold Effect Level	TEL	Effect Level Approach	Smith <i>et al.</i> 1996.
Effect Range - Low	ERL	Effect Level Approach	Long and Morgan, 1991
Threshold Effect Level for <i>Hyalella azteca</i> in 28-day tests	TEL-HA28	Effect Level Approach	Ingersoll <i>et al.</i> 1996a and 1996b
Minimal Effect Threshold	MET	Screening Level Concentration Approach	EC and MENVIQ, 1992
Chronic Equilibrium Partitioning Threshold	SQAL (Sediment Quality Advisory Level)	Equilibrium Partitioning Approach	Bolton <i>et al.</i> (1985); Zarba, (1992); U.S. EPA, 1997
Derivation of Probable Effect Concentration (PEC) CBSQG by MacDonald et al. (2000a) from the following			
Severe Effect level	SEL	Screening Level Concentration Approach	Persaud <i>et al.</i> 1993
Probable Effect level	PEL	Effect Level Approach	Smith <i>et al.</i> 1996.
Effect Range - Median	ERM	Effect Level Approach	Long and Morgan, 1991
Probable Effect Level for <i>Hyalella azteca</i> in 28-day tests	PEL-HA28	Effect Level Approach	Ingersoll <i>et al.</i> 1996a and 1996b
Toxic Effect Threshold	TET	Effect Level Approach	EC and MENVIQ, 1992
Acute Equilibrium Partitioning Threshold	No guideline developed	-----	-----

8. Additional Considerations For Some Contaminants

PAHs

Some sources of the parent or unsubstituted PAHs that are in Table 2, such as creosote, coal tars, and petroleum oils, can have co-occurring compounds such as substituted PAHs and heterocyclic aromatic compounds (carbazoles, indoles, acridines, and quinolines) that can be equally or more toxic and more soluble than the listed parent PAH compounds.

Additionally, photoactivation of certain unsubstituted and substituted PAHs, which enhances their toxicity to aquatic organisms that have bioaccumulated these compounds, has been demonstrated both in the laboratory and in the field. The latter may have implications in certain types of habitats (Ankley et al. 2002).

The possible presence of co-occurring toxic compounds where petroleum oils and coal tars are involved and photoactivation of PAHs at sites may need to be considered or toxicity may be underestimated by looking only at the sediment guidelines for the listed parent PAHs in Table 2.

Dioxins and Furans

Polychlorinated dibenzo dioxins (PCDDs) and Polychlorinated dibenzo furans (PCDFs) are unwanted by products of various chemical manufacturing and combustion processes. They are generally ubiquitous in soils and sediments in urban and rural areas. The potential for greatest levels to be found in environmental media are where chlorinated organic compounds such as certain pesticides and pentachlorophenol were either manufactured or used. Pentachlorophenol use at wood treatment operations (railroad ties, utility poles, or lumber) at some sites in Wisconsin sites has led to dioxin and furan compound contamination in floodplain soils and stream sediments. Another source of PCDDs and PCDFs is from the production of paper products from chlorine-bleached wood pulp.

There are 210 polychlorinated dibenzo-p-dioxins (PCDDs) and dibenzofurans (PCDFs) which are based on the points of attachment or substitution of chlorine atoms on the aromatic rings. Of these, 17 (7 dioxins and 10 furans) which have chlorine substituted in the 2,3,7,8 positions are thought to pose the greatest risks to receptor organisms. In order to account for the differing toxicities of the 17 2,3,7,8-substituted isomers, each has been given a toxic equivalency factor (TEF) related to the most toxic form, 2,3,7,8-TCDD (TEF = 1.0). In terms of risk assessments, those PCDDs and PCDFs not substituted in the 2,3,7,8 positions can be ignored. The summed concentration of the TEF of each 2,3,7,8-substituted isomer times its concentration equals the toxic equivalent concentration to 2,3,7,8-TCDD or TCDD-EQ concentration. Appendix C provides a table to calculate a summed TCDD-EQ concentration based on the TEF value and reported concentration for each of the 17 2,3,7,8-substituted isomers found in sediments and floodplain soils.

Cyanide

Cyanide as measured and reported as total cyanides in sediments can include hydrogen cyanide (HCN), cyanide ion (CN⁻), simple cyanides, and metallo- and organo-cyanide complexes. HCN and CN⁻ are grouped as free cyanides and are the most toxic forms of cyanide and the forms of concern.

Most complexed cyanides are relatively nontoxic and total cyanide determinations are not very useful measures of either water or sediment quality. Factors that affect the release or dissociation of free cyanides from complexed cyanide forms include pH, redox potential, photodecomposition of the complex and release of free cyanide, relative strength of the metallo- and organo-cyanide complexes, and possible presence of bacteria responsible for degradation of ferrocyanide complexes. In sediments, the cyanide in the free form present in the pore water is more relatable to toxicity to benthic organisms than the total cyanide measured in the solid phase. However, given the above factors, it is difficult to predict or model the dissociation and release of the free toxic forms of cyanide to the pore water from the less toxic total cyanide form associated with and normally measured in the solid phase sediments. A general idea of the concentrations of free cyanide in pore water that would be toxic to benthic invertebrates can be drawn from the acute and chronic toxicity criteria for free cyanides in surface waters classified as supporting Warm Water Sport Fish (NR 105, Wis. Admin. Code) which are 45.8 ug/L and 11.47 ug/L, respectively. Free cyanides as HCN, in general, are not very persistent in the environment due to their volatility, have low adsorption to sediment particles, high water solubility, and inability to substantially bioaccumulate. Where any significant levels of total cyanide are detected in sediments, additional analysis may need to be done to also determine what fractions of the total cyanide are in dissociable forms (amenable to chlorination or weak acid dissociable forms) to give an indication of the potential to release free cyanide with its attendant toxicity..

9. Background or Reference Site Concentration Considerations In Using the Effect-Based SQGs

In designing and collecting sediment samples at any phase of a site assessment, consideration may need to be given to sampling and analyzing for the same potential chemical stressors, biological data, and/or physical data that are being analyzed for within the study site area at a representative background/reference site to be used as benchmarks for comparison purposes. Establishing representative reference sites is critical because if reference sites are not highly similar to the areas under study, misleading or inappropriate conclusions may be drawn when making data comparisons (Apitz *et al.* 2002). The background/reference site selected needs to have all the characteristics of the study site sediments as close as practical, which includes similar particle size fractions, total organic carbon content, depositional attributes, and relative positioning (e.g., water depth and stream cross section) in the water body as the study site location, but needs to be out of the influence of the study site and the factors responsible for contaminating the study site. Contributions of contaminants (see Appendix E for a discussion of contamination/contaminant and relation to adverse effects) at the reference site can come from two sources: 1) natural sources based on the soils and geological features in the watershed, and 2) anthropogenic sources such as urban runoff. The reference site should be relatively unaffected by anthropogenic inputs. In urban areas, sediment sites outside of the factors that may be influencing the study site may themselves be influenced by ubiquitous urban sources. The sediment quality of reference sites should be reflective of the land uses and land cover of the watershed that the study site is in. Alternatively, suitable background values may be derived through sediment profiles by examining concentrations at depth with the assumption that the lowest concentration at depth represents the pre-industrial or pre-development sediment horizon (Persaud *et al.* 1993).

It has to be recognized that in diverse geographical and geological areas, the natural levels of metals and ubiquitous source anthropogenic organic compounds will vary. Given this variation, dependence

should be put on site-specific samples for establishing reference site concentrations rather than depending on data compiled from other unrelated sites. In areas and at sites where the background/reference site concentrations are greater than the CBSQG TEC values, the local background/reference site concentrations should be used as the practical lower limit for doing sediment evaluations and making management decisions for additional sediment assessments.

