National Coastal Condition Assessment 2015 Technical Support Document

U.S. Environmental Protection Agency
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LIST OF ACRONYMS

AUF	Area Use Factor used to calculate exposure concentration in EFTCI
AV	Assistance Visit
BW	Body Weight
CCE	Calibrator Cell Equivalent
CHLA	Chlorophyll *a*
DIN	Dissolved Inorganic Nitrogen
DIP	Dissolved Inorganic Phosphorus
DO	Dissolved Oxygen
EFTCI	Ecological Fish Tissue Contaminant Index
EPA	Environmental Protection Agency
FIR	Food Ingestion Rate
FOM	Field Operations Manual
GRTS	Generalized Random Tessellation Stratified
HQ	Hazard Quotient
IAC	Internal Amplification Control
IM	Information Management
LOAEL	Lowest Observed Adverse Effect Level
LOM	Laboratory Operations Manual
LRM	Logistic Regression Model
M-AMBI	multivariate AZTI Marine Biotic Index
MDL	Method Detection Limit
mERM-Q	mean Effects Range-Median Quotient
mPEC-Q	mean Probable Effects Concentrations Quotient
MQOs	Measurement Quality Objectives
NARS	National Aquatic Resource Surveys
NCA	National Coastal Assessment
NCCA	National Coastal Condition Assessment
NOAEL	No Observed Adverse Effect Level
OTI	Oligochaete Trophic Index
PAHs	Polycyclic Aromatic Hydrocarbons
PAR	Photosynthetically Active Radiation
PBS	Phosphate Buffer Solution
PCBs	Polychlorinated Biphenyls
PDE	Percent Difference in Enumeration
PFAS	Per- and Polyfluoroalkyl Substances
PTD	Percent Taxonomic Disagreement

QA	Quality Assurance
QAPP	Quality Assurance Project Plan
QC	Quality Control
qPCR	quantitative Polymerase Chain Reaction
QRG	Quick Reference Guide
ROC	Receptors of Concern
SAV	Submerged Aquatic Vegetation
S:N	Signal to Noise ratio
SCI	Sediment Contaminant Index
SEG	Site Evaluation Guidelines
SPC	Sample Processing Control
SQG	Sediment Quality Guideline
SQI	Sediment Quality Index
STI	Sediment Toxicity Index
SV	Screening Value
TN	Total Nitrogen
TP	Total Phosphorus
TRV	Estimated Wildlife Toxicity Reference Value
TSC	Target Sequence Copies

1 INTRODUCTION

The National Coastal Condition Assessment 2015 Report (USEPA 2021) presents an overview and results of the sampling effort undertaken by the U.S. Environmental Protection Agency (USEPA) and its state partners during the National Coastal Condition Assessment (NCCA) 2015. NCCA provides information on the ecological condition of the nation's estuaries[1] and the nearshore waters of the Laurentian Great Lakes, both on a national and a regional scale. It summarizes change in conditions in estuaries from the precursor National Coastal Assessment (NCA) conducted from 2004-2006 and the NCCA 2010, and changes in conditions in the nearshore waters of the Great Lakes from the NCCA 2010. This technical support document provides details on the quality assurance measures and analyses techniques for the survey. The objectives of the NCCA are to determine:

- **Condition of Coastal Waters.** What is the condition of the nation's estuarine and Great Lakes nearshore waters?
    - Estimate, with a margin of error of ± 5%, the proportion of area of the nation's estuarine waters in good, fair or poor conditions, with 95% confidence.
    - Estimate, with a margin of error of ± 5%, the proportion of all Great Lakes nearshore waters in good, fair or poor conditions, with 95% confidence.
    - Estimate with a margin of error of ± 15% the proportion of NCCA regional estuarine waters in good, fair or poor conditions, with 95% confidence.
    - Estimate with a margin of error of ± 15%, the proportion of each Great Lake nearshore waters in good, fair or poor conditions, with 95% confidence.
- **Change over time.** Are conditions in our coastal waters getting better, worse or staying the same?
- **Impact of stressors on aquatic and estuarine life**. How widespread are major pollutants and other stressors that affect estuarine and Great Lakes nearshore waters?

1.1 ADDITIONAL RESOURCES FOR SURVEY OPERATIONS

A series of protocols were used to ensure consistency throughout the survey operations. The following documents provide the field sampling methods, laboratory procedures, quality assurance measures, and site selection guidelines for the NCCA 2015.

- U.S. EPA. 2015. National Coastal Condition Assessment: Field Operations Manual. EPA-841-R-14-007. Washington, D.C. (FOM, USEPA 2015a)
- U.S. EPA. 2015. National Coastal Condition Assessment: Laboratory Operations Methods Manual. EPA-841-R-14-008. Washington, D.C. (LOM, USEPA 2015b)
- U.S. EPA. 2015. National Coastal Condition Assessment: Quality Assurance Project Plan. EPA-841-R-14-005. Washington, D.C. (QAPP)

[1] While areas where riverine water meets the Great Lakes are referred to as freshwater estuaries, the National Coastal Condition Assessment uses "estuary" to refer exclusively to areas where rivers meet saltwater.

- U.S. EPA. 2015. National Coastal Condition Assessment: Site Evaluation Guidelines. EPA-841-R-14-006. Washington, D.C. (SEG, USEPA 2015d)

1.2 ADDITIONAL REPORT MATERIALS

Data collected during the NCCA 2015 (**Table 1.1**) are available to download from the National Aquatic Resource Surveys (NARS) website (https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys). Data collected in conjunction with the NCCA 2015 as part of the Great Lakes Human Health Fish Fillet Tissue Study are available to download from the study's website (https://www.epa.gov/fish-tech/2015-great-lakes-human-health-fish-tissue-study). Underwater video files recorded in the Great Lakes during the NCCA 2015 are available online (gispub.epa.gov/NCCA/). Condition results for the estuarine study area, the Great Lakes study area, and additional subpopulations are available to view in the NCCA online data dashboard (https://coastalcondition.epa.gov/).

Table 1.1 NCCA 2015 data files available on NARS and other EPA websites

ncca2015_algx_data	Algal toxin data
ncca2015_bentCnt_data	Benthic invertebrate count data
ncca2015_benthicTaxa_data	Benthic invertebrate taxonomy data
ncca2015_ente_data	Enterococci data
ncca2015_fplg_data	Mercury concentration in fish fillet plug data
ncca2015_ftis_data	Contaminant concentration in whole fish data
ncca2015_hydroprofile_data	Hydrographic profile data
ncca2015_indicesCondition_data	Indicator condition data
ncca2015_micx_data	Microcystin data
ncca2015_secchi_data	Secchi depth data
ncca2015_sedChem_data	Sediment contaminant data
ncca2015_sedtoxControlRep_data	Sediment toxicity control replicate data
ncca2015_sedtoxControlSummary_data	Sediment toxicity control data summary
ncca2015_sedtoxSampleRep_data	Sediment toxicity sample replicate data
ncca2015_sedtoxSampleSummary_data	Sediment toxicity sample data summary
ncca2015_sitedata_data	Site data
ncca2015_waterChem_data	Water nutrient and chlorophyll a data
ncca2015_wide_fishcollection_data	Fish collection data
ncca2015_GreatLakes_Phytoplankton_data.xlsx	Great Lakes phytoplankton data
2015 Great Lakes Human Health Fish Fillet Tissue Study data	Mercury data, PCB Data PFAS data, Dioxin/Furan data, Fatty Acids data,

1.3 REFERENCES

U.S. Environmental Protection Agency (USEPA). 2021. National Coastal Condition Assessment 2015 Report. Office of Water and Office of Research and Development. EPA-841-R-21-001. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 2015a. National Coastal Condition Assessment: Field Operations Manual. EPA- 841-R-14-007. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 2015b. National Coastal Condition Assessment: Laboratory Operations Methods Manual. EPA-841-R-14-008. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 2015c. National Coastal Condition Assessment: Quality Assurance Project Plan. EPA-841-R-14-005. Washington, D.C.

U.S. Environmental Protection Agency (USEPA). 2015d. National Coastal Condition Assessment: Site Evaluation Guidelines. EPA-841-R-14-006. Washington, D.C.

2 QUALITY ASSURANCE

NCCA implemented measures to assess the quality of its operations and data throughout the survey. This chapter documents NCCA's adherence to the requirements of EPA's quality system described below. The following sections describe quality assurance for the statistical survey design, field operations, laboratory measurements, data management, and report preparation. These quality assurance measures are vital to conducting a national scale survey and allow for comparable data to be collected across the country.

2.1 INTRODUCTION

The EPA quality system incorporates a national consensus standard for quality systems authorized by the American National Standards Institute (ANSI) and developed by the American Society for Quality Control (ASQC, ANSI/ASQ E4-2004, Quality Systems for Environmental Data and Technology Programs – Requirements with Guidance for Use). EPA Order CIO 2105.0, dated May 5, 2000, requires all of its component organizations to participate in an agency-wide quality system. The EPA Order also requires quality assurance project plans or "equivalent documents" for all projects and tasks involving environmental data.

In accordance with the EPA order, the Office of Water (OW) developed the Office of Water Quality Management Plan (QMP; USEPA 2015e) to describe OW's quality system that applies to all water programs and activities collecting or using environmental data. As required by the EPA Order and OW QMP, NCCA developed and abided by its Quality Assurance Project Plan (QAPP; USEPA 2015c) throughout the survey. The NCCA QAPP contains elements of the overall project management, data quality objectives, measurement and data acquisition, and information management. Any data excluded for not meeting QC requirements are noted in each indicator section of this document.

The QAPP and its companion documents (Field Operation Manual (USEPA 2015a), Laboratory Operations Manual (USEPA 2015b) and Site Evaluation Guidelines (USEPA 2015d) describe detailed procedures for implementing the field and lab work for the survey (see Section 1.1):

The four documents together address all aspects of NCCA's data acquisition and evaluation. The Laboratory Operations Manual (LOM; USEPA 2015b) also lists measurement quality objectives (MQOs) used to evaluate the level of quality attainment for individual survey metrics.

Every person involved in NCCA was responsible for abiding by the QAPP (USEPA 2015c) and adhering to the procedures specified in its companion document in order for comparable data to be collected by different field and laboratory personnel. Moreover, every NCCA participant was trained in the requirements applicable to the person's role in the survey. For example, field crews were trained in the Field Operations Manual (FOM; USEPA 2015a) procedures and applicable QAPP requirements by attending a combined classroom and hands-on training in field procedures.

2.2 SURVEY DESIGN

The NCCA survey design was based upon statistical concepts that are well accepted by the scientific community. As described in the following sections, the survey design quality objectives were met by requirements of the statistical design, completeness of implementing the design, and consistency with established procedures. By applying the statistical concepts of this design, the survey was able to meet the following overarching data quality objectives:

- Estimate, with a margin of error of ± 5%, the proportion of the nation's estuarine waters in good, fair or poor conditions, with 95% confidence.
- Estimate, with a margin of error of ± 5%, the proportion of all Great Lakes nearshore waters in good, fair or poor conditions, with 95% confidence.
- Estimate with a margin of error of ± 15% the proportion of NCCA regional estuarine waters in good, fair or poor conditions, with 95% confidence.
- Estimate with a margin of error of ± 15%, the proportion of each Great Lake nearshore waters in good, fair or poor conditions, with 95% confidence.

2.2.1 *Statistical Design*

The population surveyed for NCCA is the area of estuarine and Great Lakes nearshore waters of the contiguous United States. Surveying a population of this size presents logistical and resource challenges that are overcome by using a probabilistic survey design. An extensive body of statistical literature supports making statements about large populations by sampling representative sites (Kish 1965). Sample surveys have been used in a variety of fields (*e.g.*, monthly labor estimates) to determine the status of populations of interest, especially if the population is too numerous for a complete census or if a census is unnecessary to reach the level of precision desired for describing the population's status. In natural resource fields, probability sampling surveys have often been used to estimate the conditions of the entire population. For example, the National Agricultural Statistics Survey conducted by the U.S. Department of Agriculture and the Forest Inventory Analysis conducted by the Forest Service (Bickford *et al.* 1963, Hazard and Law 1989) both use probability-based sampling to monitor and estimate the condition and productivity of agricultural and forest resources. To select the sites for the survey, NCCA used a peer-reviewed (Stevens 1994, Stevens and Olsen 1999) probability design based on the fundamental requirement of an explicitly defined regional resource population, wherein the sample is constrained to reflect the spatial dispersion of the population.

2.2.2 *Completeness*

To ensure that the implementation of the NCCA sample design resulted in adequate measurements, the survey included completeness requirements for field sampling and laboratory analyses. The QAPP requires that valid data for individual indicators be acquired from enough sites to make subpopulation estimates with a specified level of confidence or sampling precision (QAPP estimate

was 90% of planned sampling locations (or X-sites)). Samples were successfully collected at most sites. See **Table 2.1**

Crews were not able to collect some sample types at all sites for various reasons. For example, sediment contaminant and toxicity, as well as benthic macroinvertebrate sample collection rates may have been hampered by bedrock substrates without sediment (for which no sample was possible) or that were too hard or too soft to obtain a successful grab. In the Great Lakes, presence of invasive mussel beds may have also prevented successful sediment or benthic macroinvertebrate sample collection. Fish tissue sample collection success was subject to the movement of fish and availability of suitable fish habitat surrounding the X-site. Mercury in fish fillet sample collection success was lower than ecological fish tissue sample collection success because the human health target species list was more restrictive and subject to a minimum size requirement in order to be used for analysis. EPA identified ways to improve sampling success including emphasizing the importance of collecting all samples during field crew training, increasing the radius around the designated X-sites from which samples may be collected (e.g., for fish tissue), and requiring crews to attempt to sample more times at a site and document reasons for missing samples. These improvements in the 2015 NCCA sampling efforts led to across-the-board increases in sampling success in both estuaries and the Great Lakes. While collection success for some samples didn't reach 90% of planned sites, enough samples were collected to achieve statistical significance in making population estimates. Missing samples contribute to the area estimated as "unassessed" for each indicator.

Table 2.1 Sample collection success (percentage of expected sites sampled) in estuaries and the Great Lakes in 2010 and 2015

Design	Sample Type	2010	2015
Estuaries	Benthic macroinvertebrates	94%	97%
	Dissolved inorganic nutrients	99%	100%
	Enterococci*		100%
	Ecological fish tissue samples	80%	87%
	Fish fillet samples for Hg*		83%
	Microcystin in water*		100%
	Sediment contaminant	93%	97%
	Sediment toxicity	93%	97%
	Chlorophyll a	99%	100%
	Total N & P	99%	100%
Great Lakes	Benthic macroinvertebrates	79%	81%
	Dissolved inorganic nutrients	98%	100%
	Enterococci*		100%
	Ecological fish tissue samples	68%	85%
	Fish fillet plug samples for Hg*		81%
	Microcystin in water*		100%
	Sediment contaminants	78%	81%
	Sediment toxicity	73%	80%
	UW video footage**	75%	97%
	Chlorophyll a	98%	100%
	Phytoplankton**	96%	100%
	Total N & P	98%	100%
	Fish homogenized fillet samples for Hg, PCBs and PFAS	100%	100%

* Enterococci, microcystin and Hg analysis in fish fillets were introduced to the NCCA in 2015.
** Underwater video footage and phytoplankton are collected in the Great Lakes only.

2.2.3 *Comparability*

Comparability is defined as the confidence with which one data set can be compared to another (Stanley and Verner, 1985; Smith et al., 1988). For all indicators, NCCA ensured comparability by the use of standardized sampling procedures, sampling equipment and analytical methodologies by all sampling crews and laboratories. For all measurements, reporting units and format are specified, incorporated into standardized data recording forms, and securely transferred into a centralized information management system. Because EPA used the same comparable methods measures to collect data in the NCA in 2005-06 and the first NCCA in 2010, the data can be compared across those studies. The following sections on field and laboratory operations describe additional measures to ensure consistency in NCCA.

2.3 QUALITY ASSURANCE IN FIELD OPERATIONS

The requirements and methods presented in the Field Operations Manual (FOM) ensured that quality objectives were attainable and survey activities were manageable. As described below, NCCA tested its FOM, trained crews using the FOM and visited crews during the field season.

2.3.1 *Field Method Pilot Testing*

Representatives from the NCCA team, logistics and data management contractors, and state partners tested sampling methods, paper and electronic field forms, and field equipment described in the FOM. The test run assessed the accuracy and clarity of the FOM's instructions for executing the procedures and quality assurance practices. The test run also evaluated sampling logistics, sample preparation, and sample shipping instructions. As a result of lessons learned during the test run, NCCA staff amended and improved the FOM prior to field crew training.

2.3.2 *Training of Field Trainers and Assistance Visitors*

Before training field crews, members of the NCCA team, oversight staff, contractor trainers, and other experts tested the training materials during intensive classroom and hands on training sessions. This "train-the-trainer" event served two primary purposes. First, the event was designed to make sure that all trainers understood the methods and provided consistent instruction to field crews. Second, it provided another opportunity to ensure that the field documents and forms were clear and accurate. During this training event, the attendees tested the materials to ensure that the instructions were correct and easy to execute, and they practiced training the methods. The training materials included the FOM, Quick Reference Guide (QRG), field forms and PowerPoint presentations. As a result of the training, practice training sessions and expert discussions, NCCA staff amended and improved training materials, the FOM and the QRG before the field crew training.

2.3.3 *Field Crew Training*

To ensure consistency across field crews, all field crews leads and their alternates were required to attend a 2-3-day training session prior to visiting any field site. Led by NCCA trainers, regional field crew training consisted of classroom and field-based sessions. The session topics included conducting site reconnaissance; recording field observations and in situ water quality measurements; collecting field samples; preparing, packing and shipping sample containers; and use of the standardized field forms. The field crew leaders were taught to review every form and verify that all hand-entered data were complete and correct.

2.3.4 *Field Assistance Visits*

To further assist the crews in correctly implementing the field procedures and quality steps, a NCCA team member or contractor trainer visited every NCCA field crew during the field season. These visits, known as assistance visits (AV), provided an opportunity to observe field crews in the normal course of a field day, assist in correctly applying the procedures, and document the crew's adherence to sampling procedures. If circumstances were noted where a field crew was not conducting a procedure properly, the observer recorded the deficiency, reviewed the appropriate procedure with field team, and assisted the field crew until their technique conformed with expectations.

2.3.5 *Revisits of Selected Field Sites*

Useful metrics and indicators tend to have high repeatability. That is, among-site variability will be greater than sampling variability based on repeat sampling at a subset of sites. To evaluate within-year sampling variability, the NCCA design required crews to revisit 10 percent of the sites. These sites were sampled twice during the NCCA index period. To quantify repeatability between first and second visits, NARS uses one of two metrics, either signal:noise (S:N) or contingency tables. Signal:noise is defined as the ratio of variance associated with different sites (signal) to the variance associated with repeated visits to the same site (noise) (Kaufmann et al. 1999). It is used to determine the repeatability of parameters or indices that produce a continuous numerical result. For indices that produce a categorical result (i.e., Good, Fair or Poor), contingency tables are used to visualize agreement between condition ratings for the first and second visits. When calculating the S:N ratio, all sites are included in the signal, whereas only the second visit to revisit sites contribute to the noise component. Metrics with high S:N are more likely to show consistent results. Contingency tables provide a visual representation of the number of sites that were rated good, fair or poor for both visits, as well as the sites that showed disagreement between sites, and the magnitude of that difference (i.e., sites rated good for one visit and poor for the other showed greater disagreement than those that were either good for one visit and fair for the other or fair for one visit and poor for the other). Signal:noise ratios and contingency tables are not used to look at variance for indicators that have primarily non-detects for results. Where applicable, S:N and contingency tables, are presented in this document with each of the indicators.