The particle size fractions (for metals) and total organic carbon (TOC) content (for nonpolar organic compounds) of all samples should be used to normalize concentrations in order to do relevant and appropriate site-to-site comparisons of contaminant concentrations.

TOC can have its origin either from organic matter from natural sources such as plant materials deposited on sediments or anthropogenic inputs to aquatic systems. In the latter case, elevated TOC sources in sediments can be from such sources as residual petroleum oils, coal tars, or creosote. The controlling importance of the amount of natural organic matter as a TOC source for determining the fate and bioavailability of organic chemicals, especially nonpolar or neutral compounds, has been established (U.S. EPA, 1993). A chemically-unique partitioning coefficient (K_{OC}) for a nonpolar organic compound is used to estimate the pore water concentration based on its partitioning from natural TOC in the sediment. The partitioning coefficient for a compound is assumed to be relatively constant and predictable across various types of natural organic matter. The K_{OC} values for organic compounds can be found in chemical reference books. Nonpolar organic compounds associated with residual oils of anthropogenic origin as a partition media will have different partitioning coefficients compared to natural organic matter (Boyd and Sun, 1990 and Sun and Boyd, 1991) due to the quality of organic carbon. The latter situation may need to be addressed when estimating the bioavailability of nonpolar organic compounds where the TOC is predominantly contributed by some sources of anthropogenic origin.

For metals and particle size, comparing the concentrations of a contaminant in a sample dominated by a fine fraction with one dominated by a sand fraction would be inappropriate and would not yield useful information. Metals and anthropogenic organic compounds will tend to sorb and concentrate in or on finer grained sediments and TOC, respectively.

The intensity of sampling for establishing representative background/reference site concentrations of contaminants should increase at upper tiers in the sediment evaluation process. For example, for comparisons done in the lower tiers of an assessment when initially investigating the site, one to three sediment samples from the reference site, either analyzed individually or composited for one analysis may be appropriate. Where the reference site concentration comparisons may play a more important role in evaluation and management decisions for a site at upper tiers of an assessment, the sampling intensity should generally increase, with at least 10 or more samples taken at the reference site and analyzed individually. Data sets with fewer than 10 samples generally provide for poor estimates of mean concentrations (i.e., there is a large difference between the sample mean and the 95% upper confidence limit). In most cases, a maximum probable background concentration (MPBC) should be calculated for the contaminant(s) derived from the upper 95% confidence level of the mean (EPA, 1992b) after consideration of the distribution of the sample concentrations as showing either a normal or log normal distribution (see Appendix B for example calculations).

Sample results for a metal or organic compound of concern at the background/reference site may be reported out as a censored value i.e. less than a detection level based on the analytical method that meets the data quality objectives established for the sampling and analysis. There are various

methods to handle the censored data to derive values that can be used with the uncensored values in the data set to derive a mean and standard deviation to be used in the calculation of a maximum probable background concentration. Analyses of methods to handle censored data show that, in most cases, sophisticated statistical techniques recommended for estimation problems involving censored data are unnecessary or even inappropriate for statistical comparisons where the number of censored data samples in a data set are generally small. In general, the simple substitution methods work best to maintain power and control type I error rate in statistical comparisons (Clarke, 1995). The simple substitution method includes either 1) substitution of the detection limit as the quantified concentration, or 2) substitution of one-half the detection limit as the quantified concentration. Clarke (1995) recommends steps in selecting the substitution method. At its simplest, substitution method 1) above should generally be used where the number of censored data results are less than 40% of the data set, and method 2) where the censored data is greater than 40%.

9.1 Metals and Silt/Clay Fraction Relationships

There is a strong correlation between decreasing grain size and increasing metal concentrations. Sand-sized material, which is typically low in trace metal concentrations, may serve as a diluent of metal-rich finer grained particles. Larger fractions of sand can hide significant trace metal concentrations and dispersion patterns (Horowitz, 1991). Adjusting for particle grain size effects is important for 1) determining natural background levels of trace elements associated with sediments to serve as a baseline for comparison purposes with other sites, 2) for distinguishing and determining the degree of anthropogenic enrichment, 3) for comparing metal data from site-to-site on a standardized basis, and 4) providing a means for tracing the extent of metal transport and dispersion by eliminating the diluent effects of large particle size contributions.

Two methods are used to address grain size effects. One is to separate out the sand, silt, and clay sized particles from a sample by sieving and analyzing the separate fractions. The other method is to assume that the majority of the metals in a sample are associated with the fine fraction (silt + clay) and then mathematically normalize the metal data to this fraction by dividing the bulk concentration by the fine fraction percentage expressed as a decimal fraction to yield mg of a metal / kg of fines. Particle size analysis of a sediment sample is usually reported as percent sand, silt, and clay fractions. An example of normalizing a bulk sediment concentration for a metal to the fine fraction for a sample with 84 mg/kg of lead and 60% fines (40% silt + 20% clay) is $84 \text{ mg Pb/kg} \div 0.60 \text{ kg fines /kg sediment} = 140 \text{ mg lead / kg of fines}$. The assumption may not always hold true that all or most of the metals are associated with the fine fraction. Also, when the fine fraction falls below 50% of the total combined fractions, the mathematical normalization may not represent the true metal concentration in the fines (Horowitz, 1991). The normalization to the fine fractions should at a minimum be done at least qualitatively to compare on a relative basis the fine fraction contents between the sediment samples where the metal concentrations are being compared. Besides grain size, other normalizing factors have been used and include iron, aluminum, and total organic carbon (Daskalakis *et al.* 1995).

It should be noted that for the CBSQGs for the metals, MacDonald *et al.* (2000a) do not indicate what the relative percentage of the mineral particle size fractions (% sand, silt, and clay) were assumed to be associated with the expressed values. TOC may play some role in the chemical form of the metal and thus its release from the sediments and its bioavailability. TOC may serve as a secondary binding phase of metals with acid volatile sulfates (AVS) serving as the primary binding phase. It is difficult to predict or measure the role of TOC as it relates to metals. For this reason, the study site

bulk sediment metal concentrations need to be directly compared with the CBSQG concentrations in Table 1 without any adjustments for TOC or fine fraction content. The process above for adjusting metal concentrations based on the percent fines is an additional assessment tool for comparing the concentrations between the unimpacted reference site and the study site and between study sites on a fine content-normalized basis and does not play a role in SQG application.

Normalizing contaminant concentrations to the mineral fine content or TOC content is not to be done for assessing toxicity under TSCA or determining hazardous waste characteristics under the Toxicity Characteristic Leaching Procedure (TCLP) test. The sample dry weight bulk concentrations as reported by the analytical laboratory are to be used for comparison with the applicable criteria under these regulations.

9.2 Nonpolar Organic Compound and Total Organic Carbon Relationships

In the case of nonpolar organic compounds such as PAHs, PCBs, dioxins/furans, and chlorinated pesticides, the bulk sediment concentrations can be normalized to the TOC content for site-to-site comparison purposes by dividing the dry weight sediment concentration by the percent TOC in the sediment expressed as a decimal fraction. For example the TOC normalized PCB concentration for a sediment concentration of 7 mg/kg with 3.5% TOC is 200 mg PCB / kg TOC (i.e., $7 \text{ mg PCBs/kg} \div 0.035 \text{ kg TOC/kg} = 200 \text{ mg PCB/kg TOC}$). Normalization of nonpolar organic compounds to TOC content is valid only if the TOC content in the sediments is greater than 0.2%. At TOC concentrations less than 0.2%, other factors that influence partitioning to the sediment pore waters (e.g., particle size and sorption to nonorganic mineral fractions) become relatively more important (Di Toro *et al.* 1991).