2.4 LABORATORY QUALITY ASSURANCE AND QUALITY CONTROL

The NCCA laboratories used standard methods and/or followed the requirements (e.g., performance-based objectives) in the Laboratory Operations Manual (LOM). The QAPP identified the overall quality requirements and the LOM provided methods that could be used to achieve the quality requirements. If a laboratory used a different method, it still had to meet the QA requirements as described in the QAPP.

2.4.1 *Basic Capabilities*

All laboratories were required to submit documentation of their analytical capabilities prior to analyzing any NCCA samples. NCCA team members reviewed documentation to ensure that the laboratories could meet required measurement quality objectives (MQOs; e.g., reporting limits, detection limits, etc.). National Environmental Laboratory Accreditation Conference (NELAC) certification, satisfactory participation in round-robin or other quality assurance assessments were considered acceptable capabilities documentation.

2.4.2 *Benthic Macroinvertebrate Identifications*

For benthic macroinvertebrate taxonomy, laboratories were required to use the same taxa lists, conduct regular internal QA checks, and participate in an independent quality check. All participating laboratories identified organisms using the most appropriate technical literature that was accepted by the taxonomic discipline and reflected the accepted nomenclature at the time of the survey. The Integrated Taxonomic Information System (ITIS, https://www.itis.gov/) was also used to verify nomenclatural validity and reporting for freshwater species. The World Register of Marine Species (WoRMS, http://www.marinespecies.org/) was used for marine species.

Taxonomic accuracy is evaluated by comparing identifications of the same organisms by independent primary and secondary laboratories. Each primary laboratory provided the organisms from 10 percent of its samples (with a minimum of three samples per lab), to a secondary laboratory for an independent evaluation. EPA, supported by an expert contractor, assessed the primary and secondary identifications and then held reconciliation calls to allow the taxonomists to discuss organisms that were identified differently. As part of this process, recommendations and corrective actions were identified to address inaccurate taxonomic identification, and measurement objectives were established to ensure the data were of sufficient quality for the NCCA.

The NCCA 2015 resulted in the collection of 1,269 benthic samples, of which 775 were from estuarine waters and 494 were collected in the Great Lakes. The majority (1,214) of the samples were processed by EPA's primary contract lab. The remainder were processed by labs contracted to the states of Maryland and Virginia. The rate of taxonomic error in the NCCA 2015 benthic dataset was minor, and the data are acceptable for additional analyses. Results of QC analyses are detailed in the following paragraph.

As approximately 10% of the overall dataset, 127 samples were randomly selected for quality control re-identification by the secondary laboratory. Comparison of the results of whole sample re-identifications provided a Percent Taxonomic Disagreement, a measure of taxonomic precision wherein the number of agreements in identification between a primary taxonomist and a quality control taxonomist are compared to the number of specimens in a sample (PTD; Equation 2-1, **below**). The majority of Great Lakes and estuarine samples were analyzed by the same laboratory; therefore, the overall mean PTD (10.1%) reflects samples from both populations combined. The

actual PTD was better than the NCCA measurement objective identified in the QAPP, which allowed a PTD of 15 percent. Comparison of counts was quantified by calculation of percent difference in enumeration, a relative measure of count precision within a sample, wherein the difference in specimen counts in a sample between a primary and QC taxonomist is compared to the sum of the two counts (PDE; Equation 2-2, **below**). The overall PDE was 1.5 percent, which was better than the NCCA measurement objective of 5 percent as identified in the QAPP. See **Table 2.2** for a breakout of PTD and PDE by NCCA Region.

Table 2.2 Benthic taxonomy performance measure, by NCCA Region

Coastal Region	*n*	PTD		PDE	
		Avg	SD	Avg	SD
Estuarine Overall	78	10.4%	10.6%	1.9%	3.7%
Northeast	21	10.5%	8.2%	1.7%	1.4%
Mid-Atlantic	6	3.9%	3.3%	1.6%	1.9%
Southeast	13	7.4%	14.9%	2.6%	6.8%
Gulf of Mexico	25	13.3%	11.6%	2.3%	4.1%
West	13	10.7%	7.3%	0.8%	0.8%
Great Lakes	49	9.6%	11.4%	1.0%	1.3%

$$PTD = \left[1 - \left(\frac{comp_{pos}}{N}\right)\right] x\ 100 \qquad \textbf{Equation 2-1}$$

Where $comp_{pos}$ is the number of agreements, and N is the total number of individuals in the larger of the two counts.

$$PDE = \left(\frac{|Lab1 - Lab2|}{Lab1 + Lab2}\right) x\ 100 \qquad \textbf{Equation 2-2}$$

Even when the measurement objectives were met, laboratories implemented recommendations and corrective steps. If, for example, it was evident that empty mollusk shells were being identified and recorded in one or more of the QC samples, the laboratories needed to verify that they had not counted empty mollusk shells in their other samples.

2.4.3 *Chemical Analyses*

For quality assurance of chemical analyses, laboratories used QC samples which are similar in analyte concentration range to samples being measured. QC samples provide estimates of precision and bias that are applicable to sample measurements. To ensure the ongoing quality of data during analyses, every water sample analysis batch was required to include QA samples to verify the precision and accuracy of the equipment, reagent quality, and other quality measures. These checks were completed by analyzing blanks or samples spiked with known quantities of reference materials, duplicate analyses of the same samples, or other appropriate evaluations. The laboratories reported quality assurance results along with each batch of sample results to the NCCA QA Coordinator for review for compliance with the data quality objectives in the QAPP. Excursions from the limits of the data quality objectives were marked or "flagged" for further investigation. In addition, laboratories reported holding times. Holding time requirements for analyses ensure analytical results are representative of conditions at the time of sampling. The NCCA team reviewed the data and noted any quality failures in the data files. The data analysts used the information about quality to determine whether to include or exclude data from the assessment. QA data for all NCCA data are stored in the NARS Information Management database and are available for review upon request.

2.4.4 *Sediment Toxicity Analyses*

Sediment toxicity data were reviewed, and replicates were removed from the analysis if any of the following situations were met:

- Presence of predatory organisms in a replicate and the replicate percent survival was below 100% (for marine samples only). For freshwater samples, survival is not typically impacted by predators so percent survival for freshwater samples with predators were accepted for analysis.
- Large particle size in a replicate and the replicate test percent survival appeared impacted (at least 50% less than the mean of the other replicates within the sample).
- Additional organisms were present, no organisms were loaded within a test replicate, and/or the incorrect species was used as the test organism for a sample.
- The laboratory provided mean test percent survival exceeding 100% for a replicate (and insufficient information on the number of organisms loaded in the replicate).

Note that when a replicate was removed from analysis due to QA/QC concerns, the data associated with that replicate were not used in calculating control-adjusted survival for the sample, nor were they used in the significance tests for marine samples. When a sample was removed from analysis due to QA/QC concerns that impacted the entire sample (or there were an insufficient number of replicates for that sample), the condition category for the NCCA sediment toxicity index for the site was set to "Not Assessed."

2.5 DATA MANAGEMENT AND REVIEW

Information management (IM) is integral to all aspects of the NCCA from initial selection of sampling sites through dissemination and reporting of final, validated data. Quality measures implemented for the IM system are aimed at preventing corruption of data at the time of their initial incorporation into the system and maintaining the integrity of data and information after incorporation into the system through reporting and publication of results.

Reconnaissance, field observation and laboratory analysis data were transferred from NCCA survey participants and collected and managed by the NARS IM center. Data and information were managed using a tiered approach. First, all data transferred from a field team or laboratory were physically organized (e.g., system folders) and stored in their original state. Next, NARS IM created a synthesized and standardized version of the data to populate a database that represented the primary source for all subsequent data requests, uses and needs. All samples were tracked from collection to the laboratory to ensure completeness and provide quality assurance for the survey.

The IM staff applied an iterative process in reviewing the database for completeness, transcription errors, formatting compatibility, consistency issues and other quality control-related topics. This first-line data review was a joint exercise by NARS IM and the NCCA team. A second-phase data quality review consisted of evaluating the quality of data based on MQOs as described in the QAPP. This QA review was performed by the NCCA team using a variety of qualitative and quantitative analytical and visualization approaches. Data that met the MQOs were used without restriction. Data that did not meet the MQOs were qualified and further evaluated to determine the extent to which quality control results deviated from the target MQOs. Minor deviations were noted and qualified but did not prevent data from being used in analyses. Major deviations were also noted and qualified, but data were excluded from the analyses. Data quality flags are included in the data files. Data not used for analyses because of quality control concerns account for a subset of the missing data for each indicator analysis. The missing data add to the uncertainty in condition estimates and contribute (along with "missing" data where samples were not collected or for some other reason couldn't be analyzed) to the "Not Assessed" category in the report.

2.6 NCCA 2015 REPORT

The NCCA 2015 Report provides a summary of the results from the NCCA. In addition to being extensively reviewed in-house by the NCCA team, its partners, and other EPA experts, the report underwent external peer review. This review was the final step in ensuring that the main report and its findings met the quality requirements of the QAPP. EPA contracted with an outside firm to conduct an Independent External Peer Review of the main report. The firm selected three peer reviewers who were experts in water resource monitoring, biological and ecosystem assessments, and one who is an expert in ecotoxicology. The firm provided the reviewers with a copy of the main report and the technical report, links to the NCCA Dashboard and a charge that solicited

comments specifically on the technical content, completeness and clarity, and scientific integrity of the main report. EPA used the comments from the peer reviewers to refine and review the main report.

2.7 REFERENCES

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3 SELECTION OF PROBABILITY SITES

During the summer of 2015, field crews sampled 1,060 probability sites (699 sites in estuaries and 361 sites in Great Lakes nearshore) across the country representing approximately a total of 34,597square miles (27,479 square miles in estuaries and 7,118 square miles in the Great Lakes). A subset (106) of these probability sites were sampled twice during the index period. Using standardized field methods, crews sampled estuaries as large as the Chesapeake Bay in the Mid-Atlantic region and as small as Morro Bay in California during the survey index period (June through September). Sites were selected using a random sampling technique that uses a probability-based design that is described in this chapter. The following sections describe the statistical objectives, target population, sample frame, survey design, evaluation, and statistical analysis. Details for each site are included in the site information file available to download from the NARS data webpage https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys.

3.1 OBJECTIVES

- **Condition of Coastal Waters.** What is the condition of the nation's estuarine and Great Lakes nearshore waters?
    - Estimate, with a margin of error of ± 5%, the proportion of the nation's estuarine waters in good, fair or poor conditions, with 95% confidence.
    - Estimate, with a margin of error of ± 5%, the proportion of all Great Lakes nearshore waters in good, fair or poor conditions, with 95% confidence.
    - Estimate with a margin of error of ± 15% the proportion of NCCA regional estuarine waters in good, fair or poor conditions, with 95% confidence.
    - Estimate with a margin of error of ± 15%, the proportion of each Great Lake nearshore waters in good, fair or poor conditions, with 95% confidence.
- **Change over time.** Are conditions in our coastal waters getting better, worse or staying the same?
- **Extent of stressors**. How widespread are major pollutants and other stressors that affect the aquatic life in estuarine and Great Lakes Nearshore Waters?

3.2 ESTUARINE DESIGN

3.2.1 *Target Population*

The estuarine survey was designed to assess the target population of coastal waters of the United States from the head-of-salt (0.5 parts per thousand) to confluence with ocean, including inland waterways and major embayments such as Florida Bay, Cape Cod Bay and San Francisco Bay.

3.2.2 *Sample Frame*

The NCCA 2015 sample frame (the GIS construct that is used to represent the target population) was derived from the prior National Coastal Assessment sample frame developed by the EPA Office of Research and Development (ORD) Gulf Ecosystem Measurement and Modeling Division (GEMMD; Formerly Gulf Ecology Division). The GEMMD sample frame was enhanced as part of the National Coastal Monitoring Network design by including information from NOAA's Coastal Assessment Framework, boundaries of National Estuary Programs and identification of major coastal systems. Information on salinity zones for the NCCA 2010 was obtained from NOAA (Nelson and Monaco 2004). In addition, the NCA sample frames for Delaware Bay, Chesapeake Bay, Puget Sound, and the state of South Carolina were replaced by GIS layers provided by organizations within whose jurisdictions they are found. The updated sample frame ensured that no prior areas in NCA were excluded and any differences were clearly identified in the new NCCA 2010 sample frame. For the Californian Province excluding San Francisco Bay, the GEMMD sample frame was changed to match the sample frame used for the NCA 2004 study. In 2015, the sample frame was updated to include information related to 1999-2001[2] and 2005-2006 NCA sample frames in order to provide the information required to estimate change between these periods, 2010 and 2015.

3.2.3 *Survey Design*

The NCCA 2015 estuarine survey design consisted of two independent designs. One design re-sampled sites sampled during NCCA 2010. The other design selected new sites using essentially the same survey design used for NCCA 2010. Both survey designs were a stratified design with unequal probability of selection based on area within each stratum. A Generalized Random Tessellation Stratified (GRTS) survey design for an area resource was used. The details are given below.

3.2.4 *Stratification*

The population was first divided into subgroups before sites were selected. Stratification was by major estuary based on the NOAA Coastal Assessment Framework; NEP estuaries and state. The strata, listed by state, were:

[2] The 1999-2001 data are not included in change estimates because differences in sample frame definitions incorporated for the 2005-2006 survey reduced the overall area for which comparisons can be made.

Table 3.1 Strata by State

State	Stratum[3]
Maine	AP_Casco_Bay, AP_Penobscot_Bay, AP_Other_ME
New Hampshire	AP_New_Hampshire_Estuaries
Massachusetts	AP_Buzzards_Bay, AP_Massachusetts_Bay, AP_Other_MA
Rhode Island	VP_Narragansett_Bay, VP_Other_RI
Connecticut	VP_Long_Island_Sound,
New York & New Jersey	VP_NY_NJ_Harbor, VP_Peconic_Bay, VP_Other_NY, VP_NJ_Barnegat_Inland_Bays
Delaware	VP_Delaware_Bay, VP_Other_DE
Maryland & Virginia	VP_Chesapeake_Bay, VP_Other_MD, VP_Other_VA
North Carolina	CarP_Albemarle_Pamlico_Sounds, CarP_Other_NC
South Carolina	CarP_SC_OPEN, CarP_SC_CREEK
Georgia	CarP_Other_GA
Florida	CarP_Indian_River, CarP_Other_FL, WIP_Biscayne_Bay, WIP_Charlotte_Harbor, WIP_Florida_Bay, WIP_Tampa_Bay, WIP_Other_FL, LP_Apalachee_Bay, LP_Apalachicola_Bay, LP_Pensacola_Bay, LP_Other_FL
Alabama	LP_Mobile_Bay, LP_Other_AL
Mississippi	LP_Other_MS
Louisiana	LP_West_Mississippi_Sound, LP_Atchafalaya_Vermilion_Bay, LP_Barataria_Terrabonne, LP_Breton_Chandeleur_Sound, LP_Mississippi_River, LP_Other_LA
Texas	LP_Coastal_Bend_Bays, LP_Galveston_Bay, LP_Matagorda_Bay, LP_San_Antonio_Bay, LP_Other_TX
California	CalP_San_Francisco_Bay, CalP_Other_CA, ColP_Other_CA
Oregon	ColP_Lower_Columbia_River, ColP_Other_OR
Washington	ColP_Puget_Sound, ColP_Other_WA

[3] The prefix in each stratum name represent the oceanic province in which the stratum is located: AP = Acadian Province; VP = Virginian Province; CarP = Carolinian Province; WIP = West Indian Province; LP = Louisianian Province; ColP= Columbian Province; CalP = Californian Province

Sites in major estuaries that occur in two states (e.g., Chesapeake Bay, Delaware Bay, and Lower Columbia River) are not evenly divided between the states. Rather, the sites were assigned to the state in which they occur. Long Island Sound was assigned to New York as the major polygon was divided into the portion within each state. Consequently, most Long Island Sound sites were assigned to New York.

3.2.5 *Unequal Probability Categories*

Unequal probability categories were created based on area of polygons that subdivide each major estuary. The number of size categories within a major estuary ranged from 3 to 7. The categories were used to ensure that sites were selected in the smaller polygons.

Within each stratum, the sample frame for the coastal waters consisted of multiple polygons associated with subregions of the stratum that are typically smaller estuaries, coastal water regions or main bays within the stratum. The smaller estuaries are either subregions of large estuaries or separate small estuaries within the stratum. These subregions (polygons) were categorized by area and the number of sites within the categories were assigned to ensure that sites were selected in the smaller subregions. The number of size categories within a stratum ranged from 3 to 7.

3.2.6 *Panels*

The combined designs have the following panels:

1. Base10_RVT2: Sites that were sampled in NCCA 2010 that were sampled twice in 2015
2. Base10: Sites that were sampled in NCCA 2010 that were sampled once in 2015
3. Base15: New sites that were sampled once in 2015
4. Base10_OverSamp: Sites from NCCA 2010 that were oversample sites that were only used if any Base10_RVT2 or Base10 sites could not be sampled in 2015
5. Base15_OverSamp: New sites that were oversample sites that were used if any Base15 site could not be sampled in 2015

3.2.7 *Expected Sample Size*

The planned sample size for NCCA 2015 was 684 unique sites for the conterminous 21 coastal states. The planned total number of site visits was 750 where 66 sites were sampled twice in 2015. Of the 684 unique sites, 336 sites were sites that were sampled in NCCA 2010 and 348 were new sites selected for NCCA 2015. Oversample sites were drawn to be used for replacing sites that were nontarget (did not meet the definition of target waters) or were not sampleable (e.g., site was unsafe to sample).

Table 3.2 Site Selection Summary by State and Type of Site for the Estuarine Survey

	Number of Unique Sites					Number of Over Sample Sites		
State	2010 Sites Sampled Twice in 2015	2010 Sites Sampled Once in 2015	New Sites for 2015	Total Unique Sites	**Number of Site Visits**	2010 Over Sample Sites	2015 Over Sample Sites	**Total Number of Sites Available**
AL	2	5	10	17	19	10	13	40
CA	4	26	21	51	55	29	29	109
CT	1	3	0	4	5	4	3	11
DE	0	4	10	14	14	6	9	29
FL	9	37	45	91	100	47	45	183
GA	0	3	5	8	8	4	4	16
LA	11	32	43	86	97	43	48	177
MA	5	11	20	36	41	17	21	74
MD	1	15	14	30	31	15	15	60
ME	3	17	20	40	43	23	16	79
MS	0	4	4	8	8	6	6	20
NC	4	15	18	37	41	17	22	76
NH	2	4	6	12	14	6	5	23
NJ	3	8	12	23	26	17	7	47
NY	5	14	20	39	44	16	20	75
OR	2	10	7	19	21	9	5	33
RI	2	5	9	16	18	7	7	30
SC	2	8	12	22	24	10	12	44
TX	5	20	30	55	60	28	31	114
VA	2	9	11	22	24	12	18	52
WA	3	20	31	54	57	29	31	114
Sum	**66**	**270**	**348**	**684**	**750**	**355**	**367**	**1406**

Table 3.3 Number of Sites by NCCA Reporting Region

NCCA Report Region	# Base Sites	# Over Sample Sites	Total
East Coast	322	336	658
Gulf Coast	238	254	492
West Coast	124	132	256
Total	**684**	**722**	**1406**

3.2.8 *Site Usage and Replacement*

When a "Base" site could not be sampled for any reason, the site was replaced using the following rules:

1. Base10_RVT2: When a site in this category could not be sampled it was replaced by the next available site in the Base10_OverSamp list within the same state and STRATUM_15 (where sites are in numerical SITEID_15 order within the state and stratum) and the replacement site was sampled twice in 2015.
2. Base10: When a site in this category could not be sampled it would be replaced by the next available site in the Base10_OverSamp list within the same state and STRATUM_15 (where sites are in numerical SITEID_15 order within the stratum).
3. Base15: When a site in this category could not be sampled it was replaced by the next available site in the Base15_OverSamp list within the same state and STRATUM_15 (where sites are in numerical SITEID_15 order within the stratum)

3.3 GREAT LAKES NEARSHORE DESIGN

3.3.1 *Nearshore Target Population*

The Great Lakes survey was designed to assess conditions in nearshore waters of the Great Lakes of the United States and Canada. However, the 2015 NCCA Great Lakes assessment was restricted to the United States portion so only sites drawn in the United States were evaluated and sampled. The nearshore zone is defined as the region from shoreline to 30 m depth within 5 km from shoreline. The Great Lakes include Lake Superior, Lake Michigan, Lake Huron, Lake Erie, and Lake Ontario.