MacDonald *et al.* (2000a) indicate that some individual sets of guidelines that were used in their consensus-based approach were originally expressed on an organic carbon-normalized basis. They converted the values in these sets of to dry weight-normalized values at 1% organic carbon to be averaged with the other sets of guideline values to yield the CBSQGs. The final MacDonald *et al.* (2000a) CBSQG values are expressed on a dry weight basis without regard to organic carbon content. It should be noted that the consensus-based SQG values in Tables 2, 3, and 4 below are expressed on an assumed dry weight normalized basis at 1% organic carbon. It has been established that the organic carbon content of sediment is an important factor influencing the movement and bioavailability of nonpolar organic compounds (e.g., PAHs, PCBs, and chlorinated pesticides) between the organic carbon content in bulk sediments and the sediment pore water and overlying surface water. Biological responses of benthic organisms to nonionic organic chemical in sediments are different across sediments when the sediment concentrations are expressed on a dry weight basis, but similar when expressed on an organic carbon normalized basis (ug chemical / g organic carbon basis) (U.S. EPA, 2000).

To appropriately compare the CBSQG dry weight-normalized to 1% TOC values with the dry weight concentrations in the study sediments of variable TOC content, the study sediment contaminant concentrations also need to be converted to a dry weight-normalized to 1% TOC basis.

Appendix D provides a spread sheet for calculating dry weight sediment concentrations for nonpolar organic compounds normalized to 1% TOC. The concentrations given are for an example sediment. Appendix D also contains a spreadsheet for calculating the concentrations of metals normalized to the fine fraction in a sediment sample. An Excel spreadsheet is available for doing the calculations.

An example showing the necessity of doing this conversion to a common 1% TOC basis for organic compounds is shown as follows:

- The threshold effect concentration (TEC) for total PAHs (TPAHs) is 1,610 ug/kg at 1% TOC.
- The example site under assessment has a TPAH concentration of 7,300 ug/kg at 5% TOC.
- Comparing the dry weight concentrations between the guideline value and the example site concentration without consideration of the TOC content differences would appear to show that the study site concentrations are greater than the TEC guideline value (7,300 study site vs. 1,610 TEC).
- To convert the study site TPAH concentration to a dry weight concentration normalized to 1%, divide the 7,300 ug/kg value by 5 (5% TOC content) = 1,460 ug TPAH/kg at 1% TOC. On the common basis of 1% TOC, the study site TPAH concentration is less than the TEC concentration (1,460 ug/kg study site vs. 1,610 ug/kg TEC).
- In the case above, another approach for converting the concentrations to a common normalized basis is to multiply the TEC concentration by 5 that is the percent TOC of the study site sample. The common basis here are dry weight-normalized concentrations at 5% TOC (7,300 ug/kg study site vs. 8,050 ug/kg TEC).

10. Point of Application of the CBSQGs in the Bed Sediment

The numerical CBSQGs apply to the biologically active zone associated with deposited sediments in flowing (streams and rivers) and static (lakes and ponds) water bodies and wetland soils and sediments. The biologically active zone is inhabited by infaunal organisms including microbes, meiofauna, and macroinvertebrates and other organisms (e.g., egg and larval stage of fish) that spend all or part of their life cycles associated either within (infaunal) or on (epibenthic) the bottom sediments. The community of organisms present will generally depend on the physical and chemical characteristics of the waterbody and bottom sediments as determined by the watershed location and ecoregion within the State. The depth of the biologically-active zone varies between sites depending on the substrate characteristics present (including particle size fractions, organic matter content, compaction, pore-water geochemistry, and water content) which influence the composition of sediment-associated organisms present. The biologically active zone typically encompasses the top 20 to 40 cm. of sediment in freshwater environments (Clarke *et al.* 2001). The majority of benthic organisms will usually be associated with the upper strata (e.g., 15 cm) related to these depth ranges. Certain invertebrate and/or amphibian species can utilize habitats deeper in bed sediments during a portion of their life history (e.g., down to 100 cm below the sediment surface) (MacDonald *et al.* 2000a). The best available knowledge about the local composition of sediment-associated biota and the bioactive depth zone they occupy should supplement the generic depth assumptions above (Clarke *et al.* 2001) where possible. Contaminants in sediments at depths below the biologically active zone can be of concern because of their potential to move to the upper sediment strata through various mechanisms that include diffusion and being transported on groundwater flows that discharge to the surface water body. The groundwater-sediment-surface water zone is a zone of transitions in which various environmental factors can affect contaminant fate and transport.

The CBSQGs should be considered when assessing contaminated soils and sediments deposited on upper bank areas and floodplain areas that have the potential to be eroded or scoured and transported to and deposited in a nearby surface water body.

11. Other Approaches Being Used to Develop SQGs

U.S. EPA has developed national equilibrium partitioning sediment guidelines (ESGs) for a broad range of sediment types. They have finalized the methodologies for deriving ESGs for nonionic organic chemicals (2000a) and mixtures of certain metals (cadmium, copper, lead, nickel, zinc, and silver (U.S.EPA, 2000b). U.S. EPA is planning to publish final guidance (EPA, 2000c) for developing SQGs based on a combination of the equilibrium partitioning (EqP) approach, quantitative structure activity relationships, narcosis theory, and concentration addition models for mixtures of PAH found at specific sites. The EqP-based summed PAH toxicity model provides a method to address causality, account for bioavailability, consider mixtures, and predict toxicity and ecological effects (U.S. EPA, 2000). The U.S. EPA guidance indicates that the total number of PAHs that need to be considered in SQG development is 34 (18 parent and 16 with alkylated groups). Use of fewer than 34 may greatly underestimate the total toxicological contribution of PAH mixtures. The guidance requires the use of conservative uncertainty factors to be applied when fewer than the 34 are being used to estimate site-specific toxicity of PAH mixtures.

When guidance has been published in final for the use and application of the ESGs for metals, PAH mixtures, and other nonionic organic compounds, the Water Quality Standards section plans to produce additional guidance on the use of the ESGs to be used in addition to or instead of the CBSQGs. U.S. EPA's apparent intent is not to use the ESG numeric values as stand alone criteria for application as part of a States water quality standards under Section 3 (c) of the Clean Water Act, but to use them as a screening tool in conjunction with other assessment tools such as toxicity testing in evaluating and prioritizing sites under various programs (e.g., developing Total Maximum Daily Loads (TMDLs) s and WPDES permit limitations, Superfund, RCRA).