3.3.2 *Nearshore Sample Frame*

The Great Lakes nearshore sample frame was first developed for the 2010 NCCA from existing standard GIS vector shoreline coverage from NOAA (USEPA 2015d; Kelly et al. 2015). That coverage was modified to include a coverage extension 500 m upstream into river mouths and to add embayment areas missing from the existing shoreline coverage.

The 2015 Great Lakes NCCA nearshore sample frame was developed by the USEPA's Office of Research and Development (ORD) Great Lakes Toxicology and Ecology Division (GLTED; formerly Mid-Continent Ecology Division; MED). The nearshore includes river mouths and estuaries, embayments, and open waters adjacent to the US shorelines. It does not include the connecting channels of the Great Lakes (water bodies between lakes plus the upper St. Lawrence River).

3.3.3 *Survey Design*

The survey design consists of two independent designs. One design re-sampled sites sampled during NCCA 2010 Great Lakes assessment. The other design selects new sites using the same survey design used for NCCA 2010. Both designs use a Generalized Random Tessellation Stratified (GRTS) survey design for an area resource.

3.3.4 *Stratification*

Both designs were stratified by Great Lake and country.

3.3.5 *Unequal Probability Categories*

Both designs use unequal probability categories where the categories are based on states or province within each Great Lake and the expected sample size is proportional to state shoreline length within each stratum.

3.3.6 *Panels*

The combined designs had the following panels:

1. Base10_RVT2_FT: Sites sampled in NCCA 2010 that were sampled twice in 2015 and once for Fish Tissue study
2. Base10_FT: Sites sampled in NCCA 2010 that were sampled once in 2015 and for Fish Tissue study
3. Base10: Sites sampled in NCCA 2010 that were sampled once in 2015 and not for Fish Tissue study
4. Base15_RVT2: New sites in Canadian[4] portion of the design that were to be sampled twice in 2015
5. Base15_FT: New sites that were sampled once in 2015 and for Fish Tissue study
6. Base15: New sites that were sampled once in 2015 and not for Fish Tissue study
7. Base10_OverSamp: Sites from NCCA 2010 that were oversample sites that were only used if any Base10_RVT2 or Base10 sites could not be sampled in 2015
8. Base15_OverSamp: New sites that were oversample sites that were only used if any Base15 site could not be sampled in 2015

3.3.7 *Expected Sample Size*

The base sample design assigned 45 sites to the United States portion of nearshore waters of each of the five Great Lakes for a total of 225 sites (**Table 3.4**, **Table 3.5**). Samples in each lake were

[4] While sites were drawn in Canada, they were not sampled and the NCCA 2015 estimates are exclusive to waters within U.S. jurisdiction.

allocated among bordering states' waters proportionally by shoreline length. Five sites in each Great Lake were to be sampled twice in 2015 for a total of 250 site visits. All sites that were intended to be sampled twice in 2015 are sites that were sampled in 2010 and in most cases were sampled twice in 2010. Approximately 50% of the sites were sampled in NCCA 2010 and re-sampled in 2015 and 50% were new sites.

Table 3.4 Site Selection Summary by State and Type of Site for Great Lakes Survey. Number of nearshore sites by state for base sample:

	Number of Unique Sites			Number of Unique Sites	Number of Site visits	Oversample Sites		Total number of Sites Available
State	2010 Sites Sampled Twice in 2015 (Base10_RVT2)	2010 Sites Sampled Once in 2015 (Base10)	New Sites for 2015 (Base15)			2010 Over Sample Sites (Base10_Over Samp)	2015 Over Sample Sites (Base15_Over Samp)	
IL	0	0	0	0	0	0	1	1
IN	0	1	2	3	3	3	1	7
MI	11	45	55	111	122	52	52	215
MN	2	1	3	6	8	4	5	15
NY	7	21	28	56	63	27	29	112
OH	2	10	14	26	28	13	13	52
PA	0	1	2	3	3	3	1	7
WI	3	6	11	20	23	21	13	54
Sum	**25**	**85**	**115**	**225**	**250**	**123**	**115**	**463**

Table 3.5 Nearshore Site Selection Distribution by Great Lake

Great Lake	# Base Sites	# Revisit Sites	Total Site Visits
Lake Superior	45	5	50
Lake Huron	45	5	50
Lake Michigan	45	5	50
Lake Erie	45	5	50
Lake Ontario	45	5	50
Sum	**225**	**25**	**250**

3.3.8 *Site Usage and Replacement*

When a "base" site could not be sampled for any reason, the site was replaced using the following rules:

1. Base10_RVT2_FT: When a site in this category could not be sampled it was replaced by the next available site in the Base10_OverSamp list within the same Great Lake and state (where sites are in SITEID_15 order within the Great Lake and state) and the replacement site should be sampled twice in 2015. The oversample site was sampled for the fish tissue study.
2. Base10_FT: When a site in this category could not be sampled it was replaced by the next available site in the Base10_OverSamp list within the same Great Lake and state (where sites are in SITEID_15 order within the Great Lake and state). The oversample site was sampled for the fish tissue study
3. Base10: When a site in this category could not be sampled it was replaced by the next available site in the Base10_OverSamp list within the same Great Lake and state (where sites are in SITEID_15 order within the Great Lake and state).
4. Base15_RVT2: When a site in this category could not be sampled it was replaced by the next available site in the Base10_OverSamp list within the same Great Lake (where sites are in SITEID_15 order within the Great Lake)
5. Base15_FT: When a site in this category could not be sampled it was replaced by the next available site in the Base15_OverSamp list within the same Great Lake and state (where sites are in SITEID_15 order within the Great Lake and state). The oversample site was sampled for the fish tissue study
6. Base15: When a site in this category could not be sampled it was replaced by the next available site in the Base15_OverSamp list within the same Great Lake and state (where sites are in SITEID_15 order within the Great Lake and state)

3.4 GREAT LAKES EMBAYMENT DESIGN

3.4.1 *Embayment Target Population*

The target population was embayments within the nearshore waters of the Great Lakes of the United States.

3.4.2 *Embayment Sample Frame*

Embayments were defined as indentations of the shoreline for which the width from a line across the opening of the indentation to the furthest inland point is greater than the width of the opening and having an area at least as large as that of a semicircle with a diameter equivalent to the width of the opening (Kelly et al., 2015).

3.4.3 *Embayment Survey Design*

The survey design consisted of two independent designs. Both designs used a Generalized Random

Tessellation Stratified (GRTS) survey design for an area resource. One design re-sampled sites sampled during NCCA 2010 Great Lakes embayment assessment. The other design selected additional new sites using the same survey design used for NCCA 2010.

3.4.4 *Stratification*

A single Great Lake embayment stratum was used.

3.4.5 *Unequal Probability Categories*

Both designs used unequal probability categories. These unequal probability categories were based on area of embayments. These categories are represented as (x,y] where the parenthesis indicates that x is not included, and the bracket indicates that y is included. For example, for 2010 the categories were (0,50], (50,75] and (75,100] where the area is in square kilometers. For 2015 the categories were (0,20], (20,30] and (30,40] where the area is in square miles and were selected to approximately match the 2010 categories. The latter two categories identify two large embayments while the first category includes the remaining embayments.

3.4.6 *Panels*

The combined designs had the following panels:

1. Base10_RVT2: Sites from NCCA 2010 that were re-sampled twice in 2015
2. Base10: Sites from NCCA 2010 that were re-sampled once in 2015
3. Base15: New sites that were sampled once in 2015
4. Base10_OverSamp: Sites from NCCA 210 that were oversample sites that were only used if any Base10_RVT2 or Base10 sites could not be sampled in 2015
5. Base15_OverSamp: New sites that were oversample sites that were only used if any Base15 site could not be sampled in 2015

3.4.7 *Expected Embayment Sample Size*

The Embayment design expected sample size was 150 sites. Fourteen sites from 2010 Embayment assessment were sampled twice in 2015. Fifty-four sites from 2010 Embayment assessment were sampled once in 2015. Sixty-eight new sites were sampled in 2015. This resulted in 136 unique sites (**Table 3.6**).

Table 3.6 Embayment Site Selection Summary by State and Type of Site

Number of Embayment Sites by state								
	Number of Unique Sites			Number of Unique Sites	Number of Site visits	Oversample Sites		Total number of Available Sites
State	Base10_ RVT2	Base 10	Base 15			Base10_ OverSamp	Base15_ OverSamp	
IL	0	0	1	1	1	1	0	2
IN	0	1	1	2	2	3	1	6
MI	10	33	39	82	92	48	42	172
MN	0	2	3	5	5	0	2	7
NY	1	7	7	15	16	11	9	35
OH	0	2	3	5	5	2	4	11
PA	1	1	2	4	5	3	1	8
WI	2	8	12	22	24	15	9	46
Sum	**14**	**54**	**68**	**136**	**150**	**83**	**68**	**287**

3.5 EVALUATION PROCESS

To achieve the planned sample size, sites that could not be sampled were replaced with sites from oversample panels as described in Section 3.4.6. Evaluation Status (EvalStatus) was initially set to Not Evaluated (NotEval) to indicate that the site had yet to be evaluated for sampling. When a site was evaluated for sampling, then the EvalStatus for the site was changed to indicate it was sampleable or, if unsampleable, indicated using a category as listed in **Table 3.7**. Figure 3.1 shows the questions addressed during the site evaluation process and acceptable answers. For NCCA 2015, 1,171 design sites were evaluated (799 in estuaries and 372 in the Great Lakes). Of these 1,060 were classified as target (see 3.2.1, 3.3.1, and 3.4.1 for definitions of target waters) and sampled (699 in estuaries and 361 in the Great Lakes), with 106 sites sampled twice (67 in estuaries and 39 in the Great Lakes). The remaining 111 sites were dropped and replaced for various reasons (**Table 3.7**).

Question 1: Does the site meet the requirements of a target site?

1. Yes, Target
2. Maybe, requires on-site evaluation
3. Maybe, tide too low (return at appropriate time in tidal cycle)
4. Maybe, mudflat at certain times (return at appropriate time in tidal cycle)
5. Unable to access site, but clearly is target (e.g., in shipping channel)
6. Unable to access site, but probably target (e.g., site map indicates target)
7. Unable to access site, and unable to determine if target
8. No, Dry
9. No, Mudflat (permanent)
10. No, Wetland
11. No, Great Lakes site is outside of an embayment, greater than 30 m deep, or greater than 5 km from shore.
12. No, Marine site has salinity <0.5 PPT (freshwater is out of scope except within Great Lakes)
13. No, Map Error (X-site is clearly not target, for example: parking lot)
14. No, Other (explain in comments)

Question 2: Is the site accessible and safe to sample?

Note that responses to the second question reference whether the site would be sampleable if landowner permission is granted.

1. Yes, Sampleable
2. Maybe, Temporarily inaccessible (try again later)
3. Maybe, Unable to access site; available sources are insufficient to determine if target
4. No, Equipment related unsampleable (e.g., less than 1 meter in depth).
5. No, Permanently inaccessible (unable/unsafe to reach site)
6. No, EPA concurred that site could be dropped because access would require extreme efforts

Question 3: Has landowner granted permission to access the site?

1. N/A, public access available
2. Yes, Landowner granted permission
3. No, Landowner denied permission

Figure 3.1 Site Evaluation Questions

Table 3.7 Evaluation Status of Dropped Sites

Evaluation Category	Reason for Dropping	Number of sites dropped	
		Estuaries	Great Lakes
Target vs. NonTarget	Depth_Too_Deep	1	2
Target vs. NonTarget	Depth_Too_Shallow	12	1
Target vs. NonTarget	Map_Error	7	1
Accessibility and Safety	No_Access	18	4
Target vs. NonTarget	NonTarget_Other	32	0
Target vs. NonTarget	Target_Other	4	0
Target vs. NonTarget	Target_Presumed	2	0
Accessibility and Safet	Unsafe	22	0
Target vs. NonTarget	Wetland	2	3
	Total Dropped Sites	100	11

3.6 STATISTICAL ANALYSIS

Statistical analysis of the data must incorporate information about the monitoring survey design. The survey weights in the design file assumed that the survey was implemented as designed. At the end of the sampling season, EPA statisticians adjusted the weights to account for changes due to dropping and replacing sites. This weight adjustment process required the statisticians to know which sites were sampled, which sites were dropped, and if they were dropped, the reason why (Section 3.5). The NCCA statistical analyses, which were completed using the R package spsurvey (Kincaid, et al., 2019), accounted for the site weights that are based upon stratifications and unequal probability selection in the design. The weights are equivalent to the area of the system represented by each site. Procedures for developing the survey design are available from the Aquatic Resource Monitoring Web page (https://archive.epa.gov/nheerl/arm/web/html/index.html). A statistical analysis library of functions to do common population estimates in the statistical software environment R is also available from the webpage. In the NCCA 2015 Site Information data file (available to download from https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys), the adjusted weights used to calculate national condition estimates are in the column "WGT_SP".

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4 BENTHIC MACROINVERTEBRATES

4.1 OVERVIEW

The NCCA estimates biological condition by assessing the condition of estuarine and Great Lakes benthic communities.

For estuarine sites in 2015, the NCCA adopted the multivariate AMBI (M-AMBI; Pelletier et al., 2018). M-AMBI is a new national benthic index that is applicable to estuarine sites across the country and improves upon the assessment of low salinity environments. M-AMBI was also used to recalculate biological condition estimates for data collected in the 2005-2006 and 2010 surveys in order to evaluate change in benthic condition between surveys. The M-AMBI integrates three metrics of environmental condition: AMBI (AZTI Marine Biotic Index; Borja et al. 2000), the Shannon Wiener diversity index, and species richness[5]. AMBI is an abundance-weighted, tolerance value index that assesses habitat condition based upon the relative abundance of taxa in different tolerance value groups, similar in concept to the Hilsenhoff Biotic Index (Hilsenhoff 1977) or the Southern California Benthic Response Index (Smith et al. 2001). M-AMBI uses factor analysis to combine the three metrics of environmental condition into a single index value. Index values range from 0 to 1 with lower scores indicating degraded conditions and higher scores indicating good conditions. M-AMBI is designed to reflect changes in benthic community diversity and the abundance of pollution-tolerant and pollution-sensitive species. Good sites have a wide variety of species, including low proportions of pollution-tolerant species and high proportions of pollution-sensitive species, while poor sites are less diverse and are populated by more pollution-tolerant species and fewer pollution-sensitive species.

In the Great Lakes, the NCCA assesses benthic community condition using an oligochaete trophic index (OTI) that is used by State of the Lakes Ecosystem Conference (SOLEC 2007; ECCC and USEPA 2017). It is based on Howmiller and Scott's (1977) index with subsequent modifications by Milbrink (1983) and Lauritsen et al. (1985). The OTI is a weighted index based on the classification of oligochaete species by their known tolerance to organic enrichment (Environment Canada and USEPA 2014; ECCC and USEPA 2017). OTI scores range from 0 to 3 with lower scores indicating oligotrophic conditions and higher scores indicating eutrophic conditions. In the NCCA 2015 report, oligotrophic equates to good condition, and eutrophic equates to poor condition.

[5] In tidal freshwater habitat, percent oligochaetes is substituted for species richness in the calculation of M-AMBI.

SUMMARY OF BIOLOGICAL QUALITY COMPONENTS

BENTHIC INDEX:

Estuarine

- Multivariate AMBI (M-AMBI):
  - AMBI
  - Shannon diversity (H')
  - Species richness (or % oligochaetes in tidal freshwater habitat)*

Great Lakes

- Oligochaete trophic index (OTI)

Figure 4.1 Summary of indices used to estimate biological quality

4.2 FIELD COLLECTION

Sediment samples were collected using different sediment grab apparatus, as shown in **Table 4.1**. Crews sieved the sediment through a 0.5 mm screen, retaining macroinvertebrates, which were preserved and distributed to laboratories for identification (to the lowest practical taxonomic level) and enumerated.

Table 4.1 Sediment grab sampler type, surface area and location used

Grab type	Grab area (m^2)	Location
Small van Veen or Young-modified van Veen	0.04	CT, DE, FL, GA, LA, MA, MD, NC, NH, NJ, NY, RI, VA
Large van Veen	0.1	CA, ME, OR, WA
Standard Ponar	0.052	AL, IN, IL, MI, MN, MS, NC, NY, OH, PA, RI, SC, TX, WA, WI
Petite Ponar	0.023*	FL, TX, VA
Ekman Grab	0.02*	TX
Diver-collected	0.063	FL
6-inch core	0.0182*	FL
*For grab areas < 0.03 m^2, multiple grabs were composited.		

4.3 DATA PREPARATION

Because state crews used various grab apparatus to collect sediment samples (**Table 4.1**), it was necessary to standardize the raw count of organism abundance by grab area for each sample. Standardization for both estuarine and Great Lakes samples used the following formula:

$$\boldsymbol{Abundance/m^2 = \frac{Abundance/grab}{grab\ area * number\ of\ grabs}} \qquad \textbf{Equation 4-1}$$

4.4 DATA ANALYSIS

4.4.1 *Estuarine Samples*

Estuarine benthic index scores are based on the expectations of best and worst condition in distinct salinity zones (**Table 4.2**). Bottom water salinity measurements (from the hydrographic profile data) were merged with the estuarine benthos dataset. If salinity data were not available for a sample, M-AMBI could not be calculated and the sample was designated as 'Not Assessed.' All taxa in the dataset were also matched with M-AMBI tolerance values, hereafter referred to as Ecological Groups (EG; Gillett et al. 2015; Appendix A https://ars.els-cdn.com/content/image/1-s2.0-S1470160X14005287-mmc1.xlsx). For those species without an EG classification, the genus EG classification, if available, was applied.).

Table 4.2 M-AMBI salinity zones

Salinity zone	Salinity range (ppt)
Tidal freshwater	< 0.5
Oligohaline	≥ 0.5 and < 5
Mesohaline	≥ 5 and < 18
Polyhaline	≥ 18 and < 30
Euhaline	≥ 30 and < 40
Hyperhaline	≥ 40

4.4.1.1 *Multivariate-AMBI (M-AMBI) Index calculations*

First, standard benthic community metrics, including total abundance, Shannon Wiener diversity (H') and species richness (the number of unique species)[6], were calculated for each sample.

Species diversity (H') was calculated as follows:

$$H' = \sum p_i * \ln(p_i) \quad \textbf{Equation 4-2}$$

where

$$p_i = n_i/N \quad \textbf{Equation 4-3}$$

where n is the number of individuals of a given species, i, and N is the total number of species.

Next, the percentage (P) of taxa in each EG was calculated. AMBI was calculated as follows:

$$AMBI = 0 * P_{EG\,I} + 1.5 * P_{EG\,II} + 3 * P_{EG\,III} + 4.5 * P_{EG\,IV} + 6 * P_{EG\,V} \quad \textbf{Equation 4-4}$$

The percentage of uncategorized (i.e., organisms that did not correspond to established EGs) was also calculated for each sample. If the value of uncategorized taxa exceeded 50%, AMBI (and M-AMBI) were not calculated and the sample was designated as 'Not Assessed.'

The above metrics were compiled into a .csv file for input into R (R Core Team, 2017). The M-AMBI factor analysis based on benchmarks in **Table 4.3** was calculated using R scripts from Sigovini et al. (2013). Reference (High) and highly degraded (Bad) anchor points for each salinity zone/grab size (**Table 4.3)** are included in the factor analysis and used to create a pollution gradient (**Figure 4.1**). The Bad benchmark was the worst possible value for that metric (e.g. AMBI score of 6, diversity score of 0). The High benchmark was based on the 95th percentile of the data for a metric that was higher at unimpacted sites (richness, diversity), and the 5th percentile for a metric that was higher at impacted sites (AMBI, % oligochaetes[7]). The station values from factor analysis are projected onto the pollution gradient in Euclidean space (**Figure 4.1**), producing the index score of the sample (Muxika et al. 2007). Because the factor analysis is calculated separately based on habitat and grab size, it allows for the interpretation of benthic samples relative to local-specific expectations of condition (Pelletier et al. 2018).

[6] In the tidal freshwater habitat, percent oligochaetes, the number of oligochaetes divided by the total number of organisms in the sample multiplied by 100, was substituted for species richness in the calculation of M-AMBI.

[7] Based upon data from the 1999 through 2006 National Coastal Assessment.

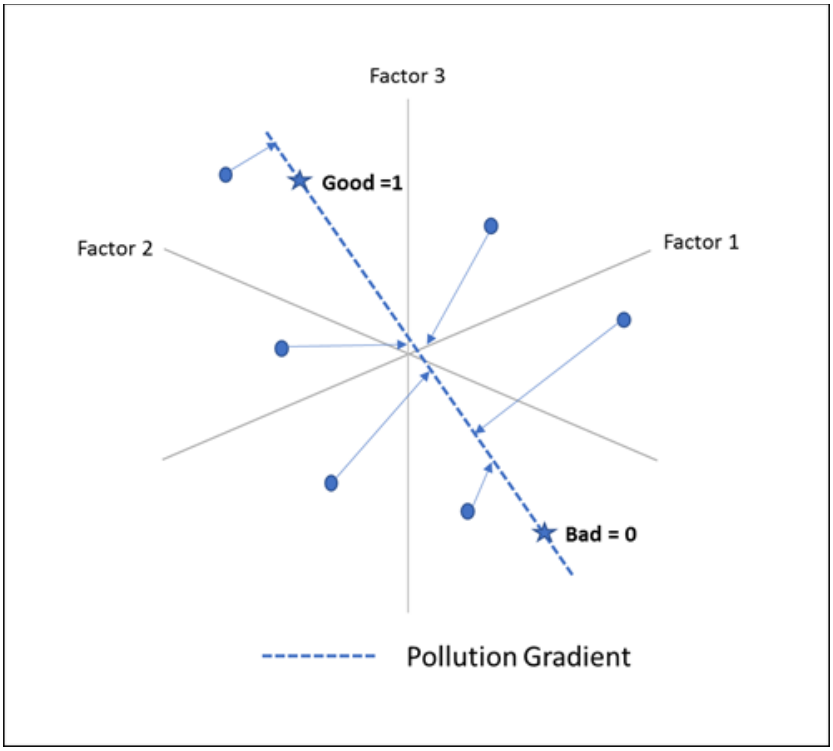

Figure 4.1 Points from M-AMBI factor analysis for each site are projected onto a pollution gradient based upon reference (High) and degraded (Bad) anchor points to obtain an M-AMBI Score ranging from 0 to 1 for each site.

Table 4.3 Reference (High) and degraded (Bad) benchmarks for each salinity zone/grab area combination used in factor analysis to calculate M-AMBI scores.

Salinity Zone	Final grab area (m^2)	Scale		AMBI	Species Richness	Diversity (H')	Percent oligochaetes
All	All	Bad		6	0	0	100
Tidal Freshwater	All	High		0.15		1.93	0
Oligohaline	All	High		0.53	16	2.12	
Mesohaline	All	High		0.85	26	2.48	
Polyhaline	0.03-0.06	High		0.72	44	2.96	
Polyhaline	0.08-0.10	High		0.18	77	3.30	
Euhaline	0.03-0.06	High		0.56	61	3.29	
Euhaline	0.08-0.10	High		0.66	92	3.62	
Hyperhaline	All	High		0.32	55	3.45	

The M-AMBI score output files for each salinity/grab area group were imported into Excel, and samples were designated as Good, Fair or Poor based on M-AMBI values based on Borja et al. 2007 and Borja et al. 2012; **Table 4.4**). These benchmarks were developed and refined through an extensive process by European Water Framework Directive intercalibration exercises in order to provide consistent and accurate condition assessment. For NCCA, M-AMBI index benchmarks were assessed for classification accuracy based on sediment contaminant data, amphipod toxicity, total organic carbon, and dissolved oxygen concentrations from regional validation datasets (see Pelletier et al. 2018 for more details).

Table 4.4 Benchmarks for NCCA estuarine benthic index (M-AMBI)

Benthic Index Condition	
Condition	**Estuarine**
Good	M-AMBI ≥ 0.53
Fair	M-AMBI ≥ 0.39 and < 0.53
Poor	M-AMBI < 0.39

Variance in the M-AMBI results was evaluated by calculating the signal to noise ratio as described in Section 2.3.5 and resulted in S:N of 2.970. In addition, the contingency table for the good, fair and poor rating derived from the M-AMBI scores showed agreement between visits 1 and 2 in 35 of 52 sites. See **Table 4.5**

Table 4.5 M-AMBI contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	28	3	1
	Fair	9	6	1
	Poor	1	2	1

4.4.2 ***Great Lakes Samples***

4.4.2.1 *Oligochaete Trophic Index (OTI) Calculations*

For Great Lakes samples, benthic community condition was assessed using the oligochaete trophic index (OTI). All oligochaetes were classified into five groups – the four classes listed below (**Table 4.6**) and those that were unidentified.

Table 4.6 Trophic classifications of oligochaete species[1]

Group 0	Group 1	Group 2	Group 3
Limnodrilus profundicola	*Arcteonais lomondi*[2]	*Aulodrilus pluriseta*	*Limnodrilus hoffmeisteri*
Lumbriculidae[3]	*Aulodrilus americanus*	*Limnodrilus angustipenis*	*Tubifex tubifex*[4]
Rhyacodrilus coccineus	*Aulodrilus limnobius*	*Limnodrilus cervix*	
Rhyacodrilus montana	*Aulodrilus pigueti*	*Limnodrilus claparedianus*	
Rhyacodrilus sp.	*Dero digitata*[2]	*Limnodrilus maumeensis*	
Spirosperma nikolskyi	*Ilyodrilus templetoni*	*Limnodrilus udekemianus*	
Stylodrilus heringianus	*Isochaetides freyi*	*Potamothrix bedoti*	
Trasserkidrilus superiorensis	*Slavina appendiculata*[2]	*Potamothrix moldaviensis*	
Trasserkidrilus americanus	*Spirosperma ferox*	*Potamothrix vejdovskyi*	
Tubifex tubifex[4]	*Uncinais uncinata*[2]	*Quistadrilus multisetosus*	

[1]Based on Environment Canada and the U.S. Environmental Protection Agency (2014). Only species in the families *Naididae* and *Lumbriculidae* are included.

[2]Species added due to taxonomic reclassification

[3]All immature *Lumbriculidae* were classified by SOLEC as *Stylodrilus heringianus*, so all *Lumbriculidae* were classified as Group 0.

[4]*Tubifex tubifex* was assigned to Group 0 or Group 3 according to the relative abundance of Groups 0 and 3, or the value of c.

The abundance of oligochaete species in each group was calculated for each site, and the OTI was calculated as:

$$OTI = c * \frac{\frac{1}{2}\sum n_0 + \sum n_1 + 2\sum n_2 + 3\sum n_3}{\sum n_0 + \sum n_1 + \sum n_2 + \sum n_3} \qquad \textbf{Equation 4-5}$$

where n_0, n_1, n_2, n_3 refer to the total abundance of species in Group 0, 1, 2, 3, respectively, and c adjusts the ratio to the total abundance of tubificid and lumbriculid oligochaetes (n = number per m^2) as follows:

c = 1	when	$n \geq 3600$
c = 0.75	when	$1200 \leq n < 3600$
c = 0.5	when	$400 \leq n < 1200$
c = 0.25	when	$130 \leq n < 400$
c = 0	when	$n < 130$

Tubifex tubifex was assigned to Group 0 or Group 3 according to the following rules:

if $n_0 / n_3 < 0.75$	then classified as Group 3
if $n_0 / n_3 > 1.25$	then classified as Group 0;
if $n_0 / n_3 = 0.75 – 1.25$	
and if $c < 0.5$,	then classified as Group 0,
or if $c \geq 0.5$	then classified as Group 3;
if $n_0 / n_3 = 0$	
and if n_0 is relatively high and/or c is low[8],	then classified as Group 0, otherwise classified as Group 3.

The OTI values were classified into Good, Fair, and Poor categories based on benchmarks developed and validated in Milbrink 1983 and adopted for the State of the Great Lakes reporting (SOLEC 2007; ECCC and USEPA 2017; See **Table 4.7**).

Table 4.7 Benchmarks for NCCA Great Lakes benthic index (OTI)

Benthic Index Condition	
Condition	**Great Lakes**
Good	OTI < 0.6
Fair	OTI ≥ 0.6 and ≤ 1
Poor	OTI > 1

Variance in the OTI was evaluated by calculating the signal to noise ratio as described in Section 3.4.5 and resulted in S:N of 4.420. In addition, the contingency table illustrates agreement among 18 good, fair and poor ratings between the first and second visits at 24 revisit sites (See **Table 4.8**)

Table 4.8 OTI contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	6	2	1
	Fair	1	3	1
	Poor	1		9

[8] Note that 'relatively high' n_0 was operationally defined as greater than the average of Group 0 abundance, and 'low' c was defined as 0.25.

4.5 REFERENCES

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5 EUTROPHICATION INDEX

5.1 BACKGROUND

NCCA 2015 used the same approaches that were used in previous surveys to calculate the estuarine and Great Lakes Eutrophication Indices[9]. At both estuarine and Great Lakes sites, surface nutrients, surface chlorophyll-*a* (CHLA), bottom water dissolved oxygen (DO) and water clarity were measured. However, the specific nutrient parameters and water clarity metrics that were integrated into the overall Eutrophication Index were different between estuarine and Great Lakes sites (See **Figure 5.1**). In addition to the nutrient parameters contributing to the index, the NCCA has also adopted surface total nitrogen (TN) and total phosphorus (TP) as measures of nutrient enrichment in estuaries and the Great Lakes.

SUMMARY OF EUTROPHICATION INDEX COMPONENTS

Estuarine

- Surface dissolved inorganic phosphorus, PO_4 (DIP)
- Surface dissolved inorganic nitrogen (DIN; NO_3+NO_2,+NH_4)
- Surface chlorophyll-*a* (CHLA)
- Bottom water dissolved oxygen (DO)
- Water clarity as transmittance at 1m calculated from Photosynthetically Active Radiation (PAR)*

Great Lakes

- Surface total phosphorus (TP)
- Surface chlorophyll *a* (CHLA)
- Bottom water dissolved oxygen (DO)
- Water clarity as Secchi depth*

ADDITIONAL METRICS: TOTAL NUTRIENTS QUARTILE RANKINGS**

Estuarine

- Surface total nitrogen (TN)
- Surface total phosphorus (TP)

Great Lakes

- Surface total nitrogen (TN)
- Surface total phosphorus (TP)

*Secchi depth was used at some estuarine sites if PAR measurements were missing; PAR measurements were used at some Great Lakes sites if Secchi depth was missing.

**Quartile rankings are based on total nutrients quartiles from 2010 and are included in the online data dashboard (https://coastalcondition.epa.gov/)

Figure 5.1 Summary of eutrophication index components

[9] Formerly called the "Water Quality Index". In 2015, the name was changed to "Eutrophication Index" to reflect the focus on eutrophication related parameters and not other "water quality" parameters such as pathogens, contaminants, pH or temperature.

5.2 FIELD COLLECTION

Field crews used similar methods to collect water samples in estuaries and the Great Lakes. Dissolved oxygen (DO) downcast and upcast profile data were collected at regular intervals from the surface to 0.5 meters from the bottom, using a calibrated multi-parameter water quality meter (or sonde). The DO value used for the Eutrophication Index is the average of the downcast and upcast values measured 0.5 m above the bottom sediment. Water clarity was measured both with a 20 cm Secchi disk and a Photosynthetically Active Radiation (PAR) meter. Water samples were collected 0.5 m below the surface using either a pumped system or a water sampling bottle such as a Niskin, Van Dorn, or Kemmerer bottle and then transferred to a rinsed 250 mL amber Nalgene bottle for total nutrients and a 2 L amber HDPE bottle for chlorophyll-*a* and dissolved nutrients. The CHLA and dissolved nutrient sample was filtered using a Whatman GF/F 47 mm 0.7-micron filter. The filter was analyzed for CHLA content and the filtrate was used for dissolved nutrients analyses. Refer to the NCCA 2015 Field Operations Manual for detailed descriptions of sample collection and analysis protocols (USEPA 2015a).

5.3 LABORATORY METHODS

Eutrophication Index parameters were analyzed using methods that are the same as or equivalent to those listed in **Table 5.1**. Refer to the NCCA 2015 Laboratory Operations Manual for detailed descriptions of sample analysis protocols (USEPA 2015b).

Table 5.1 Laboratory methods for water chemistry analyses

Dataset	Parameter	Symbol	Method*
Estuarine	Chlorophyll-*a*	CHLA	EPA 445.0 †
	Total phosphorus	TP	APHA Standard Method 4500-P.E
	Total nitrogen	TN	APHA Standard Method 4500-N.C
	Dissolved inorganic nitrogen	DIN	(calculated)
	Ammonia	NH_3	EPA 350.1 †
	Nitrate	NO_3	EPA 353.2
	Nitrite	NO_2	EPA 353.2; USGS I2540-85
	Nitrate + Nitrite	NO_2NO_3	EPA 353.2 ‡; ASTM 7781
	Dissolved Inorganic phosphorus; Orthophosphate	DIP	APHA Standard Method 4500-P.E (filtered before analysis)
Great Lakes	Chlorophyll-*a*	CHLA	EPA 446.0 †
	Total phosphorus	TP	APHA Standard Method 4500-P.E
	Total nitrogen	TN	APHA Standard Method 4500-N.C

*Multiple state and federal laboratories participate in the survey. Any acceptable method that meets the required data quality objectives may be used. See dataset for exact method used for any specific sample.

† EPA's Willamette Research Station (WRS) Laboratory has modified some procedures to lower the method detection limits for samples sent to the national lab.

‡ For estuarine samples, EPA's WRS Lab analyzed nitrate and nitrite together and modified EPA Method 353.2 to be performed on a Flow Injection Analyzer.

5.4 INDEX CALCULATION

5.4.1 *Estuarine Sites*

Five metrics contributed to the estuarine Eutrophication Index: surface DIN, DIP, and CHLA concentrations; bottom water DO; and water clarity (% transmittance at 1 m). Note that DIN is a derived parameter, calculated as the sum of nitrate (NO_3), nitrite (NO_2), and ammonium (NH_4) concentrations. Some labs reported nitrate (NO_3) and nitrite (NO_2) concentrations separately; others reported these analytes as the sum of nitrate and nitrite ($NO_3 + NO_2$).

DIN, DIP, and CHLA concentrations were evaluated as good, fair or poor relative to benchmarks listed in **Table 5.2 - Table 5.4**. The benchmarks were set according to NCCA reporting regions:

- Northeast: Coasts of Maine through Virginia
- Southeast: Southern Atlantic seaboard from North Carolina to Florida
- Gulf: Gulf of Mexico from coastline Florida through Texas
- West: Coasts of California, Oregon, and Washington
- Tropics: Florida Bay, Biscayne Bay and waters of the Florida Keys

The nutrient and chlorophyll-*a* benchmarks were established by a consensus of experts including academic scientists, state and federal government scientists, and others after evaluation of literature, best professional judgement, and expert opinions. Information and long-term data were systematically compiled from over 300 regional experts on estuarine eutrophication across the country during the National Estuarine Eutrophication Assessment (Bricker et al. 1999). The benchmarks developed were designed to characterize eutrophication conditions on a national basis. For NCCA, adjustments in benchmark values for different regions were made to account for regional differences in background nutrient concentrations during the NCCA summer index sampling period based on comments from peer reviewers and consultations with state water quality managers. (USEPA 2004; USEPA 2012).

DO was evaluated as good, fair or poor relative to benchmarks listed in **Table 5.5**. DO benchmarks reflect levels that are shown to disrupt estuarine communities (Diaz and Rosenberg 1995; USEPA 2000) and are often used as state regulatory DO limits.

Table 5.2 Estuarine indicator benchmarks for Dissolved Inorganic Phosphorus (DIP)

DIP (mg/L) Condition					
Condition	**Northeast**	**Southeast**	**Gulf**	**West**	**(South Florida)[10]**
Good	≤ 0.01	≤ 0.01	≤ 0.01	≤ 0.07	≤ 0.005
Fair	> 0.01 and ≤ 0.05	> 0.01 and ≤ 0.05	> 0.01 and ≤ 0.05	> 0.07 and ≤ 0.1	> 0.005 and ≤ 0.01
Poor	> 0.05	> 0.05	> 0.05	> 0.1	> 0.01

Table 5.3 Estuarine indicator benchmarks for Dissolved Inorganic Nitrogen (DIN)

DIN (mg/L) Condition					
Condition	**Northeast**	**Southeast**	**Gulf**	**West**	**(South Florida)**
Good	≤ 0.1	≤ 0.1	≤ 0.1	≤ 0.35	≤ 0.05
Fair	> 0.1 and ≤ 0.5	> 0.1 and ≤ 0.5	> 0.1 and ≤ 0.5	> 0.35 and ≤ 0.5	> 0.05 and ≤ 0.1
Poor	> 0.5	> 0.5	> 0.5	> 0.5	> 0.1

Table 5.4 Estuarine indicator benchmarks for Chlorophyll a (CHLA)

CHLA (µg/L) Condition					
Condition	**Northeast**	**Southeast**	**Gulf**	**West**	**(South Florida)**
Good	≤ 5	≤ 5	≤ 5	≤ 5	≤ 0.5
Fair	> 5 and ≤ 20	> 5 and ≤ 20	> 5 and ≤ 20	> 5 and ≤ 20	> 0.5 and ≤ 1
Poor	> 20	> 20	> 20	> 20	> 1

Table 5.5 Estuarine indicator benchmarks for Dissolved Oxygen (DO)

DO (mg/L) Condition	
Condition	**All regions**
Good	> 5
Fair	≤ 5 and > 2
Poor	≤ 2

[10] For the NCCA 2015 Report, "South Florida" benchmarks were used to assess to waters of Florida Bay, Biscayne Bay and the Florida Keys.