Table 1. Recommended Sediment Quality Guideline Values For Metals and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

Metal	mg/kg dry wt. ⁺⁺							Source of SQG Effect-Based Concentrations
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	
Antimony	↔	2	↔	13.5	↔	25	↔	NOAA (1991) ¹ .
Arsenic	↔	9.8	↔	21.4	↔	33	↔	CBSQG (2000a) ²
Cadmium	↔	0.99	↔	3.0	↔	5.0	↔	CBSQG (2000a)
Chromium	↔	43	↔	76.5	↔	110	↔	CBSQG (2000a)
Copper	↔	32	↔	91	↔	150	↔	CBSQG (2000a)
Iron	↔	20,000	↔	30,000	↔	40,000	↔	Ontario (1993) ³
Lead	↔	36	↔	83	↔	130	↔	CBSQG (2000a)
Manganese	↔	460	↔	780	↔	1,100	↔	Ontario (1993)
Mercury	↔	0.18	↔	0.64	↔	1.1	↔	CBSQG (2000a)
Nickel	↔	23	↔	36	↔	49	↔	CBSQG (2000a)
Silver	↔	1.6	↔	1.9	↔	2.2	↔	BC (1999) ⁴ .
Zinc	↔	120	↔	290	↔	460	↔	CBSQG (2000a)

++ The CBSQGs for organic compounds are expressed on a dry weight concentration at 1% TOC in sediments. However, unlike the organic compounds, the CBSQG and study site metals concentrations can be compared on a bulk chemistry basis and do not need to be adjusted to a 1% TOC basis to do the comparison. TOC does not play the same role in determining metals availability as it does in determining organic compound availability.

1. NOAA (1991) = Long, E.R. and L.G. Morgan. 1991. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration. Seattle, Washington.
2. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20-31.
3. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.
4. MacDonald, D.D. and M. MacFarlane. 1999. (Draft). Criteria for managing contaminated sediment in British Columbia. British Columbia Ministry of Environment, Lands, and Parks. Victoria, British Columbia.

Table 2. Recommended Sediment Quality Guideline Values For Polycyclic Aromatic Hydrocarbons (PAHs) and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

PAH	ug/kg dry wt. at 1% TOC **							Source of SQG Effect-Based Concentrations
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	
Low Molecular Weight PAHs (3 or less benzene rings)								
Acenaphthene	↔	6.7	↔	48	↔	89	⇒	CCME (1999) ¹ .
Acenaphthylene	↔	5.9	↔	67	↔	128	⇒	CCME (1999)
Anthracene	↔	57.2	↔	451	↔	845	⇒	CBSQG (2000a) ² .
Fluorene	↔	77.4	↔	307	↔	536	⇒	CBSQG (2000a)
Naphthalene	↔	176	↔	369	↔	561	⇒	CBSQG (2000a)
2-methylnaphthalene	↔	20.2	↔	111	↔	201	⇒	CCME (1999)
Phenanthrene	↔	204	↔	687	↔	1,170	⇒	CBSQG (2000a)
High Molecular Weight PAHs (4 or more benzene rings)								
Benz(a)anthracene	↔	108	↔	579	↔	1,050	⇒	CBSQG (2000a)
Benzo(a)pyrene	↔	150	↔	800	↔	1,450	⇒	CBSQG (2000a)
Benzo(e)pyrene	↔	150	↔	800	↔	1,450	⇒	Similar as above ³ .
Benzo(b)fluoranthene	↔	240	↔	6,820	↔	13,400	⇒	Similar as below ⁴ .
Benzo(k)fluoranthene	↔	240	↔	6,820	↔	13,400	⇒	Persaud <i>et al.</i> 1993 ⁵
Benzo(g,h,i)perylene	↔	170	↔	1,685	↔	3,200	⇒	Persaud <i>et al.</i> 1993
Chrysene	↔	166	↔	728	↔	1,290	⇒	CBSQG (2000a)
Dibenz(a,h)anthracene	↔	33	↔	84	↔	135	⇒	CBSQG (2000a)
Fluoranthene	↔	423	↔	1,327	↔	2,230	⇒	CBSQG (2000a)
Indeno(1,2,3-cd)pyrene	↔	200	↔	1,700	↔	3,200	⇒	CBSQG (2000a)
Pyrene	↔	195	↔	858	↔	1,520	⇒	CBSQG (2000a)
Total PAHs								
Total PAHs	↔	1,610	↔	12,205	↔	22,800	⇒	CBSQG (2000a)

** To compare the study site concentrations with the Table 2 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. normalized value at 1% TOC. If no site TOC information is available, assume a 1% TOC content.

1. CCME (1999) = Canadian Council of Ministers of the Environment (CCME). 1999. Canadian sediment quality guidelines for the protection of aquatic life: Summary tables. In: Canadian environmental quality guidelines. 1999. Canadian Council of Ministers of the Environment, Winnipeg.
2. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. *Arch. Environ. Contam. Toxicol.* 39:20-31.
3. There are no guideline values for Benzo(e)pyrene. "Similar as above" assumes the similarity of the chemical structure of Benzo(e)pyrene with Benzo(a)pyrene would yield similar quantitative structure activity relationships (QSARs) as it relates to toxicity, therefore the effect level concentrations that were derived for Benzo(a)pyrene would also apply to Benzo(e)pyrene.
4. There are no guideline values for Benzo(b)fluoranthene. "Similar as below" assumes the similarity of the chemical structure of Benzo(b)fluoranthene with Benzo(k)fluoranthene would yield similar quantitative structure activity relationships (QSARs) as it relates to toxicity, therefore the effect level concentrations that were derived for Benzo(k)fluoranthene would also apply to Benzo(b)fluoranthene.
5. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.

Table 3. Recommended Sediment Quality Guideline Values For Polychlorinated Biphenyls (PCBs) And Chlorinated and Other Pesticides and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

PCB and Pesticides	ug/kg dry wt. at 1% TOC ⁺⁺							Source of SQG Effect-Based Concentrations
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	
PCBs								
Total PCBs	↔	60	↔	368	↔	676	⇒	CBSQG (2000a) ¹ .
Pesticides								
Aldrin	↔	2	↔	41	↔	80	⇒	Ontario (1993) ² .
BHC	↔	3	↔	62	↔	120	⇒	Ontario (1993)
α-BHC	↔	6	↔	53	↔	100	⇒	Ontario (1993)
β-BHC	↔	5	↔	108	↔	210	⇒	Ontario (1993)
γ-BHC (lindane)	↔	3	↔	4	↔	5	⇒	CBSQG (2000a)
Chlordane	↔	3.2	↔	10.6	↔	18	⇒	CBSQG (2000a)
Dieldrin	↔	1.9	↔	32	↔	62	⇒	CBSQG (2000a)
Sum DDD	↔	4.9	↔	16.5	↔	28	⇒	CBSQG (2000a)
Sum DDE	↔	3.2	↔	17	↔	31	⇒	CBSQG (2000a)
Sum o,p' + p,p' DDT	↔	4.2	↔	33.6	↔	63	⇒	CBSQG (2000a)
Sum of DDT +DDD + DDE	↔	5.3	↔	289	↔	572	⇒	CBSQG (2000a)
Endrin	↔	2.2	↔	104.6	↔	207	⇒	CBSQG (2000a)
Heptachlor Epoxide	↔	2.5	↔	9.3	↔	16	⇒	CBSQG (2000a)
Mirex	↔	7	↔	10.5	↔	14	⇒	BC (1999) ³ .
Toxaphene	↔	1	↔	1.5	↔	2	⇒	BC (1999)

⁺⁺ To compare the study site concentrations with the Table 3 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. - normalized value at 1% TOC. If no site TOC information is available, assume a 1% TOC content.

1. CBSQG (2000a) = MacDonald, D.D., C.G. Ingersoll, and T.A. Berger. 2000a. Development and evaluation of consensus-based sediment quality guidelines for freshwater ecosystems. Arch. Environ. Contam. Toxicol. 39:20-31.
2. Ontario (1993) = Persaud, D.R., R. Jaagumagi, and A. Hayton. 1993. Guidelines for the protection and management of aquatic sediments in Ontario. Standards Development Branch. Ontario Ministry of Environment and Energy. Toronto, Canada.
3. MacDonald, D.D. and M. MacFarlane. 1999. (Draft). Criteria for managing contaminated sediment in British Columbia. British Columbia Ministry of Environment, Lands, and Parks. Victoria, British Columbia.