Water clarity in estuaries was characterized as transmittance, the percent of photosynthetically active radiation (PAR) transmitted through 1 m of water. PAR attenuation was measured using two PAR sensors: one sensor was lowered through the water column, measuring PAR intensity (I_z) at depths z (m), and a second sensor remained in the air measuring incident PAR intensity (I_o). The normalized PAR attenuation (I_z/I_o) is assumed to follow Beer's law, i.e., light intensity decreases exponentially with distance:

$$\frac{I_z}{I_o} = e^{-K_d * z} \quad \textbf{Equation 5-1}$$

where K_d is the PAR attenuation coefficient; larger K_d magnitudes indicate greater attenuation, or poorer water clarity. Equation 5-1 (**above**) is equivalently expressed as follows, highlighting that intensity $\ln(I_z/I_o)$ is linearly proportional to depth:

$$\ln \frac{I_z}{I_o} = -K_d * z \quad \textbf{Equation 5-2}$$

At each site, K_d is calculated as the negative slope of the regression of $\ln(I_z/I_o)$ on the y-axis vs. depth on the x-axis using the downcast measurements[11]. Once K_d values were calculated, % transmittance at 1 m (i.e. I_z/I_o at $z = 1$) was calculated as:

$$\%\,Trans\ @\ 1\ m = e^{-K_d} * 100 \quad \textbf{Equation 5-3}$$

The water clarity condition at each site (good, fair, or poor) was determined by %Trans @ 1 m values relative to benchmarks in **Table 5.6**. These transmittance benchmarks vary depending on the turbidity level or status of submerged aquatic vegetation (SAV) at each site. Benchmarks for naturally turbid regions allow for reduced clarity, while those for waters supporting SAV on the Atlantic and Gulf Coasts (either naturally occurring or due to ongoing restoration efforts) support a higher degree of clarity.

Regional delineations of turbidity classes (Figure 5.2) for the 2010 and 2015 NCCA reports are described in Smith et al. (2006). Naturally turbid regions consisted of waters in Alabama, Louisiana, Mississippi, South Carolina, Georgia, and Delaware Bay. Regions supporting SAV included Laguna Madre (TX), the entire west coast of Florida, Biscayne Bay (FL), the Indian River lagoon (FL), and portions of Chesapeake Bay (VA). Sites on the West Coast and all other Atlantic and Gulf Coast sites were considered to exhibit normal turbidity. During review, EPA received comments that there are additional areas around the country that should be classified as supporting SAV restoration and should be subject to more stringent water clarity benchmarks. EPA will review turbidity classification and apply updates in future assessments.

[11] In the unlikely event that a downcast value was suspect for any water quality parameter, and there was greater confidence in the upcast value at that corresponding depth, the upcast value may have been substituted for the downcast value.

Table 5.6 Estuarine indicator benchmarks for water clarity

Water Clarity (% Transmittance at 1 m) Condition			
Condition	**Naturally Turbid**	**Normal Turbidity**	**SAV Restoration**
Good	> 10%	> 20%	> 40%
Fair	≤ 10% and > 5%	≤ 20% and > 10%	≤ 40% and > 20%
Poor	≤ 5%	≤ 10%	≤ 20%
c-values*	1	1.4	1.7
***If PAR is not measured, K_d is estimated from Secchi depth using c-values: $K_d = c/\text{Secchi depth}$**			

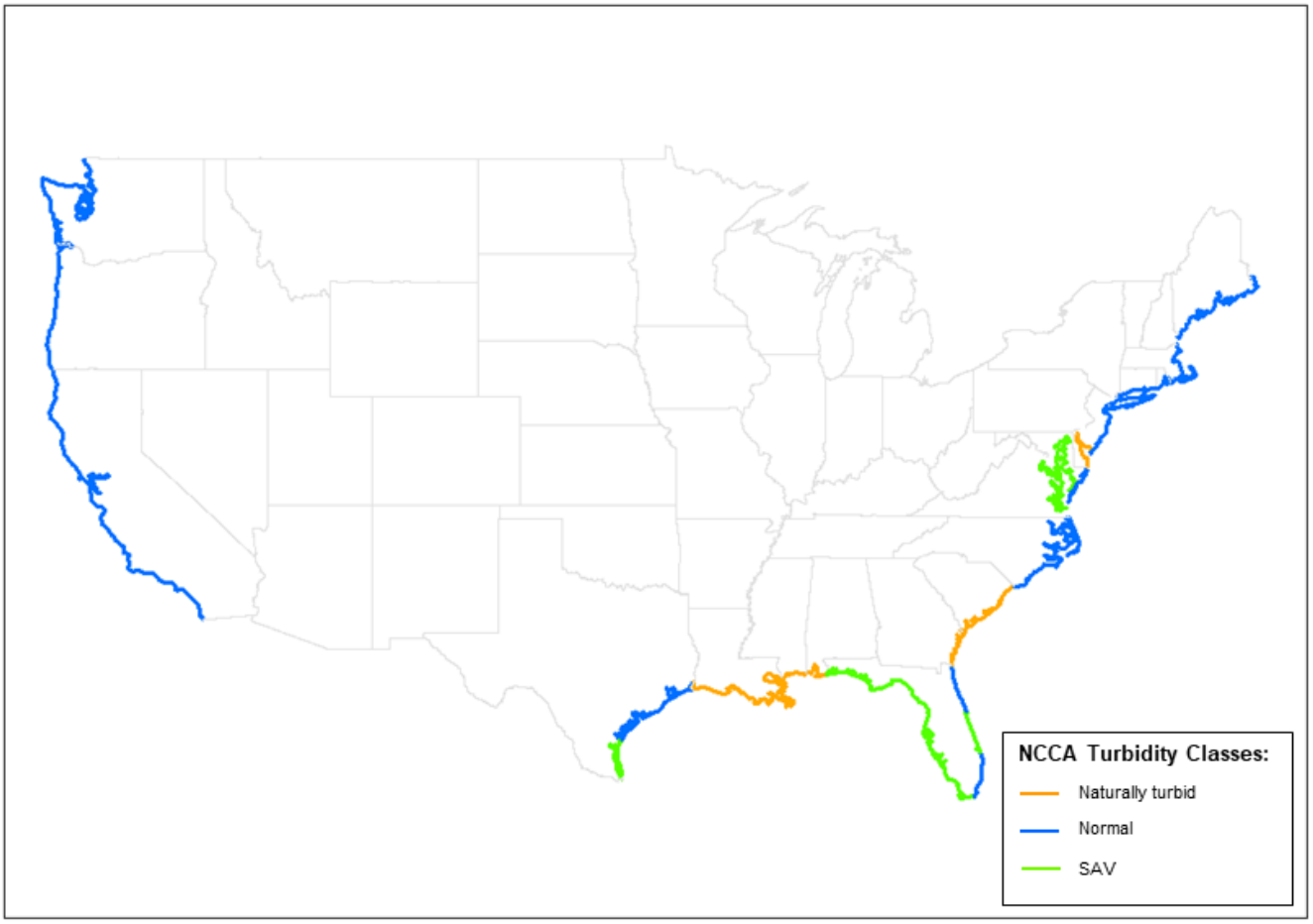

Figure 5.2 Turbidity classes used for water clarity condition rating

In the instance where PAR data were not available for a site, K_d was estimated from Secchi depth as:

$$K_{d(est)} = \frac{c}{Secchi\ depth\ (m)} \qquad \textbf{Equation 5-4}$$

where c is a constant specific to the water type, as indicated in **Table 5.6** (Smith et al. 2006; Poole and

Atkins 1929). If neither PAR data nor Secchi depth were available, the condition at the site was set to "missing".

The Eutrophication Index for each estuarine site was based on the condition of the component metrics, evaluated according to the rules in **Table 5.7**.

Table 5.7 Rules for determining Eutrophication index condition at estuarine sites

Eutrophication Index	
Condition	**Benchmarks**
Good	A maximum of 1 indicator is rated fair; no indicators are rated poor
Fair	1 of the indicators is rated poor; or ≥ 2 indicators are rated fair
Poor	≥ 2 of the 5 indicators are rated poor
Missing	2 indicators are missing, and the available indicators do not suggest a fair/poor rating

Variance in the components used in the estuarine eutrophication index was evaluated by calculating the signal to noise ratio as described in Section 2.3.5 and resulted in S:N of 1.945 for DIN, 7.429 for DIP, 2.486 for CHLA, 2.199 for DO and 5.445 for Transmissivity. In addition, the contingency table for the overall eutrophication index illustrates the agreement of good, fair and poor ratings between the first and second visits at 55 of 67 revisit sites (See **Table 5.8**)

Table 5.8 Eutrophication index contingency table (estuaries)

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	21	3	
	Fair	2	24	5
	Poor		2	10

5.4.1.1 *Total nitrogen (TN) and total phosphorus (TP)*

TN and TP concentrations were characterized as low, moderate, high, or very high based on the 25th, 50th, and 75th percentile TN or TP values at sites in estuaries in the 2010 NCCA survey (**Table 5.9**). Quartile results are reported in the online data dashboard at https://coastalcondition.epa.gov/.

Table 5.9 Estuarine benchmarks for total nutrients derived from 2010 concentrations

Total Nutrients		
Condition	**TN (mg/L)**	**TP (mg/L)**
Low	≤ 0.31	≤ 0.037
Moderate	> 0.31 to ≤ 0.48	> 0.037 to ≤ 0.062
High	> 0.48 to ≤ 0.68	> 0.062 to ≤ 0.101
Very High	> 0.68	> 0.101

5.4.2 *Great Lakes Sites*

Four metrics were employed in assessing Great Lakes water quality: TP and CHLA concentrations in surface water; DO at the bottom; and Secchi depth as a measure of water clarity. TN was measured in the Great Lakes but was not included in the Eutrophication Index because nitrogen has generally not been considered a limiting nutrient in this system so there are no published benchmarks for nitrogen suitable to be used in the Great Lakes.

The International Joint Commission (IJC) Phosphorus Management Strategies Task Force (PMSTF; IJC 1980) developed total phosphorus, chlorophyll a, and Secchi depth benchmarks for each Great Lake and each basin of Lake Erie based on expected trophic status (Figure 5.3). The benchmarks were developed for "open waters", but data used to generate the benchmarks included nearshore samples (Gregor and Rast 1979), so they were considered relevant to the nearshore and embayment sites for the Great Lakes assessment. The PMSTF only identified a single benchmark based on the trophic status for each lake (fair to good), so the lower benchmark (fair to poor) was defined for the NCCA report as the value indicative of crossing into the next more nutrient-enriched trophic status. The NCCA analysts and partners used IJC study results (Gregor and Rast 1979) to identify trophic status benchmarks for selected basins (i.e., Saginaw Bay in Lake Huron and western, central, and eastern basins of Lake Erie), that were not specified in the 1980 PMSTF report. **Table 5.9**- **Table 5.13** list the benchmarks used to evaluate conditions in Great Lakes coastal waters. Note that benchmarks vary by lake and basin. DO benchmarks were the same as those used in estuaries and the 2 mg/L is used to define a hypoxic condition in the Great Lakes (Diaz and Rosenberg 1995; USEPA 2000).

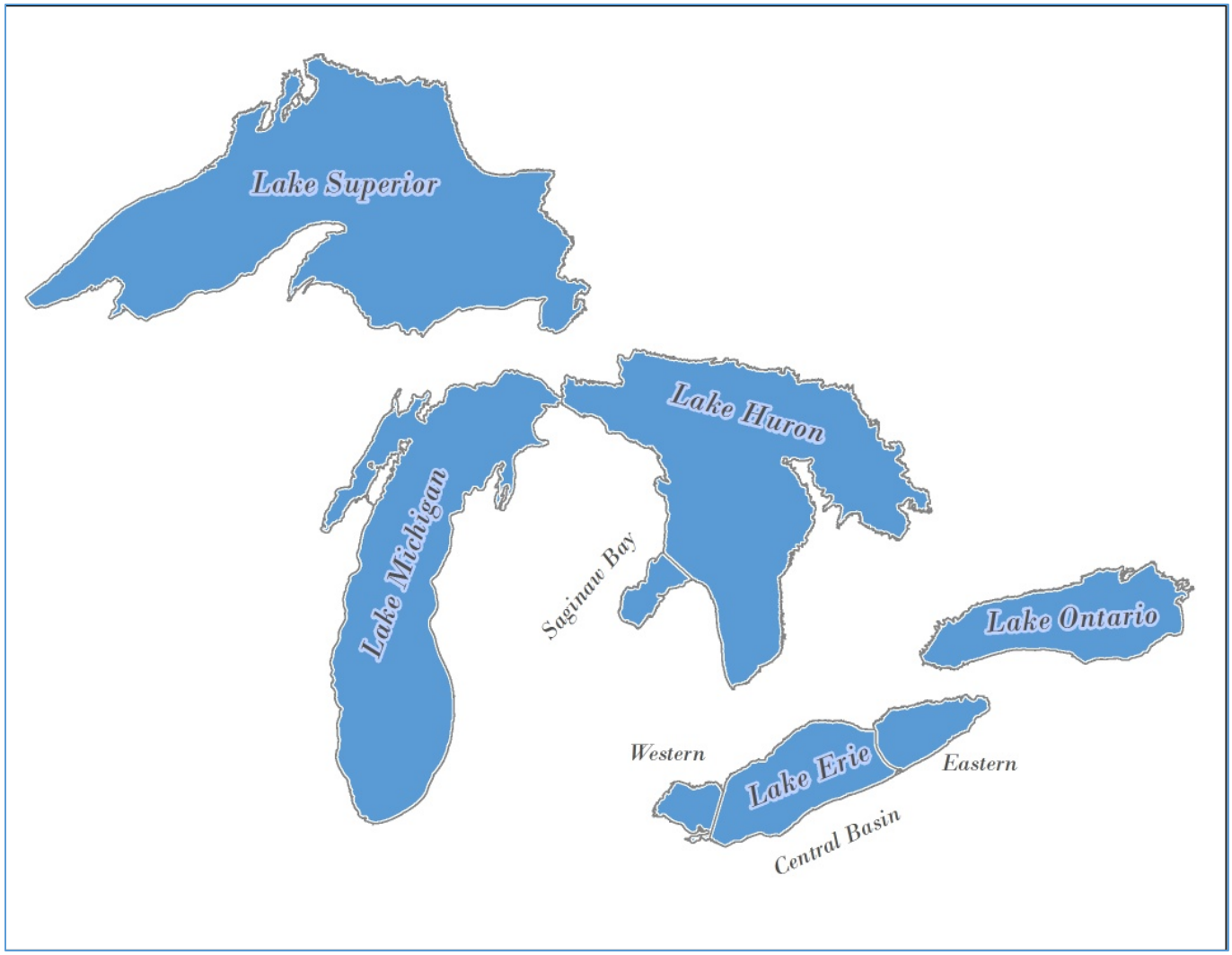

Figure 5.3 Basin boundaries of the Great Lakes

Table 5.10 Great Lakes indicator benchmarks for Total Phosphorus (TP)

TP (µg/L) Condition								
Condition	**Lake Superior**	**Lake Michigan**	**Lake Huron**	**Saginaw Bay**	**Western Lake Erie**	**Central Lake Erie**	**Eastern Lake Erie**	**Lake Ontario**
Good	≤ 5	≤ 7	≤ 5	≤ 15	≤ 15	≤ 10	≤ 10	≤ 10
Fair	> 5 and ≤ 10	> 7 and ≤ 10	> 5 and ≤ 10	> 15 and ≤ 32	> 15 and ≤ 32	> 10 and ≤ 15	> 10 and ≤ 15	> 10 and ≤ 15
Poor	> 10	> 10	> 10	> 32	> 32	> 15	> 15	> 15

Table 5.11 Great Lakes indicator benchmarks for Chlorophyll a (CHLA)

CHLA (µg/L) Condition								
Condition	**Lake Superior**	**Lake Michigan**	**Lake Huron**	**Saginaw Bay**	**Western Lake Erie**	**Central Lake Erie**	**Eastern Lake Erie**	**Lake Ontario**
Good	≤ 1.3	≤ 1.8	≤ 1.3	≤ 3.6	≤ 3.6	≤ 2.6	≤ 2.6	≤ 2.6
Fair	> 1.3 and ≤ 2.6	> 1.8 and ≤ 2.6	> 1.3 and ≤ 2.6	> 3.6 and ≤ 6	> 3.6 and ≤ 6	> 2.6 and ≤ 3.6	> 2.6 and ≤ 3.6	> 2.6 and ≤ 3.6
Poor	> 2.6	> 2.6	> 2.6	> 6	> 6	> 3.6	> 3.6	> 3.6

Table 5.12 Great Lakes indicator benchmarks for Dissolved Oxygen (DO)

DO (mg/L) Condition	
Condition	**All regions**
Good	> 5
Fair	≤ 5 and > 2
Poor	≤ 2

Table 5.13 Great Lakes indicator benchmarks for water clarity

Water Clarity (Secchi depth in m) Condition								
Condition	**Lake Superior**	**Lake Michigan**	**Lake Huron**	**Saginaw Bay**	**Western Lake Erie**	**Central Lake Erie**	**Eastern Lake Erie**	**Lake Ontario**
Good	> 8	> 6.7	> 8	> 3.9	> 3.9	> 5.3	> 5.3	> 5.3
Fair	≤ 8 and > 5.3	≤ 6.7 and > 5.3	≤ 8 and > 5.3	≤ 3.9 and > 2.1	≤ 3.9 and > 2.1	≤ 5.3 and > 3.9	≤ 5.3 and > 3.9	≤ 5.3 and > 3.9
Poor	≤ 5.3	≤ 5.3	≤ 5.3	≤ 2.1	≤ 2.1	≤ 3.9	≤ 3.9	≤ 3.9

Water clarity was characterized in the Great Lakes primarily by Secchi depth, and secondarily by Secchi depth estimated from PAR attenuation at sites lacking Secchi data or at sites where the Secchi disk was visible clear to the bottom. To assign a Secchi depth condition class to a clear-to-bottom site (CTB), site depth was considered first. If site depth was greater than the good/fair Secchi depth benchmark for that waterbody (**Table 5.12**), then the Secchi depth condition class was rated "good". If site depth was less than or equal to the good/fair benchmark, then a condition class could not be unambiguously assigned as fair or poor. At those sites, the missing Secchi depth could be estimated using the site's k_d.

If neither PAR data nor Secchi depth were available and site depth was less than the good/fair benchmark, then Secchi depth and its condition class, were considered "missing".

To predict Secchi depth at clear-to-bottom sites where site depth was less than the good/fair benchmark for the water body and to assign those sites a condition class, a power function was used to model the relationship between Secchi depth and k_d. The Secchi depth – k_d model is derived for each NCCA cycle based on that year's dataset. The relationship between Secchi depth and k_d depends on a combination of site-specific factors like chlorophyll *a*, suspended solids, colored dissolved organic matter (e.g. Brezonik et al. 2019). If these factors change across the Great Lakes over time, this relationship may also change. By basing estimated Secchi depth on a model derived based on that years' data, each NCCA cycle may differ in its k_d value.

For 2015 sites located in the Great Lakes, the Secchi depth estimation model was based on all sites located in the lakes, including multiple site visits and enhancement sites (embayments, Lake Erie enhancement; 298 sites total) and was described as:

$$\boldsymbol{Secchi\ depth_{est} = 1.3891 * k_d^{-0.983}} \qquad \mathbf{r^2 = 0.90} \qquad \textbf{Equation 5-5}$$

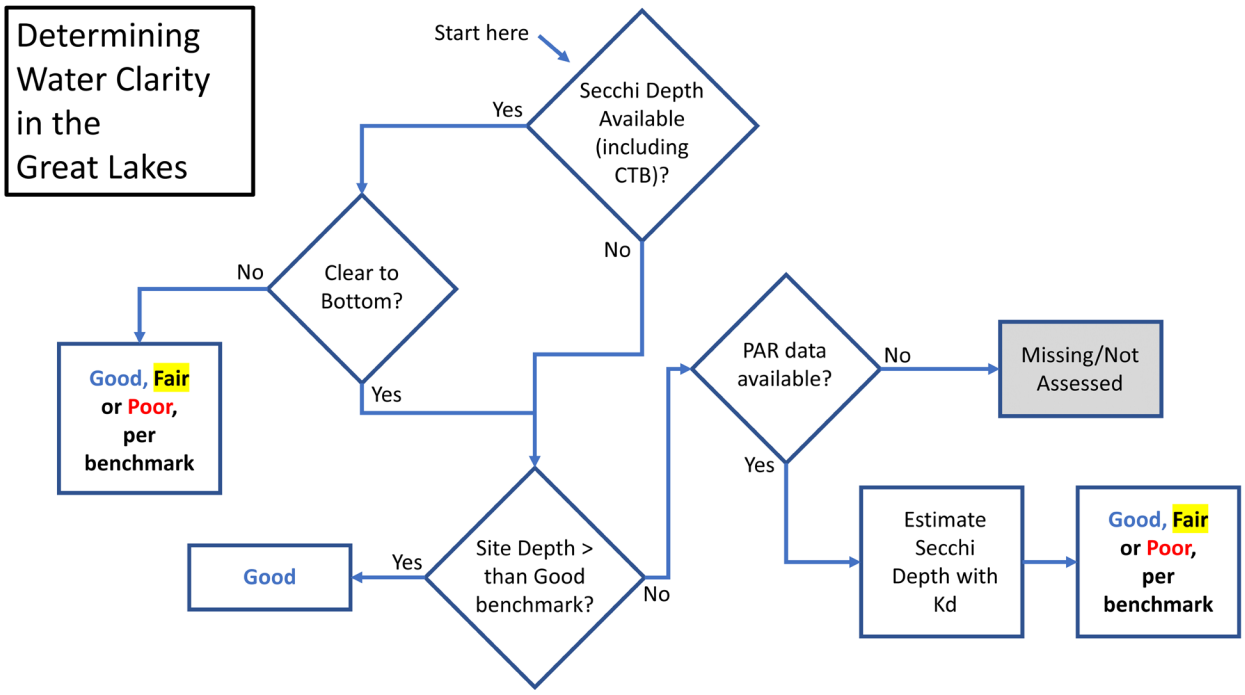

Figure 5.4 Determining Water Clarity in the Great Lakes

The Eutrophication index for Great Lakes sites was then determined based on the condition of the component metrics, evaluated according to the rules in **Table 5.13**.

Table 5.14 Rules for determining Eutrophication Index condition at Great Lakes sites

Eutrophication Index	
Condition	**Benchmarks**
Good	A maximum of 1 indicator is rated fair; no indicators are rated poor
Fair	1 of the indicators is rated poor; or ≥ 2 indicators are rated fair
Poor	≥ 2 of the 5 indicators are rated poor
Missing	2 indicators are missing, and the available indicators do not lead to a fair/poor rating

Variance in the components used in the Great Lakes eutrophication index was evaluated by calculating the signal to noise ratio as described in Section 2.3.5 and resulted in S:N ratios of 1.849[12] for total phosphorus (PTL), 15.375[12] for CHLA, 1.752 for DO and 5.369 for mean Secchi depth. In addition a contingency table for the Great Lakes eutrophication index illustrates the agreement of good, fair and poor ratings between the first and second visits at 39 revisit sites (See **Table 5.15**).

Table 5.15 Great Lakes eutrophication index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	21	2	
	Fair	2	2	4
	Poor			8

5.4.2.1 *Total nitrogen (TN) and total phosphorus (TP)*

TN and TP concentrations are characterized as low, moderate, high, or very high based on the 25th, 50th, and 75th percentile TN or TP values at sites in the Great Lakes in the 2010 NCCA survey (**Table 5.16**). Results are reported in the online data dashboard at https://coastalcondition.epa.gov/.

[12] One second visit sampling event in western Lake Erie occurred during a record-breaking algal bloom. The chlorophyll *a* concentration for that site was extremely high. Due to the unusual nature of this event, the reported S:N ratios for the eutrophication index were calculated twice. S:N ratios reported in text do not include results from the second visit sampled during this algal bloom. Appreciably different S:N ratios when extreme second visit values are included in the calculation are listed below:

- Total phosphorus: -0.469;
- Chlorophyll *a*: 0.713

Table 5.16 Great Lakes quartile-based benchmarks for total nutrients, derived from ranked 2010 concentrations

Total Nutrients		
Condition	**TN (mg/L)**	**TP (mg/L)**
Low	≤ 0.36375	≤ 0.0028
Moderate	> 0.36375 to ≤ 0.4025	> 0.0028 to ≤ 0.00522
High	> 0.4025 to ≤ 0.48	> 0.00522 to ≤ 0.00988375
Very High	> 0.48	> 0.00988375

5.5 REFERENCES

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6 SEDIMENT QUALITY INDEX

6.1 BACKGROUND

The NCCA 2015 used the same approach as the NCCA 2010 for sediment collection, analysis and interpretation. Surficial sediment was collected in the field using a grab sampler and composited to be analyzed for sediment contaminants, sediment toxicity, total organic carbon and grain size. The NCCA assesses sediment for possible adverse effects on the benthic community using a two-component sediment quality index (SQI). The SQI consists of a sediment contaminant index (SCI) and a sediment toxicity index (STI). For NCCA 2015, 677 sediment samples were collected at the estuarine visit one sites and 294 sediment samples were collected at Great Lakes visit one sites.

The SCI uses literature-based sediment quality guidelines (SQG) and is calculated into a quotient. For estuarine sites, the SCI uses both the Effects Range Median (ERM) to calculate the SQG quotient and a logistic regression model. For Great Lakes sites, the SCI uses the Probable Effect Concentration (PEC) to calculate the SQG quotient. For both estuarine and Great Lakes sites, the STI uses the results of a 10-day amphipod toxicity test. The organisms are exposed to collected sediments, capturing responses to a broader range of sediment properties that might contribute to overall toxicity. In the estuarine samples, sediment toxicity tests use the amphipod *Leptocheirus plumulosus*, while Great Lakes tests use the amphipod *Hyalella azteca*. The toxicity indices were primarily based on control-corrected survival (statistical significance was included for estuarine sites only).

SEDIMENT CONTAMINANT INDEX (SCI):

Estuarine

- Mean effects range median quotient (mean ERM-Q)
- Logistic regression model (LRM) maximum probability (P_{max})

Freshwater (*Great Lakes only*)

- Mean probable effects concentration quotient (mean PEC-Q)

SEDIMENT TOXICITY INDEX (STI):

Estuarine

- Control-corrected survival of amphipods (*Leptochirus plumulosus*)
- Statistical significance (one-sided t-test) between control and test survivals

Freshwater (*Great Lakes only*)

- Control-corrected survival of amphipods (*Hyalella azteca*)

Figure 6.1 Summary of sediment quality index components

6.2 FIELD COLLECTION

Field crews collected the top 2 cm of surface sediments at predetermined probabilistic sites as prescribed in the Field Operations Manual (USEPA 2015a). Estuarine crews used assorted stainless steel grab samplers, (e.g., Young-modified Van Veen, ponar or Eckman; See **Table 4.1**), whereas Great Lakes crews used a stainless-steel standard Ponar sampler. Crews composited the surface sediments from multiple grab samples to collect approximately 2 liters of sediment—the total sediment volume required for analysis.

6.3 LABORATORY ANALYSES

6.3.1 *Sediment Contamination*

Samples were analyzed for contaminant concentrations of metals (including mercury), polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and organochlorine pesticides using a variety of spectrometry methods (**Table 6.1** and see LOM (USEPA 2015b)). Total organic carbon, grain size and percent moisture were measured to provide supplementary information but were not included in index calculations and condition assessment.

6.3.2 *Sediment Toxicity*

Sediment toxicity tests were performed to determine the percent survival of laboratory amphipods (estuarine species: *Leptocheirus plumulosus*; freshwater species: *Hyalella azteca*) following 10 days of exposure to sample sediments (**Table 6.2** and see LOM (USEPA 2015b; ASTM 1990; ASTM 2005). Control tests were run in parallel with the sample tests. Control tests were run using reference sediments, organisms from the same batch as used in sample tests and water from the same sources.

6.3.2.1 *Estuarine*

The estuarine test was run using a static water method with 5 replicate chambers and 20 organisms exposed in each chamber. A minimum of 90% survival of control organisms was required to meet test acceptability criteria (USEPA 2001; USEPA 2015b; ASTM 1990).

6.3.2.2 *Freshwater*

The freshwater toxicity test was run using a flow-through method with 4 replicate chambers and 10 organisms in each chamber; a minimum of 80% survival of control organisms was required to meet test acceptability criteria (USEPA 2000; USEPA 2015b; ASTM 2005).

Table 6.1 Laboratory methods for sediment analyses

Dataset	Parameter	Method
Sediment contaminants	Metals (excluding Mercury)	EPA 6020
	Mercury	EPA 245.7
	PAHs, PCBs, pesticides	EPA 8270D
Sediment Toxicity	Marine amphipod 10-day acute toxicity test	Test Method 100.4 in ASTM E1367-03
	Freshwater amphipod 10-day acute toxicity test	Test Method 100.1 in EPA 600/R-99/064
Sediment Characteristics	Total organic carbon	EPA 9060
	Percent solids	SM 2540B
	Grain size	SM 2560

6.4 SEDIMENT CONTAMINANT INDEX CALCULATIONS

Sediment quality guidelines (SQGs) identify concentrations of individual contaminants that may be associated with adverse effects to benthic organisms (Long 2006). While SQGs are adequate for assessing individual contaminant levels in sediments, they do not address combinations of contaminants typically found in a sample. Therefore, the NCCA used mean SQG quotients to produce overall unitless assessments of contamination to predict aggregate toxicity due to multiple contaminants (Fairey et al. 2001; Long et al. 2006). The mean effects-range median quotient (mERM-Q; Long et al. 1995) was used to assess contamination in estuarine sediments while the mean probable effects concentrations quotient (mPEC-Q; MacDonald et al. 2000; Ingersoll et al. 2001) was used for freshwater sediments.

A logistic regression model (LRM) was used in addition to the mERM-Q to assess contamination in estuarine sediments. The LRM is based upon modeled relationships between concentrations of individual contaminants and their documented effects on benthic organisms (Field et al. 2002; USEPA 2005). The individual LRM models were combined to generate the maximum probability (P_{max}) of toxicity due to contamination.

Table 6.2 Estuarine Sediment Quality Guidelines used in calculating the mERM-Q and LRM Pmax.

Sediment Contaminant	Included in mean Effects Range Median Quotient (mERM-Q)	Effects Range Low (ERL)	Effects Range Median (ERM)	Included in Logistic Regression Model (LRM)	LRM b_0	LRM b_1	LRM T25
Metals (ug/g dry weight)							
Antimony				x	-0.9005	2.4111	0.83
Arsenic	x	8.2	70	x	-4.1407	3.1674	9.13
Cadmium	x	1.2	9.6	x	-0.34	2.5073	0.5
Chromium	x	81	370	x	-6.4395	2.9952	60.69
Copper	x	34	270	x	-5.7878	2.9325	39.72
Lead	x	46.7	218	x	-5.4523	2.7662	37.49
Mercury	x	0.15	0.71	x	0.8041	2.5461	0.18
Nickel		20.9	51.6	x	-4.6119	2.7658	18.63
Silver	x	1	3.7	x	-0.1117	1.9684	0.32
Zinc	x	150	410	x	-7.9834	3.342	114.84
Organic pollutants (ng/g dry weight)							
Acenaphthene	x	16	500	x	-3.6165	1.7532	27.3
Acenaphthylene	x	44	640	x	-2.962	1.3797	22.42
Anthracene	x	85.3	1100	x	-3.6574	1.4854	52.8
Benz(a)anthracene	x	261	1600	x	-4.2013	1.5747	93.4
Benzo(b)fluoranthene				x	-4.5409	1.4916	203.13
Benzo(k)fluoranthene				x	-4.2781	1.5669	106.94
Benzo(a)pyrene	x	430	1600	x	-4.3005	1.5832	105.3
Biphenyl				x	-4.1144	2.2085	23.2
Chrysene	x	384	2800	x	-4.3241	1.5372	125.4
Dibenz(a,h)anthracene	x	63.4	260	x	-3.6308	1.7692	26.99
2,6-dimethylnapthalene				x	-4.0456	1.904	35.3
Fluoranthene	x	600	5100	x	-4.4574	1.4787	186.83
Fluorene	x	19	540	x	-3.7146	1.8071	28.03
Indeno(1,2,3-c,d)pyrene				x	-4.3674	1.6245	102.84
1-methylnapthalene				x	-4.1405	2.0961	28.26
2-methylnapthalene	x	70	670	x	-3.7579	1.7833	30.99
1-methylphenanthrene				x	-3.5884	1.7501	26.46
Napthalene	x	160	2100	x	-3.7753	1.6152	45.41
Perylene				x	-4.6827	1.7632	107.82
Phenanthrene	x	240	1500	x	-4.4576	1.6768	100.74
Pyrene	x	665	2600	x	-4.708	1.5854	189.08
Total PCB congeners	x	22.7	180	x	-3.4613	1.3488	56.45
4,4'-DDD				x	-1.8983	1.4913	3.44
4,4'-DDE		2.2	27	x	-1.8392	0.9129	6.48
4,4'-DDT				x	-1.7705	1.6786	2.51
Total DDT	x	1.6	46.1				
Dieldrin				x	-1.1728	2.558	1.07
*Total PCBs included the following congeners: 8, 18, 28, 44, 52, 66, 77, 101, 105, 110, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209							
**Total DDT represents the sum of 4,4'-DDT; 2,4'-DDT; 4,4'-DDE; 2,4'-DDE; 4,4'-DDD; 2,4'-DDD							
Sources: Long et al. 1995; Field et al. 2002							

Table 6.3 Great Lake sediment quality guidelines used in calculating the mPEC-Q.

Sediment Contaminant	Consensus-based Benchmark Effects Concentration (TEC) Values	Consensus-based Probable Effects Concentration (PEC) Values
Metals (ug/g dry weight)		
Arsenic	9.79	33
Cadmium	0.99	4.98
Chromium	43.4	111
Copper	31.6	149
Lead	35.8	128
Nickel	22.7	48.6
Zinc	121	459
Organic pollutants (ng/g dry weight)		
Total PAHs*	1610	22800
Total PCB congeners	60	676
*Total PAHs represents the sum of low molecular weight PAHs Acenaphthene, Acenaphthylene, Anthracene, Fluorene, 2-methylnaphthalene, Naphthalene, Phenanthrene, and high molecular weight PAHs Benz(a)anthracene, Benzo(a)pyrene, Chrysene, Dibenz(a,h)anthracene, Fluoranthene, Pyrene		
**Total PCBs included the following congeners: 8, 18, 28, 44, 52, 66, 77, 101, 105, 110, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209		
Sources: CCME 1999; MacDonald et al. 2000; Crane and Hennes 2007; Crane et al. 2002		

6.4.1 *Data Preparation*

For any given contaminant, results were excluded from the sediment contaminant calculations if the associated laboratory method detection limits (MDLs) exceeded the corresponding effects range low (ERL) and LRM T25 values for estuarine sediments (**Table 6.2**; Field and Norton 2014), or Threshold Effect Concentration (TEC) values for Great Lakes sediments (**Table 6.3**).

Sample results reported as nondetects (values less than the MDL) were substituted with one-half of the MDL. Total contaminant classes were calculated as the sum of concentrations of individual contaminants in each class:

- Total PAHs (calculated for Great Lakes sites only):
    - low molecular weight PAHs Acenaphthene, Acenaphthylene, Anthracene, Fluorene, 2-methylnaphthalene, Naphthalene, Phenanthrene, plus
    - high molecular weight PAHs Benz(a)anthracene, Benzo(a)pyrene, Chrysene, Dibenz(a,h)anthracene, Fluoranthene, Pyrene),
- Total PCBs: congeners 8, 18, 28, 44, 52, 66, 77, 101, 105, 110, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, and 209
- Total DDTs: 4,4'-DDT; 2,4'-DDT; 4,4'-DDE; 2,4'-DDE; 4,4'-DDD; 2,4'-DDD.

6.4.2 *Estuarine Contaminant Index Calculations*

6.4.2.1 *Mean effects range median quotient (mean ERM-Q)*

To calculate an individual ERM-Q, each sample contaminant result (*conc*) was divided by the ERM SQG (**Table 6.2**):

$$\textbf{\textit{Individual ERM-Q}} = \frac{conc}{ERM} \qquad \textbf{Equation 6-1}$$

To calculate the mean ERM quotient for each sample, individual ERM quotients were averaged:

$$\textbf{\textit{Mean ERM-Q}} = \frac{ERM\text{-}Q_{arsenic} + ERM\text{-}Q_{chronium} + \ldots ERM\text{-}Q_{Total\ PCBs}}{n} \qquad \textbf{Equation 6-2}$$

where n is the number of analytes included in the analysis (n = 23; see **Table 6.2**).

6.4.2.2 *Logistic regression model (LRM) maximum probability (P_{max})*[13]

Individual LRM probabilities were calculated as:

$$p = \frac{e^{b_0 + (b_1 * \log_{10} conc)}}{1 + e^{b_0 + (b_1 * \log_{10} conc)}} \qquad \textbf{Equation 6-3}$$

where *p* is the probability of observing a toxic effect; b_0 and b_1 *values* are provided in **Table 6.2**.

P_{max} is derived from the maximum *p* ($p_{maximum}$) result from a sample and calculated as:

$$P_{max} = 0.11 + (0.33 * p_{maximum}) + (0.4 * p_{maximum}^2) \qquad \textbf{Equation 6-4}$$

Nickel was excluded from LRM P_{max} calculations for West Coast samples due to naturally high background levels of nickel in the region (Lauenstein et al. 2000; Long et al. 1995; and Nelson 2008).

6.4.3 *Great Lakes Contaminant Index Calculations*

6.4.3.1 *Mean Probable Effects Concentration Quotient (mean PEC-Q)*

Individual PEC-Qs were calculated for metals, total PAHs and total PCBs as the contaminant concentration result divided by the PEC SQG (**Table 6.3**):

$$\textbf{\textit{Individual PEC-Q}} = \frac{conc}{PEC} \qquad \textbf{Equation 6-5}$$

The mean PEC-Q for metals was calculated as:

[13] ERMQ and PEC-Q look at a range of priority pollutants and calculate the mean probability of adverse effects. In contrast, LRM uses a single contaminant (the highest) at each station to predict impairment.

$$\boldsymbol{Mean\ PEC\text{-}Q_{metals}} = \frac{\sum \boldsymbol{Individual\ PEC\text{-}Qs}}{\boldsymbol{n}} \qquad \textbf{Equation 6-6}$$

where *n* is the number of metals included in the analysis (*n* = 7). Only metals with reliable PECs (i.e.: arsenic, cadmium, chromium, copper, lead, nickel, and zinc) were included.

The mean PEC-Q was calculated as the average of the above PEC-Qs:

$$Mean\ PEC\text{-}Q = \frac{(mean\ PEC\text{-}Q_{metals} + PEC\text{-}Q_{Total\ PAHs} + PEC\text{-}Q_{Total\ PCBs})}{n} \qquad \textbf{Equation 6-7}$$

Where *n* is the number of contaminant classes (*n* = 3).

Once the index is calculated, benchmarks are applied to categorize results in to good, fair and poor condition. Sediment contaminant benchmarks are based on literature review and best professional judgement and are intended to represent the probability of toxicity (**Table 6.4**). Mean SQG quotients were developed from prior studies relating guideline exceedances and observed toxicity levels (Ingersoll et al. 2001; Crane et al. 2002; Field et al. 1999; Field et al. 2002; Long et al. 1998). The benchmarks for good correspond to low incidence of toxicity, fair with less known incidence of toxicity and poor with a higher incidence of toxicity.