Table 4. Recommended Sediment Quality Guideline Values For Assorted Contaminants and Associated Levels of Concern To Be Used In Doing Assessments of Sediment Quality.

Sediment Contaminant	ug/kg dry wt. at 1% TOC ⁺⁺							Source of SQG Effect-Based Concentrations
	Level 1 Concern ≤ TEC	TEC	Level 2 Concern > TEC ≤ MEC	MEC	Level 3 Concern > MEC ≤ PEC	PEC	Level 4 Concern > PEC	
Benzene	↔	57	↔	83.5	↔	110	↔	BC (1999) ^{1.}
Toluene	↔	890	↔	1,345	↔	1,800	↔	BC (1999)
Xylene	↔	25	↔	37.5	↔	50	↔	BC (1999)
2,3,7,8-TCDD (pgTEQ/g)	↔	0.85	↔	11.2	↔	21.5	↔	Canada (2002) ^{2.}
Pentachlorophenol	↔	150	↔	175	↔	200	↔	Janisch (1990) ^{3.}
Tributyltin	↔	0.52	↔	1.73	↔	2.94	↔	Janisch (1994) ^{4.}
1,2-Dichlorobenzene	↔	23	↔	-----	↔	23	↔	Washington (1991) ^{5.}
1,4-Dichlorobenzene	↔	31	↔	60.5	↔	90	↔	Washington (1991)
1,2,4-Trichlorobenzene	↔	8	↔	13	↔	18	↔	Washington (1991)
Dimethyl Phthalate	↔	530	↔	-----	↔	530	↔	Washington (1991)
Diethyl Phthalate	↔	610	↔	855	↔	1,100	↔	Washington (1991)
Di-N-Butyl Phthalate	↔	2,200	↔	9,600	↔	17,000	↔	Washington (1991)
Di-N-Octyl Phthalate	↔	580	↔	22,790	↔	45,000	↔	Washington (1991)
Dibenzofuran	↔	150	↔	365	↔	580	↔	Washington (1991)
Phenol	↔	4,200	↔	8,100	↔	12,000	↔	Washington (1991)
2-Methylphenol	↔	6,700	↔	-----	↔	6,700	↔	Washington (1991)
2,4-Dimethyl Phenol	↔	290	↔	-----	↔	290	↔	Washington (1991)
Benzyl Alcohol	↔	570	↔	650	↔	730	↔	Washington (1991)
Benzoic Acid	↔	6,500	↔	-----	↔	6,500	↔	Washington (1991)

++ To compare the study site concentrations with the Table 4 concentrations on a common basis, divide the study site concentrations by the %TOC at the study site to yield a dry wt. - normalized value at 1% TOC. If no site TOC information is available, assume a 1% TOC content.

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Appendix A

Recommended Procedure for Calculating Mean Probable Effect Quotients (Mean PEC Quotients) for Mixtures of Chemicals found at Contaminated Sediment Sites and Their Reliability of Predicting the Presence or Absence of Toxicity (Adopted from Ingersoll *et al.* 2000, 2001).

- Step 1.** Based on existing databases, the reliability to predict toxicity is greatest for the organic compound groups of total PAHs and total PCBs and the metals arsenic, cadmium, chromium, copper, lead, nickel, and zinc. Inclusion of other compounds or metals that have a PEC value, where there is insufficient data available to evaluate its predictive reliability (e.g., mercury, dieldrin, DDD, DDT, endrin, and lindane) into the overall PEC-Q calculation may result in an overall PEC-Q value with lower predictive ability.
- Step 2.** Calculate the individual PEC Quotients (PEC-Qs) for chemicals with reliable PECs within each of the chemical classes. Since the PECs for PAH and PCB chemical classes are based on total concentrations, individual PEC-Qs for individual compounds in these classes do not need to be calculated.

$$\text{Individual Chemical PEC-Q} = \frac{\text{Chemical concentration in Study Site Sediments (in dry wt.)}}{\text{PEC SQG Concentration for Chemical (in dry wt.)}}$$

For the nonpolar organic compounds (total PCBs and total PAHs), the PEC SQG is expressed on a dry weight basis normalized to 1% organic carbon. The concentration for these groups of nonpolar compounds in the study site sediments also needs to be expressed on this same basis. To do this, divide the concentration in the study site sediments by the percent TOC in the sediments expressed as a whole number (e.g., 7,300 ug/kg PCB at 5% TOC is $7,300 \div 5 = 1,460$ mg/kg dry weight normalized to 1% TOC).

- Step 3.** In the case of metals, a mean $\text{PEC-Q}_{\text{metals}}$ for the metals involved needs to be calculated based on summing the PEC-Q for the individual metals and dividing by the number of metals.

$$\text{Mean PEC-Q}_{\text{metals}} = \frac{\sum \text{individual metal PEC-Qs}}{\text{Number of metals for which individual PEC-Qs calculated}}$$

- Step 4.** Calculate the overall mean PEC-Q for the three main classes of chemicals.

$$\text{Mean PEC-Q}_{\text{overall}} = \frac{(\text{mean PEC-Q}_{\text{metals}} + \text{PEC-Q}_{\text{total PAHs}} + \text{PEC-Q}_{\text{total PCBs}})}{n}$$

Where n = number of classes of chemicals for which sediment chemistry available (e.g., in this case, there are three classes – metals, PAHs and PCBs. In other cases, metals and PAHs may be the only chemicals of concern at a site and therefore PEC-Qs may only be calculated for these two groups and therefore $n = 2$).

Appendix A (continued)

The database used by Ingersoll et al. (2001) to determine the ability of the PEC-Qs to predict toxicity is based on testing freshwater sediments from a number of sites using 10- to 42-day toxicity tests with the amphipod *Hyalella azteca* or the 10- to 14-day toxicity tests with the midges *Chironomus tentans* or *C. riparius*. Toxicity of samples was determined as a significant reduction in survival or growth of the test organisms relative to a control or reference sediment. A relative idea of the predictive ability of the overall mean PEC-Qs and individual PEC-Qs for each group of chemicals is shown in the table below from Ingersoll et al. (2001). Mean PEC quotients were calculated to provide an overall measure of chemical contamination and to support an evaluation of the combined effects of multiple contaminants in sediments.

Test Species and Test Duration	Incidence of Toxicity (% of samples where toxicity observed versus no toxicity) Based on the Mean PEC Quotients (Number of Samples in Parentheses)					Total Number of Samples
	Range of Mean PEC Quotients					
	< 0.1	0.1 to < 0.5	0.5 to < 1.0	1.0 to < 5.0	> 5.0	
<i>Hyalella azteca</i> 10- to 14-day tests						
Mean Overall PEC-Q ^{1.}	19 (79)	26 (89)	38 (34)	49 (35)	86 (29)	266
Q _{metals} ^{2.}	23 (40)	24 (139)	33 (45)	81 (31)	100 (11)	266
PEC-Q _{total PAHs} ^{3.}	25 (123)	33 (76)	35 (20)	49 (33)	100 (14)	266
PEC-Q _{total PCBs} ^{4.}	20 (98)	25 (61)	47 (43)	47 (34)	73 (30)	266
<i>Hyalella azteca</i> 28- to 42-day tests					> 1.0	
Mean Overall PEC-Q	4 (45)	6 (18)	50 (18)	NC ^{5.}	100 (28)	109
PEC-Q _{metals}	5 (40)	25 (24)	60 (33)	NC	100 (12)	109
PEC-Q _{total PAHs}	8 (57)	64 (37)	55 (9)	NC	100 (6)	109
PEC-Q _{total PCBs}	4 (26)	6 (35)	17 (12)	NC	97 (36)	109
<i>Chironomus spp.</i> 10- to 14-day tests					> 5.0	
Mean Overall PEC-Q	29 (21)	35 (78)	35 (26)	50 (34)	78 (18)	177
PEC-Q _{metals}	8 (12)	43 (107)	22 (36)	75 (12)	90 (10)	177
PEC-Q _{total PAHs}	26 (64)	33 (73)	77 (13)	85 (20)	71 (7)	177
PEC-Q _{total PCBs}	48 (58)	23 (31)	34 (32)	35 (34)	68 (22)	177

1. Mean Overall PEC-Q = Based on samples where average metal quotient, total PAH quotient, and PCB quotient summed and divided by 3.