Table 6.4 Benchmarks for NCCA sediment contaminant index (SCI)

Sediment Contaminant Index (SCI) Condition		
Condition	**Estuarine**	**Great Lakes**
Good	mean ERM-Q <0.1; LRM Pmax benchmark ≤0.5	mean PEC-Q ≤ 0.1
Fair	mean ERM-Q 0.11-0.5; LRM Pmax benchmark >0.5-<0.75	mean PEC-Q > 0.1 and ≤ 0.6
Poor	mean ERM-Q >0.5; LRM Pmax benchmark ≥0.75	mean PEC-Q > 0.6

There was very little variance in the estuarine sediment contaminant index results. 58 of 63 sites showed agreement between good and fair ratings between visit 1 and 2 (See **Table 6.5**). Only one site was rated poor at visit 2.

Table 6.5 Estuarine sediment contaminant index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	58	1	
	Fair	1	2	
	Poor		1	

There was also very little variance in the Great Lakes sediment contaminant index. 30 of 33 sites showed agreement between good and fair ratings between visit 1 and 2 (See **Table 6.6**). Zero sites were rated poor.

Table 6.6 Great Lakes sediment contaminant index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	26		
	Fair	3	4	
	Poor			

6.5 SEDIMENT TOXICITY INDEX CALCULATIONS

Sediment toxicity tests provide a second line of evidence used for assessing whether sediments can support a healthy benthic ecosystem. They compare the ability of test organisms to survive in samples collected in the field to samples that come from an area of known clean sediment. Control corrected survival is used to assess both estuarine and Great Lakes sediment samples, statistically significant differences between field and control samples are only assessed for estuarine samples.

6.5.1 *Data Preparation*

NCCA sediment toxicity testing data underwent quality checks to ensure that the results were suitable for use in the sediment toxicity index. In some cases, one or two replicates from a sample were excluded from analysis, while in others entire samples were excluded. When a *replicate* is removed from analysis due to QA/QC concerns, the data associated with that replicate are neither used in calculating control-adjusted survival for the sample, nor are they used in the significance tests for marine samples. When a *sample* is removed from analysis due to QA/QC concerns or there are an insufficient number of replicates for that sample, the condition category for the sediment toxicity index at the site will be considered missing. Sediment toxicity results excluded from analysis are deprecated from the sediment toxicity data published on the NARS website. See **Table 6.7** for a summary of excluded sediment toxicity test results.

Table 6.7 Summary of excluded sediment toxicity test results

	Number of results removed			
	Estuarine		Great Lakes	
Reason	**Replicate**	**Sample**	**Replicate**	**Sample**
Predators present in test chamber	7	2	0	0
Large particle size impacted test organism survival	1	0	0	0
Laboratory misload: 0 organisms loaded	1	0	0	0
Laboratory misload: unknown number of organisms greater than protocol number	1	0	1	0

6.5.2 *Control-corrected survival*

Control-corrected survival for each sample was calculated for both estuarine and freshwater sediments. Sample mean percent survival, or average survival across sample replicates, was divided by control mean percent survival, or average survival across corresponding control replicates, as follows:

$$\textbf{\textit{Control-corrected survival}} = \frac{\textbf{\textit{sample mean percent survival}}}{\textbf{\textit{control mean percent survival}}} \qquad \textbf{Equation 6-8}$$

6.5.3 *Significance tests (estuarine only)*

For each sample, mean field sample survival was compared to mean control survival. First, normality of the raw sample replicate data was assessed via Shapiro-Wilkes tests (data were considered normal if *p-value* > 0.10). If raw data were not normally distributed, the arc-sine square root transformation was applied, and normality of transformed values was assessed again. For normally distributed data (either raw or transformed), one-sided t-tests with equal or unequal variances (homogeneity of variance was assessed via Bartlett tests; variances were considered equal if *p-value* > 0.10) were performed for each sample and the associated control batch. For data that were not normally distributed (even after transformation), one-sided Wilcoxon tests were performed. Sample replicate survival was considered significantly less than control replicate survival if *p-value* < 0.05 for both t-tests and Wilcoxon tests.

Once the control-corrected survival and significance tests are calculated, benchmarks are applied to categorize results into good, fair and poor condition. Sediment toxicity index benchmarks are based on literature values for estuarine samples (Long et al. 1998; Greenstein et al. 2011) and Great Lakes samples (USEPA 2004; Norberg-King et al. 2006; See **Table 6.8**).

Table 6.8 Benchmarks for NCCA sediment toxicity index (STI)

Sediment Toxicity Index (STI) Condition		
Condition	**Estuarine**	**Great Lakes**
Good	control-corrected survival ≥ 80% and not statistically significantly less than control (p > 0.05)	control-corrected survival ≥ 90%
Fair	control-corrected survival ≥ 80% and statistically significantly less than control (p ≤ 0.05) *or* control-corrected survival < 80% and not significantly less than control (p > 0.05)	control-corrected survival ≥ 75% and < 90%
Poor	control-corrected survival < 80% and statistically significantly less than control (p < 0.05)	control-corrected survival < 75%

There was a large amount of disagreement in the estuarine sediment toxicity index. Of the 60 sites that were visited twice, 33 agreed in good or fair ratings. Most of the disagreement was from sites that were rated fair for the first visit and good for the second visit. There was no agreement in poor ratings (See **Table 6.9**)

Table 6.9 Estuarine sediment toxicity index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	32	18	1
	Fair	5	1	1
	Poor	1	1	

In contrast to the estuarine sediment toxicity index, the Great Lakes showed very little variance and agreed on good rating for 28 of the 32 revisit sites. There were no sites rated poor at visit 1 or visit 2, and no sites rated fair at visit 1 (See **Table 6.10**).

Table 6.10 Great Lakes sediment toxicity contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	28		
	Fair	4		
	Poor			

6.6 SEDIMENT QUALITY INDEX CALCULATIONS

The sediment contaminant indices and sediment toxicity indices contribute equally to the sediment quality index (**Table 6.11**). For a site to be rated good, both component indices must be rated good, and a site will be rated poor if either of the component indices are rated poor.

Table 6.11 Benchmarks for NCCA sediment quality index

Sediment Quality Index Condition	
Good	SCI = good and STI = good
Fair	SCI = fair and/or STI = fair (SCI ≠ poor; STI ≠ poor)
Poor	SCI = poor and/or STI = poor

Driven by the sediment toxicity index, the estuarine sediment quality index shows very little agreement between visits 1 and 2 (See **Table 6.12**).

Table 6.12 Estuarine sediment quality index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	31	18	1
	Fair	5	4	1
	Poor	1	2	

The overall Great Lakes sediment quality index showed agreement with the exception of a few sites rated fair at visit 1 or 2. No sites were rated poor (See **Table 6.13**).

Table 6.13 Great Lakes sediment quality index contingency table

		Visit 1		
		Good	Fair	Poor
Visit 2	Good	22		
	Fair	7	4	
	Poor			

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7 ECOLOGICAL FISH TISSUE CONTAMINANT INDEX

7.1 OVERVIEW

The NCCA is designed to characterize ecological conditions in near-shore marine and freshwater coastal resources at regional and continental scales. The ecological fish tissue contaminant index (EFTCI) is a generalized approach that accommodates the wide variety of species, climate zones, geographies, salinity regimes and sampling methods that comprise the target populations of this national assessment. This approach provides a nationally consistent way to screen a wide variety of fish species in diverse ecosystems for contaminants that may lead to adverse ecological effects within the food web.

Contaminant concentrations in fish provide a time-integrated measurement of chemical bioavailability, fate and distribution. The NCCA measures concentrations of select contaminants in whole-fish tissues to assess the biologically available contaminant levels in the Nation's coastal waters. Tissue chemistry results are compared to a suite of contaminant screening values to evaluate whether potential exposure may lead to adverse effects for predatory wildlife that depend on fish as a primary food source (or "piscivorous" wildlife). This analysis culminates with the EFTCI that creates ratings of good, fair or poor based upon the degree to which contaminants are found in fish composite samples.

Wildlife, such as fish, birds and mammals, can be exposed to contaminants in several ways (e.g., ingestion, dermal contact and inhalation). Ingestion is a common mode of wildlife exposure that typically occurs by incidental consumption of soils or sediments associated with the food source; drinking contaminated surface water; or eating prey that have accumulated contaminants in their bodies. This approach specifically assesses the potential that piscivorous wildlife may experience adverse effects when exposed to ingested contaminants that have accumulated in the tissues of target fish caught during the NCCA survey. To assess contaminant levels in tissues, the EFTCI is calculated using an adaptation of EPA's ecological risk assessment guidelines (USEPA 1997) based on a wildlife exposure framework (USEPA 1993). The NCCA analyzes whole-body fish samples for a broad list of legacy environmental contaminants, including metals and metalloids, some pesticides, and other persistent organic pollutants such as PCBs. For a full list of the analytes analyzed for in whole fish samples, see the NCCA 2015 LOM (USEPA 2015b). For a list of species used for analyses in each NCCA Region, please see Appendix A.5.

The EFTCI relies on development of screening values[14] that can be summarized to characterize the potential for multiple contaminant exceedances in a fish tissue composite from each site. Screening values derived for each of the NCCA contaminants of interest in relation to broad categories of piscivorous wildlife (i.e., fish, birds and mammals) are used to account for the challenges of assessing the ecological relevance of fish tissue contaminants at a national scale. Information on the development

[14] In contrast to other contaminants in the EFTCI that do not have published screening values, the EPA has published a final national aquatic life criterion recommendation for selenium in freshwater (USEPA 2016b). The selenium screening values used in the EFTCI are described in Section 7.4.6.

of these screening values and EFTCI was first prepared for US EPA Region 6 by Tetra Tech, Inc. (Tetra Tech 2012) and subsequently presented in the *NCCA 2010 Technical Report* (USEPA 2016a). Updated information for 2015, including revised equations[15], are described in detail below.

7.2 FIELD COLLECTION AND LABORATORY ANALYSIS

Composited whole-body fish samples of select forage-size fish species were analyzed for a suite of metals, metalloids, and organic pollutants (**Table 7.1**; USEPA 2015a; 2015b).

Table 7.1 Laboratory methods for fish tissue contaminant analysis

Parameter	Method
Metals (excluding mercury)	EPA 6020
Mercury	EPA 245.7
PCBs, pesticides	EPA 8270D

7.3 DATA PREPARATION

In cases where the laboratory reported values that were non-detect (i.e., below the method detection limit), results were set to zero. Result values were then converted to a dry weight (mg/kg dw) concentration by dividing by a constant (0.28) that approximates the proportion of solids in wet fish tissue composite samples (USEPA 1993). The inorganic fraction of arsenic was estimated as 10% of the total arsenic concentration reported (USEPA 2003). Total mercury was assumed to consist primarily of methylmercury and was not adjusted to remove the non-methylated fraction (Wagemann et al. 1997). Total organic contaminant groups were calculated by summing the concentration of components in the fish tissue composite samples (See **Table 7.2**)

[15] The EFTCI used for the NCCA 2010 Report has been updated for the NCCA 2015 Report with revised formulae. The updated formulae have also been used to revise results for the NCCA 2010 EFTCI, which is reflected in an addendum to the NCCA 2010 Report (https://www.epa.gov/national-aquatic-resource-surveys/national-coastal-condition-assessment-2010-report) and NCCA Dashboard (https://coastalcondition.epa.gov/).

Table 7.2 Organic Contaminant groups used for calculating the EFTCI.

Organic Contaminant Group	Components
Total DDT	2,4'-DDD; 2,4'-DDE; 2,4'-DDT; 4, 4'-DDD; 4,4'-DDE; 4,4'-DDT
Total Chlordane	Alpha-chlordane, cis-nonachlor, gamma-chlordane, heptachlor, heptachlor epoxide, oxychlordane, trans-nonachlor
Total Endosulfan	Endosulfan sulfate, Endosulfan I, Endosulfan II
Total Endrin	Endrin, Endrin Ketone, Endrin Aldehyde
Total PCB	PCB8, 18, 28, 44, 52, 66, 77, 101, 105, 110, 118, 126, 128, 138, 153, 170, 180, 187, 195, 206, 209

7.4 SCREENING VALUES

There are few published ecologically based fish contaminant assessment approaches that are appropriate in the context of the NCCA, wherein a single composite sample is collected and analyzed for contamination once every five years from a probabilistically selected site. However, the evaluation of risk using food webs for contaminant exposure through dietary uptake is well documented (ODEQ 2007; US ARMY 2006; CCME 1998; Sample et al. 1996; Newell et al. 1987). For its Superfund program under RCRA (USEPA 1997; 1998; 1999), EPA suggests the use of a tiered approach, including a screening-level analysis (such as used for the NCCA EFTCI), as a part of determining the level of effort needed for ecological risk assessments. Methods described in the *Wildlife Exposure Factors Handbook* (USEPA 1993) serves as the analytic framework for conducting screening-level risk assessments for common wildlife species. These screening-level risk assessments may be used for several purposes, including: to assess potential effects of environmental contamination on wildlife species and to support site-specific decisions (e.g., for hazardous waste sites); to support the development of water-quality or other media-specific criteria for limiting environmental levels of toxic substances to protect wildlife species; or to *focus research and monitoring efforts*. A screening-type approach is a cost-effective first step in conducting wildlife exposure assessments, and is suitable for use in the NCCA to characterize the potential for contaminants in fish to adversely affect predators.

For the EFTCI, ecologically relevant screening values were calculated to evaluate whether the concentrations of metals, metalloids and organic contaminants measured in whole-body fish tissue potentially lead to adverse effects when consumed by predatory fish and piscivorous wildlife. See Appendix A for the laboratory-based endpoints based on surrogates for each group of receptors.

7.4.1 *Receptors of Concern*

Receptors of concern (ROCs) are typically those animals that are exposed to contaminants through ingestion, dermal contact, and/or inhalation (USEPA 1997). The exposure of ROCs to contaminants by ingestion is through either incidental media uptake (i.e., eating soil or sediment that is associated with prey items), drinking contaminated surface water, or through the ingestion of prey organisms that

have accumulated contaminants in their tissues. The EFTCI considers upper trophic level organisms including birds, fish and mammals to be ROCs. For NCCA, contaminant concentrations were measured in whole-body forage fish tissue composite samples and risk assessment considerations were evaluated based solely on the uptake of contaminants that have been accumulated in the tissues of prey items consumed by groups of ROCs.

Generalized ROC groupings (See **Table A.3.1**) informed development of exposure-based screening values because fish composite samples were collected in both freshwater and marine waters across all US coastal resources. These classes include freshwater predatory fish, marine predatory fish, piscivorous birds, piscivorous freshwater mammals and piscivorous marine mammals. Receptors were chosen based on having predominantly fish diets and the availability of contaminant exposure data in the literature. Species that comprise receptors of concern groupings for NCCA evaluations represent those aquatic dependent organisms that are commonly included in ecological risk assessments (**Appendix A.1**).

7.4.2 *Ecological Risk Assessment Based Approach for Deriving Screening Values*

Under EPA (1997) guidelines, risk is defined as a ratio of exposure concentration of contaminant to a concentration that is known to have toxicological effects (toxicity reference value) in specific biological species. In a typical application, risk potential is derived by calculating a hazard quotient (HQ), which is a ratio expressed by dividing exposure concentration by a reference concentration (or toxicity reference value) known to elicit adverse toxicological effect (Low Observed Adverse Effects Level or LOAEL, Equation 7.1, **below**). Similarly, a HQ can be calculated for more conservative contaminant exposure concentrations that are known to *not* elicit toxicological effects by substituting the No Observed Adverse Effect Level or NOAEL as the toxicity reference value. Risk can be expressed as:

$$\boldsymbol{HQ\ (risk) = \frac{Exposure\ Concentration}{Toxicity\ Reference\ Value}} \qquad \textbf{Equation 7-1}$$

Thus, when the exposure concentration of a contaminant is greater than the concentration known to elicit toxic effects, the HQ is greater than 1.0, and the receptor is at risk.

Following through, the exposure concentration can be represented by Equation 7-2**, below**:

$$\boldsymbol{Exposure\ Concentration = \frac{FIR \times [fish] \times AUF}{BW}} \qquad \textbf{Equation 7-2}$$

Where:

FIR = food ingestion rate (kg food/d)

[fish] = concentration in fish tissue (mg/kg)

AUF = area use factor

BW = body weight of receptor (kg)

If the concentration known to elicit toxic effects can be estimated for different ROCs, then the exposure concentration equation can be rearranged to produce screening values representing an HQ = 1 for each receptor and contaminant combination. See Section 7.5 *Application of Screening Values for the EFTCI* for specific usage. The following sections describe the methods for parameterizing receptor and contaminant characteristics such as body weight, food ingestion rate, and toxicity reference values (TRVs), which are components needed to quantify NCCA fish tissue contaminant screening values. To be most protective, the AUF is set to 1.0 indicating all foraging, resting, breeding and other activities are expected to occur within the exposure area of concern.

7.4.3 *Receptor Characteristics: Body Weight and Food Ingestion Rate*

Because food ingestion for birds and mammals and daily ration for fish are based on the metabolism of the animal, smaller individuals generally consume more food per unit of body weight than larger receptors (USEPA 2016a; See **Table 7.3**).

Food ingestion rates were available for marine fish (Maldeniya 1996) and freshwater fish (Carlander 1969). In the absence of FIR in the literature, we used allometric regression models based on metabolic rate and expressed in terms of body weight (g) for birds and mammals (Nagy et al. 1987). These models are described in more detail in Sample et al (1996) and Sample and Arenal (1999). To be most protective, the food ingestion rates were calculated based on the minimum body weight of the receptor. The receptor exhibiting the lowest body weight and highest ingestion rate was selected to represent each respective receptor group (See **Table 7.3**). The body weight and food ingestion rate each of the generalized receptors were used to calculate toxicity reference values (Equation 7-3) and screening values (Equation 7.4) for each contaminant. While the body weights and food ingestion rates for freshwater and marine mammals and fish are listed in **Table 7.3**, the freshwater organisms were used in this generalized screening because they have higher food ingestion rates per body weight and are therefore contribute to more protective TRVs and subsequent screening values.

Table 7.3 Summary of generalized receptor body weights and daily food ingestion rates used to calculate screening fish tissue values.

Receptor Group	Body Weight (kg)	Food Ingestion Rate (kg food/kg BW/d)	Daily Food Ingestion Rate (kg/d)
Birds (*Megaceryle alcyon*)	0.13	0.118	0.0156
Freshwater Mammals (*Nevision vison*)	0.55	0.076	0.0420
Marine Mammals (*Phoca vitulina*)	58.8	0.033	1.956
Freshwater Fish (*Esox masquinongy*)	0.34	0.064	0.02176
Marine Fish (*Thunnus albacares*)	23.42	0.023	0.539

7.4.4 *Wildlife Toxicity Reference Value (TRV) Calculations*

For NCCA, toxicity is defined by a toxicity reference value (TRV), which is a derived from no observable adverse effects level (NOAEL) values generated from laboratory-based experimental studies. NOAELs represents the contaminant concentration above which ecologically relevant adverse effects might occur in wildlife populations after long-term dietary exposure. Literature-based NOAEL values used to derive the NCCA TRVs are based on laboratory surrogate species (e.g., chickens, quail, duck, rat mouse, rainbow trout and Japanese medaka). For some contaminants and receptors of concern, laboratory-based tests used to develop TRVs may not have resulted in endpoints that were protective for chronic exposure. In such cases, a chronic exposure endpoint was determined from the reported endpoint using a conversion factor (Sample, et al. 1996). Fish TRVs were extracted from existing literature (See Appendix A.1). Avian and mammal TRVs were acquired from reported laboratory tests, then scaled to the food ingestion rates[16] and body weights of the selected NCCA ROCs using the interspecies allometric model introduced in Sample and Arenal (1999). NOAEL concentrations for the contaminants evaluated for the NCCA for each generalized receptor of concern is shown in **Table 7.4**. See Appendix A.1 for sources of each NOAEL value.