In samples where the metals, total PAHs, and total PCBs were all measured, each of the three PEC-Qs were evaluated individually to determine their predictive ability, yielding the individual PEC-Q values below.

2. PEC-Q_{metals} = Average PEC quotient for the number of metals involved calculated .
3. PEC-Q_{total PAHs} = Based on the samples where individual PAHs measured in samples which were summed to yield a total PAHs value.
4. PEC-Q_{total PCBs} = Based on samples where total PCBs measured in samples.
5. NC = Not calculated.

Appendix A (continued)

Observations from Ingersoll et al. (2001):

- There was an overall increase in the incidence of toxicity with an increase in the mean quotients in toxicity tests involving all three test organisms.
- A consistent increase in the toxicity in all three tests occurred at a mean quotient of > 0.5. However, the overall incidence of toxicity was greater in the *Hyalella azteca* 28-day test compared to shorter term tests. The longer term tests, in which survival and growth are measured, tend to be more sensitive than the shorter term tests, with the acute to chronic ratios on the order of six indicated for *Hyalella azteca*.
- The use of chronic laboratory toxicity tests better identified chemical contamination in sediments compared to many of the commonly used measures of benthic invertebrate community structure. The use of longer-term toxicity tests in combination with SQGs may provide a more sensitive and protective measure of potential toxic effects of sediment contamination on benthic communities compared to use of the 10-day toxicity tests.
- There appears to be different patterns of toxicity when the PEC-Qs for the chemical classes are used alone or combined. The different patterns in toxicity may be the result of unique chemical signals associated with individual contaminants in samples. While the combined mean PEC quotient value from the chemical classes can be used to classify samples as toxic or nontoxic, individual PEC quotients of each chemical class might be useful in helping identify substances that may be causing or substantially contributing to the observed toxicity.
- The results of the evaluation indicate that the consensus-based PECs can be used to reliably predict toxicity of sediments on both a regional and national basis.

Example Calculation

The analytical results for a sediment sample and the steps to derive a mean overall PEC-Q for all the contaminants are as follows:

mg/kg dry wt.									
Sample Bulk Sediment Concentrations									
Metals							Organics		
Arsenic	Cadmium	Copper	Chromium	Lead	Nickel	Zinc	Total PAHs	Total PCBs	TOC
75	9	170	90	270	65	320	108	9.2	2.5%
Since TOC does not play a major role in the partitioning of metals from the sediments to the sediment pore water and its subsequent bioavailability, it is not necessary to convert metals concentrations to a dry weight normalized concentration at 1% TOC. Use the bulk sediment concentration as reported on the lab sheets to compare directly with the PEC SQGs. Normalization of metals concentrations to the fine fraction is done for the purposes of comparing the study site metal concentrations with the reference site concentrations on a common basis and is not related to the SQGs.							Convert the PAH and PCB concentrations dry wt. normalized concentrations at 1% TOC. Divide concentrations by 2.5. Step 2 above.		
75	9	170	90	270	65	320	43.2	3.68	
Determine the PEC concentrations for each contaminant (from Tables 1, 2, and 3 above).									
33	5	150	110	130	49	460	22.8	0.68	
Calculate the PEC-Q for each contaminant. Step 2 above.									
2.27	1.8	1.13	0.82	2.08	1.33	0.70	1.89	5.41	
Calculate a mean PEC-Q for the metals. Step 3 above.									
1.45							1.89	5.41	
Calculate an overall mean PEC-Q value from the 3 chemical classes (metals, PAHs, and PCBs). Step 4 above.									
Mean PEC-Q = 2.92									
Compare the 2.92 value with the ranges of PEC-Q values in the table above. For the shorter-term toxicity tests with <i>Hyalella azteca</i> and <i>Chironomus spp.</i> , a value of 2.92 is in a range where 50% of the samples were toxic. For the longer-term tests with <i>H. azteca</i> , all of the samples were toxic at the PEC-Q value of 2.92. It appears based on these results, <i>H. azteca</i> or benthic organisms of similar sensitivity in the field populations may be significantly impacted by the concentrations of contaminants present. If these results represented an actual site, further assessments of the site is warranted.									

Observations From MacDonald et al. (2000)

MacDonald *et al.* (2000) also looked at the predictive ability of the CBSQGs. To examine the relationships between the degree of chemical contamination and probability of observing toxicity in freshwater sediments, the incidence of toxicity within various ranges of mean PEC quotients was calculated from an existing database. The data were plotted in a graph (Table 1, MacDonald *et al.* 2000). The interpolated data from this graph is in the table below. MacDonald et al. found that subsequent curve-fitting indicated that the mean PEC-quotient is highly correlated with incidence of toxicity ($r^2 = 0.98$), with the relationship being an exponential function. The resulting equation ($Y = 101.48 (1-0.36^X)$) can be used to estimate the probability of observing sediment toxicity at any mean PEC quotient.

Relationship between Mean PEC Quotient and Incidence of Toxicity in Freshwater Sediments (Derived and Interpolated from MacDonald <i>et al.</i> 2000a)	
Mean PEC Quotient	Average Incidence of Toxicity (%)
0	0
0.25	20
0.50	40
0.75	54
1.00	64
1.25	70
1.50	77
1.75	84
2.00	87
2.25	90
2.50	92
2.75	95
3.00	96
3.25	98
3.50	99
3.75	99.5
≥ 4.00	100

Utilizing the mean PEC-Quotient of 2.92 calculated in the example above yields a predicted average incidence of toxicity of approximately 95% based on the table immediately above. The chances are likely that if a sampled site yields a mean PEC-Q of 2.92, significant toxicity to infaunal species will be present.

Appendix B

Recommended Procedure for Calculating the Maximum Probable Background Concentration (MPBC) For a Metal or Organic Compound at Reference or Background Sites

Calculating the 95% upper confidence limit (UCL) of the mean of a data set of background concentrations for a parameter. Use of the UCL as the maximum probable background concentration (MPBC) for comparison purposes with the study site concentrations (Adapted from EPA, 1992b).

Statistical confidence limits are a tool for addressing uncertainties of a distribution average. The 95% UCL of the arithmetic mean concentration is used as the average concentration because it is not possible to know the true mean. The 95% UCL therefore accounts for uncertainties due to limited sampling data. As sample numbers increase, uncertainties decrease as the UCL moves closer to the true mean. Sampling data sets with fewer than 10 samples may provide a poor estimate of the mean concentration (i.e., there is a large difference between the sample mean and the 95% UCL). Data sets with 10 to 20 samples may provide a somewhat better estimate of the mean (i.e., the 95% UCL is close to the sample mean). In general, the UCL approaches the true mean as more samples are included in the calculation.

Transformation of the Data

The data set for the background concentrations should be looked at to determine if the data is lognormally or normally distributed. A statistical test should be used to identify the best distributional assumption for the data set. The W-test (Gilbert, 1987) is one statistical method that can be used to determine if a data set is consistent with a normal or lognormal distribution. In all cases, it is useful to plot the data to better understand the parameter distribution in the background or reference site area.

Assuming the data set for the background concentrations is normally distributed, the 95% UCL is calculated by the following four steps:

- 1) Calculate the arithmetic mean of the untransformed data.
- 2) Calculate the standard deviation of the untransformed data.
- 3) Determine the one-tailed *t*-statistic (see a statistical text for the Student *t Distribution* table).
- 4) Calculate the UCL using the following equation:

$$\text{UCL} = \bar{x} + t (s / \text{square root of } n)$$

Where;

UCL = Upper Confidence Level of the Mean to be used as the maximum probable background concentration (MPBC).

x = Mean of the data

s = Standard deviation of the data

t = Student-t statistic from statistical textbook

n = number of samples

APPENDIX B (continued)

Example Calculation

10 samples were taken at a background site for mercury that had comparable hydrologic and sediment characteristics as the site under study but was not influenced by the sources of mercury contamination at the study site. The background sample concentrations for mercury were: 15, 30, 33, 55, 62, 83, 97, 104, 125, and 155 ug/kg.

Following the 4 steps above –

- 1) Mean mercury concentration - 75.9 ug/kg
- 2) Standard deviation – 45.02
- 3) Student t-statistic value for one-tail test. $n = 10$ samples. Degrees of freedom $10 - 1 = 9$.
t-distribution - 1.833
- 4) $UCL = \bar{x} + t (s / \text{square root of } n)$
 $UCL = 75.9 + 1.833 (45.02 / \text{square root of } 10)$
 $UCL = 75.9 + 1.833 (45.02 / 3.16)$
 $UCL = 75.9 + 1.833 (14.25)$
 $UCL = 75.9 + 26.12$
 $UCL = 102.02 \text{ ug/kg}$

The UCL value for mercury of 102.02 ug/kg becomes the maximum probable background concentration (MPBC) that will be used to compare the study site concentrations against. Concentrations of mercury in study site sediment samples that are greater than the 102.02 ug/kg value can be considered to be influenced by the sources of mercury other natural or ubiquitous (e.g., atmospheric depositions) sources. As discussed above in the main body of this document, the percent fine fractions need to be looked at in the sediment samples under comparison. If the relative contribution of fines are the same in the samples from the background site and the study site, then no adjustments need to be made. If the percent fines are significantly different between the samples and the sites, then considerations for normalization of the mercury concentrations to the fine content should be looked at in order to do relevant site-to-site comparisons of metal concentrations.

The CBSQG TEC value for mercury is 180 ug/kg (Table 1 above). The MPBC for mercury in this example at 102.02 ug/kg is less than the MPBC value. An interpretation of this relationship is that benthic macroinvertebrates are possibly tolerant of mercury concentrations that are somewhat greater than background concentrations. This relationship may come into play if a decision is made to use the greater of the MPBC or the TEC value to drive the cleanup of a site.

An example of what fewer background samples would mean to the resulting MPBC value can be seen by the following example using only 4 of the sample results for mercury – 30, 62, 104, and 155 ug/kg.

- 1) Mean mercury concentration – 87.8ug/kg
- 2) Standard deviation – 54.11
- 3) Student t-statistic value for one-tail test for n = 4 samples. Degrees of freedom 4 – 1 = 3
t-distribution – 2.353
UCL = $x + t (s / \text{square root of } n)$
UCL = 87.8 + 2.353 (54.11 / square root of 4)
UCL = 87.8 + 2.353 (54.11 / 2)
UCL = 87.8 + 2.353 (27.06)
UCL = 87.8 + 63.7
UCL = 151.5 ug/kg

APPENDIX C

Notes on Dioxins and Furans

- Polychlorinated dibenzo-p-dioxins and dibenzofurans are ubiquitous contaminants, primarily from combustion sources. Background concentrations are normally in the range 0.15 - 2.5 pg TCDD-EQ/g Sediment.
- There are concerns with the other 2,3,7,8-substituted congeners beside 2,3,7,8-TCDD and TCDF. There is a need to request that all 17 - 2,3,7,8 substituted congeners be analyzed for. Analytical costs are high. To do an adequate environmental assessment, detection levels for 2,3,7,8-TCDD need to be at the single digit pg/g level.
- Dioxins and furans are not produced commercially but are unintended by-products from various chemical manufacturing and other sources.
- Dioxins and furans are found in discharges from wood treatment facilities that use pentachlorophenol, kraft pulp mills, and chemical manufacturing plants that produced pentachlorophenol, trichlorophenol, and the pesticides 2,4-D and 2,4,5-T. Also, if a water body has a history of aquatic applications of the herbicide Silvex, residual dioxins and furans may be present
- For some perspective, the department's landspreading program for paper mill sludges sets limits for spreading based on land uses - Silviculture - 10 pg/g; Agriculture - 1.2 pg/g; Grazing - 0.5 pg/g.
- Examples of high levels of dioxins/furans at Wisconsin sediment sites include - Crawford Creek - discharge from wood treatment facility that used pentachlorophenol - 5,500 pg TCDD-EQ/g; Military Creek-discharge from wood treatment facility that used pentachlorophenol- 2,500 pgTCDD-EQ/g; Fox River - paper mill discharges - 21 - 441 pg TCDD-EQ / g; and Wisconsin River - paper mill discharges - 31 - 78 pg TCDD-EQ / g.
- The recommendation is that dioxin and furan analysis only be done where there is a demonstrated need given the identification of possible historical sources at a site.
- The different 2,3,7,8 – substituted dioxins and furans have toxic equivalency factors (TEF) assigned to them relative to their toxicity compared to 2,3,7,8-TCDD. The table below provides a method to calculate the summed TCDD equivalent concentration for all the substituted forms in a sample.

2,3,7,8 - Substituted Dioxin and Furan Congeners

Worksheet For Calculating 2,3,7,8-TCDD Equivalent Concentrations	Sediment Concentration pg/g (ppt) dry weight	Toxic Equivalency Factors (TEF) (Equivalency to 2,3,7,8-TCDD)	pg/g x TEF = Toxic Equivalency to 2,3,7,8-TCDD Or TCDD-EQ
Dioxins			
2,3,7,8-TetraCDD		1.0	
1,2,3,7,8-PentaCDD		0.5	
1,2,3,4,7,8-HexaCDD		0.1	
1,2,3,6,7,8-HexaCDD		0.1	
1,2,3,7,8,9-HexaCDD		0.1	
1,2,3,4,6,7,8-HeptaCDD		0.01	
OctaCDD		0.001	
Furans			
2,3,7,8-TetraCDF		0.1	
2,3,4,7,8-PentaCDF		0.5	
1,2,3,7,8-PentaCDF		0.05	
1,2,3,4,7,8-HexaCDF		0.1	
1,2,3,6,7,8-HexaCDF		0.1	
2,3,4,6,7,8-HexaCDF		0.1	
1,2,3,7,8,9-HexaCDF		0.1	
1,2,3,4,6,7,8-HeptaCDF		0.01	
1,2,3,4,7,8,9-HeptaCDF		0.01	
OctaCDF		0.001	
Sum of TCDD-EQ of Individual Substituted Dioxin and Furan Congeners (___ pg TCDD-EQ / kg sediment) =			

APPENDIX D

Dry Weight Sediment Concentrations of Organic Compounds Normalized to 1% TOC for Comparison with CBSQGs and Grain Size Normalizations of Metals for Site-to-Site Comparisons					
Sample Site:		Example Calculations (Request a copy of Excel Spreadsheet)			
Sample Description:					
Date:					
ug/g = ppm = mg/kg					
ng/g = ppb = ug/kg					
TOC reported as mg/kg ÷ 10,000 = % TOC					
	Bulk Chemistry				
Parameter	Concentration	Units	% TOC in Sample		
TOC	25,000	mg/kg	2.5%		
Dry Wt. Concentration ÷ TOC expressed as a % = Concentration Normalized to 1% TOC					
PAHs	Dry Weight Concentration		Normalized to 1% TOC for Comparison With CBSQG Values		
Acenaphthene	3.2	ug/kg	1.3	ug/kg @ 1% TOC	
Acenaphthylene	5.9	ug/kg	2.4	ug/kg @ 1% TOC	
Anthracene	57.2	ug/kg	22.9	ug/kg @ 1% TOC	
Fluorene	77.4	ug/kg	30.9	ug/kg @ 1% TOC	
Naphthalene	176	ug/kg	70.4	ug/kg @ 1% TOC	
2-Methylnaphthalene	20.2	ug/kg	8.1	ug/kg @ 1% TOC	
Phenanthrene	204	ug/kg	81.6	ug/kg @ 1% TOC	
Benzo(a)anthracene	108	ug/kg	43.2	ug/kg @ 1% TOC	
Benzo(a)pyrene	150	ug/kg	60	ug/kg @ 1% TOC	
Benzo(e)pyrene	150	ug/kg	60	ug/kg @ 1% TOC	
Benzo(b)fluoranthene	240	ug/kg	96	ug/kg @ 1% TOC	
Benzo(k)fluoranthene	240	ug/kg	96	ug/kg @ 1% TOC	
Benzo(g,h,i)perylene	170	ug/kg	68	ug/kg @ 1% TOC	
Chrysene	166	ug/kg	66.4	ug/kg @ 1% TOC	
Dibenz(a,h)anthracene	33	ug/kg	13.2	ug/kg @ 1% TOC	
Fluoranthene	423	ug/kg	169.2	ug/kg @ 1% TOC	
Indeno(1,2,3-c,d)pyrene	200	ug/kg	80	ug/kg @ 1% TOC	
Pyrene	195	ug/kg	78	ug/kg @ 1% TOC	
Total PAHs (sum of 18 PAHs listed above)	2618.9	ug/kg	1,047.6	ug/kg @ 1% TOC	

PCB and Pesticides		Concentration	Units	Normalized to 1% TOC for Comparison With CBSQG Values	
PCBs (total)		60	ug/kg	21	ug/kg @ 1% TOC
Aldrin		2	ug/kg	0.8	ug/kg @ 1% TOC
BHC		3	ug/kg	1.2	ug/kg @ 1% TOC
a-BHC		6	ug/kg	2.4	ug/kg @ 1% TOC
B-BHC		5	ug/kg	2	ug/kg @ 1% TOC
Y-BHC (lindane)		3	ug/kg	1.2	ug/kg @ 1% TOC
Chlordane		3.2	ug/kg	1.3	ug/kg @ 1% TOC
Dieldrin		1.9	ug/kg	0.8	ug/kg @ 1% TOC
Sum pp DDD		4.9	ug/kg	1.9	ug/kg @ 1% TOC
Sum pp DDE		3.2	ug/kg	1.3	ug/kg @ 1% TOC
Sum op + pp DDT		4.2	ug/kg	1.7	ug/kg @ 1% TOC
Sum of DDT and metabolites		5.3	ug/kg	2.1	ug/kg @ 1% TOC
Endrin		3	ug/kg	1.2	ug/kg @ 1% TOC
Heptachlor Epoxide		2.5	ug/kg	1.0	ug/kg @ 1% TOC
Mirex		7	ug/kg	2.8	ug/kg @ 1% TOC
Toxaphene		1	ug/kg	0.4	ug/kg @ 1% TOC
Metals					
Particle Size	% sand	50	%		
	% silt	25	%	Fine Fraction	
	% clay	25	%	Silt + Clay = 50% or 0.50	
Dry Wt. Concentration ÷ Fines expressed as decimal fraction = Normalized to Fine Concentration					
Metals	Dry Weight Concentration (Compare with CBSQGs)		Normalized to Fine Concentration for Site-to-site Comparisons(Not for Comparison with CBSQGs)		
Antimony	2	mg/kg	4	mg/kg fines	
Arsenic	9.8	mg/kg	19.6	mg/kg fines	
Cadmium	0.99	mg/kg	1.98	mg/kg fines	
Chromium	43	mg/kg	86	mg/kg fines	
Copper	32	mg/kg	64	mg/kg fines	
Iron	20,000	mg/kg	40,000	mg/kg fines	
Lead	36	mg/kg	72	mg/kg fines	
Manganese	460	mg/kg	920	mg/kg fines	
Mercury	0.18	mg/kg	0.36	mg/kg fines	
Nickel	23	mg/kg	46	mg/kg fines	
Silver	1.6	mg/kg	3.2	mg/kg fines	
Zinc	120	mg/kg	240	mg/kg fines	

Appendix E

Identification of Contamination that Leads to Adverse Effects

Contamination of a chemical nature (i.e., a contaminant) is a substance or substances (either organic or inorganic) that are present in environmental media such as sediments or surface waters that are found above levels that would normally occur. What is normal or background for metals or nutrients (e.g., nitrogen, phosphorus) would be those metals and nutrients at levels that originate from the natural soil types and the geochemical components of the watershed. What is normal for natural organic compounds would generally be those compounds that originate from natural watershed-source vegetative or animal matter that are deposited on the bottoms of lakes, streams, and wetlands. Organic chemicals manufactured by humans and released to the environment by various mechanisms generally do not have counterparts found in nature and therefore any levels found in environmental media would be considered potential contamination. Many manufactured organic compounds may be found ubiquitously at low levels in sediments especially in urban areas. ,

Environmental concerns arise when the level of contamination (concentration of contaminants) in surface waters and sediments leads to observed and measurable effects to biological receptors, such as 1) chronic and/or acute toxicity (the contaminant becomes a toxicant) to aquatic receptors (for example directly to aquatic life such as bottom inhabiting macroinvertebrates), and/or 2) concerns about humans and wildlife that are upper food chain organisms who may become exposed to harmful levels of contaminants principally through consumption of aquatic organisms that have bioaccumulated the contaminants. For the toxicity to aquatic organisms to be realized and/or unacceptable levels of bioaccumulation to occur, the aquatic organism has to (a) be exposed to the potential toxicant in its habitat, (b) the potential toxicant has to be in a form available for uptake, and (c) the uptake or dose of the contaminant has to be at a level that causes toxicity to the particular exposed receptor or results in levels of bioaccumulation that may pose risks to humans and/or wildlife who consume the exposed receptor as food.

Elevated levels of nutrients can lead to eutrophication of water bodies and production and deposition plant materials in sediments that deplete oxygen levels in the water body when they decompose. Addition and decomposition of natural organic matter and anthropogenic-added organic matter in sediments can lead to production of hydrogen sulfide and ammonia levels that may be detrimental to benthic organisms.