Table 7.4 $NOAEL_{test}$ values (for use in Equation 7-3 to calculate $TRV_{wildlife}$) for contaminants or contaminant classes calculated for each generalized ROC.

Analyte	Avian $NOAEL_{test}$ (mg/kg-bw/d)	FW Mammal $NOAEL_{test}$ (mg/kg-bw/d)	FW Fish $NOAEL_{test}$ (mg/kg-bw/d)
Arsenic, Inorganic	5.1	0.126	0.02563
Cadmium	1.45	0.75	20
Chlordane, Total	0.8	4.58	NA
DDT, Total	0.3	0.8	0.143
Dieldrin	0.08	0.028	0.024
Endrin, Total	0.02	0.18	0.04
Endosulfan, Total	10	1	0.0002393
Hexachlorobenzene	0.11	1	0.000685
Lindane	0.56	8	10
Mercury	0.03	0.032	0.06768
Mirex	0.01	0.07	0.3
PCB, Total	0.18	0.068	0.05

[16] Food ingestion rates for fish were found in Carlander 1969. Avian and mammalian food ingestion rates were calculated using regression equations derived from Nagy (1987).

$$\mathbf{TRV_{wildlife}} = \mathbf{NOAEL_{test}} \times \left(\frac{\mathbf{BW_{test}}}{\mathbf{BW_{wildlife}}}\right)^{(1-x)} \qquad \textbf{Equation 7-3}$$

Where:

$TRV_{wildlife}$	=	toxicity reference value for wildlife species (See Table 7.4)
$NOAEL_{test}$	=	no observed adverse effect level for test species
BW_{test}	=	body weight for test species (See Appendix A)
$BW_{wildlife}$	=	body weight for wildlife species
x	=	scaling factor[17]

Table 7.5 Wildlife TRVs for each contaminant and generalized ROC, based upon NOAELS reported in Table 7.3. (for use in Equation 7-4 to calculate screening values).

Analyte	Avian $TRV_{wildlife}$ (mg/kg-bw/d)	FW Mammal $TRV_{wildlife}$ (mg/kg-bw/d)	FW Fish $TRV_{wildlife}$ (mg/kg-bw/d)
Arsenic, Inorganic	3.391242502	0.105822023	0.025846448
Cadmium	0.937110516	0.892564279	72.0589905
Chlordane, Total	0.531959608	3.846546545	NA
DDT, Total	0.147013886	0.778596251	0.26802071
Dieldrin	0.061867004	0.0333224	0.06181342
Endrin, Total	0.018739766	0.151174318	0.147421822
Endosulfan, Total	7.986871041	1.190085705	0.000254478
Hexachlorobenzene	0.106896406	0.973245314	0.001665614
Lindane	0.544199885	7.785962511	14.14703843
Mercury	0.019948485	0.03114385	0.134918462
Mirex	0.006649495	0.063889156	0.372954577
PCB, Total	0.119690912	0.054557547	0.073362105
Selenium[18]	0.265979804	0.194649063	

[17] Scaling factors presented by Sample and Arenal (1999) indicate that mammalian sensitivity increases with increased body weight, and avian sensitivity increases with decreased body weight. Scaling factors were unavailable for fish receptors but, like avian receptors, an increase in sensitivity with decreased body weight was reported (Buhler and Shanks, 1970). A scaling factor of 0.94 is used for mammalian receptors (Sample and Arenal, 1999) and a scaling factor of 1.2 is used for avian receptors (Sample and Arenal, 1999) and fish receptors (Buhler and Shanks, 1970).

[18] TRVs are the best available science for developing screening values for most elemental and organic contaminants analyzed for this index. In 2016b, EPA developed an aquatic life criterion for selenium, which was used to derive the screening values for Se. See Section 7.4.6.

7.4.5 *Screening Values*

Using the proxy TRVs (**Table 7.5**), and the body weights and daily food ingestion rates from **Table 7.3**, adverse dietary exposure concentrations in the fish that NCCA ROCs may eat can be estimated. Screening values for NCCA tissue contaminants of interest were derived using the following equation:

$$\mathbf{SV} = \frac{(\mathbf{TRV_{wildlife}} \times \mathbf{BW_{wildlife}})}{\mathbf{FIR_{wildlife}}} \qquad \textbf{Equation 7-4}$$

Where:

SV = screening value concentration in fish tissue (mg/kg) for specific analyte

TRV = estimated wildlife toxicity reference value (mg/kg bw/d)

BW = body weight (kg) of wildlife ROC used represent receptor group (kg)

FIR = daily food ingestion rate (kg food/d) of wildlife ROC used to represent receptor group

Each screening value represents an estimated contaminant concentration in fish tissue composite samples that could result in the minimum risk for exposure to each receptor group. Minimum potential exposure risks (HQ=1.0) were calculated for each analyte and receptor group combination to serve as screening benchmarks.

7.4.6 *EPA Tissue-Based Criteria Deriving Selenium Screening Values for Fish*

In 2016, the EPA published the final national aquatic life criterion recommendation for selenium in freshwater (USEPA 2016b).[19] The EPA selenium criterion recommendation is composed of four elements, including two criterion elements based on the concentration of selenium in fish tissue (i.e., egg-ovary element and whole-body or muscle element). This criterion represents the best available science on the toxicity of selenium to freshwater aquatic life; therefore, the EFTCI utilized the whole-body chronic selenium criterion element to derive the selenium screening value for fish ROCs.

The methods used to derive the EPA selenium criterion recommendation differ from the TRV methods described in previous sections. The EPA derived the selenium criterion recommendation using the procedure outlined in EPA's *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (Stephan et al. 1985). The dataset used to derive the

[19] The 2016 "Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, 2016," is a chronic criterion that is composed of four elements. All elements are protective against chronic selenium effects. Two elements are based on the concentration of selenium in fish tissue and two elements are based on the concentration of selenium in the water column. The recommended elements are: (1) a fish egg-ovary element; (2) a fish whole-body and/or muscle element; (3) a water column element (one value for lentic and one value for lotic aquatic systems); and (4) a water column intermittent element to account for potential chronic effects from short-term exposures (one value for lentic and one value for lotic aquatic systems).

tissue-based criterion consists primarily of fish species: 12 fish species representing 10 genera and 7 families. Reproductive toxicity tests using dietary exposures yielded effects endpoints (e.g., larval mortality and deformities) that were set to the 10% effect concentration (EC_{10}) for use in the derivation of a selenium criterion. An EC_{10} is generally of similar magnitude to a NOAEL (Iwasaki et al. 2015), which is the TRV effect level used for other contaminants making it appropriate for use in the EFTCI. Whole body reproductive chronic values were calculated directly from whole body tissue concentrations measured in the study or by applying an egg-ovary (EO) to whole-body (WB) conversion factor, detailed in Section 3.2.2.2 of *Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater 2016* (EPA, 2016b). The final criteria recommendation is derived from the species sensitivity distribution and represents the concentration of selenium in egg-ovary, whole body, or muscle tissue that would be protective against selenium effects on fish reproduction.

EFTCI screening values are set as dietary concentrations in prey species rather than the selenium body-burden of the receptor. Therefore, we translated the whole-body selenium criterion element (8.5 mg/kg dw) to a dietary endpoint by accounting for the trophic transfer of selenium through the dietary pathway with trophic transfer factor (TTF) values. TTFs quantify the degree of bioaccumulation between trophic levels. The EPA derived TTF values for invertebrates and a wide range of fish based on taxonomic relationships that do not directly align with the EFTCI fish ROC (EPA 2016b; see Appendix B). To account for the expected trophic transfer of selenium for the EFTC fish ROCs, we used the median TTF value for fish species that are piscivorous as adults (median = 1.41, see **Table 7.6**). The median TTF was then applied to the whole-body fish tissue criterion of 8.5 mg/kg dry weight resulting in a selenium SV of 6.05 mg/kg dry weight for fish ROCs.

Table 7.6 EPA-derived trophic transfer factor (TTF) values presented in the Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater (USEPA 2016b; see Table 3.11) and used to derive the median TTF for piscivorous fish.

Order	Species	TTF
Esociformes	Northern pike	1.78
Perciformes	Largemouth bass	1.39
Perciformes	Smallmouth bass	0.86
Perciformes	Striped bass	1.48
Perciformes	Walleye	1.6
Perciformes	Yellow perch	1.42
Salmoniformes	Brown trout	1.38
Salmoniformes	Rainbow trout	1.07
	Median TTF	1.41

7.5 APPLICATION OF SCREENING VALUES FOR THE EFTCI

The ROC with the lowest body weight and highest ingestion rate was selected to represent a sensitive species in each wildlife receptor group. The receptor group and ROC species combinations used for this report are: mammals – mink (*Neovison vison*); birds – belted kingfisher (*Megaceryle alcyon*); and predatory fish – muskellunge (*Esox masquinongy*).

NCCA whole-body fish tissue contaminant analysis results were compared to these calculated benchmarks to determine if piscivorous fish and wildlife may be at risk due to the consumption of fish. In **Table 7.7**, screening values for each group of receptors are summarized.

Table 7.7 NCCA ecological risk-based screening values for receptors of concern

Contaminant	NCCA Whole-Body Fish Tissue Contaminant Screening Values (mg/kg dw) (NOAEL-based)		
	MAMMAL[a]	**AVIAN**	**FISH[b]**
Arsenic (inorg)	1.3849	28.5892	0.4039
Cadmium	11.6807	7.9001	1125.9217
Mercury	0.4076	0.1682	2.1081
Selenium	2.5473	2.2423	6.05[c]
Dieldrin	0.4361	0.5216	0.9658
Total Endosulfan	15.5742	67.3318	0.004
Total Endrin	1.978	0.158	2.3035
Total Chlordane	50.3383	4.4846	NA
Hexachlorobenzene	12.736	0.9012	0.026
Lindane	101.892	4.5878	221.0475
Mirex	0.836	0.0561	5.8274
Total DDTs	10.1892	1.2394	4.1878
Total PCBs	0.714	1.009	1.1463

[a] Two mammal receptor group screening values calculated. The more sensitive freshwater mammal group was used in the EFTCI.

[b] Two fish receptor group screening values calculated. The more sensitive freshwater fish group was used in the EFTCI.

[c] Selenium screening value derived using the median EPA trophic transfer factor value of piscivorous species.

Table 7.8 describes how screening values were translated to good, fair and poor ratings for each NCCA site.

Table 7.8 Application of SV in the NCCA ecological fish tissue contaminant assessment. The result is an EFTCI for each site surveyed.

Ecological Fish Tissue Contaminant Index Condition	
Good	All of the measured contaminant concentrations < screening value for all receptor group.
Fair	At least one measured contaminant concentration ≥ screening value for one receptor group.
Poor	At least one measured contaminant concentration ≥ screening value for two or more receptor groups.

7.6 REFERENCES

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8 HUMAN HEALTH FISH TISSUE INDICATOR

Fish are time-integrating indicators of persistent pollutants, and contaminant bioaccumulation in fish tissue has important human health implications. Contaminants in fish pose various health risks (e.g., cancer risks, and noncancer risks such as reproduction or neurological development impacts) to human consumers. The NCCA 2015 human health fish tissue indicator consists of collection and analysis of two types of fish composite samples, including fish fillet plug samples and whole fish samples for homogenized fillet analyses. Collectively, these samples provide information on the distribution of selected persistent, bioaccumulative, and toxic (PBT) chemical residues (e.g., mercury, polychlorinated biphenyls or PCBs, and per- and polyfluoroalkyl substances or PFAS). The fish fillet plug samples were collected from both the marine and Great Lakes sites and analyzed for mercury only. The whole fish samples for homogenized fillet composite analysis were collected from Great Lakes nearshore sites only and analyzed for mercury, PCBs, and PFAS.[20] **Table 8.5** for a summary of PFAS results.

Field and analysis procedures for the Great Lakes human health homogenized fillet tissue indicator described below were based on EPA's National Study of Chemical Residues in Lake Fish Tissue (USEPA 2009) and EPA's *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories*, Volume 1, third edition (USEPA 2000). Fish were scaled and filleted in the laboratory where muscle fillets from both sides of each fish were prepared with skin on and the belly flap attached, and fillets from all of the individual specimens that comprise a composite sample from a site were homogenized together.

8.1 FIELD FISH COLLECTION

8.1.1 *Whole Fish Samples for Chemical Analyses of Homogenized Fillet Tissue Composite Samples*

The NCCA 2015 crews collected whole fish samples for chemical analyses of homogenized fillet composite samples from Great Lakes sites only. The 152 fish samples collected for this Great Lakes human health fish tissue indicator consisted of a composite of fish (i.e., typically five similarly sized individuals of one target species) from each site. The fish had to be large enough to provide sufficient tissue for analysis and for archiving (when possible) (i.e., 155 grams of fillets for analysis and 330 g for archive, collectively). Additional criteria for each fish composite sample included fish that were:

- All of the same species (for each site);
- Harvestable size per legal requirements or of consumable size if there were no harvest limits; and
- Similar size so that the smallest individual in the composite from a site was no less than 75% of the total length of the largest individual in the composite.

[20] For the NCCA 2015 survey, a composite sample was formed by combining fillet tissue from up to five adult fish of the same species and similar size from the same site. Use of composite sampling for screening studies is a cost-effective way to estimate average contaminant concentrations while also ensuring that there is sufficient fish tissue to analyze for all contaminants of concern.)

Crews were provided with a recommended list of target fish species and a list of alternative species in the field operations manual (USEPA 2015a); however, if none of the recommended fish species were available, crews chose an appropriate substitute. **Table 8.1** provides a list of the fish species successfully collected for this human health fish tissue indicator and identifies the number of samples collected for each fish species.

8.1.2 *Fish Tissue Plug Samples for Mercury Analysis*

The NCCA crews removed fish fillet tissue plugs (taken from dorsal muscle) from whole fish that were collected for the ecological fish tissue indicator if they were also on the target list for mercury analysis (See **Table 8.2**). They attempted to collect fish fillet plug samples from all marine and Great Lakes sites. To form a fillet plug sample, the crews collected fillet tissue plugs from two fish of the same species (one plug per fish). Crews collected each fillet tissue plug by inserting a biopsy punch into a de-scaled thicker area of dorsal muscle section of a fish. After plug sample collection from live fish, they placed antibiotic salve over the wound and released the fish. The crews were provided with a recommended list of target fish species for fish plug sample collection in the field operations manual (USEPA 2015a); however, if none of the recommended fish were available, crews chose an appropriate substitute. **Table 8.2** provides a list of the fish species collected for fillet plug sample removal by geographic area and identifies the number of fillet plug samples collected from each fish species in a geographic area.

8.2 MERCURY ANALYSIS AND HUMAN HEALTH FISH TISSUE BENCHMARK

All fish tissue samples (both homogenized fillet composite tissue and fillet tissue plug samples) were analyzed for total mercury. The samples were prepared using EPA Method 1631B, Appendix A (USEPA 2001a) and analyzed using EPA Method 1631E (USEPA 2002), which utilizes approximately 1 g of fillet tissue for analysis. In screening-level studies of fish contamination, EPA guidance recommends monitoring for total mercury rather than methylmercury (an organic form of mercury) because most mercury in adult fish is in the toxic form of methylmercury, which will be captured during an analysis for total mercury. Applying the conservative assumption that all mercury is present in fish tissue as methylmercury is also more protective of human health (USEPA 2001b and Bloom 1992). The human health benchmark used to interpret mercury concentrations in fillet tissue is 0.3 milligrams (mg) of methylmercury per kilogram (kg) of tissue (wet weight) or 300 parts per billion (ppb), which is EPA's tissue-based water quality criterion for methylmercury (USEPA 2001b). This human health fish tissue benchmark represents the chemical concentration in fish tissue that, if exceeded, may adversely impact human health. NCCA fish tissue collection data were screened to exclude samples where non-target species were collected (i.e., species that are not typically consumed by humans) or the average fish length was less than 190 mm. All of the Great Lakes fish collected for homogenized fillet analysis were species that are commonly consumed by humans (**Table 8.1).** In contrast, some fillet plug samples were not analyzed because they were from fish that were inappropriate for human health objectives based on size or species (**Table 8.2**). Application of the mercury human health fish tissue benchmark to the homogenized fillet composite data from this study identifies the number and percentage of square

miles in the nearshore Great Lakes sampled population that contained fish composite samples with mercury fillet concentrations that are above the mercury human health fish tissue benchmark. Mercury concentration data from analysis of homogenized fish fillet samples are available to download from the NCCA Great Lakes Fish Tissue Studies webpage - https://www.epa.gov/fish-tech/national-coastal-condition-assessment-great-lakes-human-health-fish-tissue-studies. Mercury concentration data from fish fillet tissue plugs are available to download from the NARS data webpage - https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys.

Table 8.1 NCCA 2015 Great Lakes Human Health Fish Composite Sample Species for Homogenized Fillet Analyses (All species were appropriate for human health objectives).

Scientific Name	Common Name	Great Lakes Fillet Samples
Aplodinotus grunniens	Freshwater Drum	11
Catostomus catostomus	Longnose Sucker	10
Catostomus commersonii	White Sucker	9
Coregonus clupeaformis	Lake Whitefish	17
Esox lucius	Northern Pike	1
Ictalurus punctatus	Channel Catfish	3
Lota lota	Burbot	2
Micropterus dolomieu	Smallmouth Bass	11
Morone americana	White Perch	4
Morone chrysops	White Bass	4
Oncorhynchus kisutch	Coho Salmon	3
Oncorhynchus mykiss	Rainbow Trout	9
Oncorhynchus tshawytscha	Chinook Salmon	6
Perca flavescens	Yellow Perch	19
Salmo trutta	Brown Trout	4
Salvelinus namaycush	Lake Trout	26
Sander vitreus	Walleye	13

Table 8.2 NCCA 2015 Fish Plug Species for Mercury Analysis. Checkmarks indicate species that are not appropriate for human health objectives and were not analyzed for mercury.

Scientific Name	Common Name	Number Caught Per Region				Inappropriate for Human Health Objectives
		East Coast	Great Lakes	Gulf Coast	West Coast	
Alosa pseudoharengus	Alewife		5			

Scientific Name	Common Name	Number Caught Per Region				Inappropriate for Human Health Objectives
		East Coast	Great Lakes	Gulf Coast	West Coast	
Ambloplites rupestris	Rock Bass		8			
Ameiurus catus	White Catfish	1				
Ameiurus nebulosus	Brown Bullhead		1			
Anguilla rostrata	American Eel	3				
Aplodinotus grunniens	Freshwater Drum		39			
Ariopsis felis	Hardhead Catfish	9		96		
Bagre marinus	Gafftopsail Catfish			50		
Bairdiella chrysoura	Silver Perch	5		1		
Brevoortia smithi	Yellowfin Menhaden	2				✓[A]
Brevoortia tyrannus	Atlantic Menhaden	3				✓[A]
Caranx crysos	Blue Runner	1				
Caranx hippos	Crevalle Jack	2				
Catostomus catostomus	Longnose Sucker		15			
Catostomus commersonii	White Sucker	1	25			
Centroprostis striata	Black Sea Bass	8		1		
Cheilotrema saturnum	Black Croaker				1	
Citharichthys sordidus	Pacific Sanddab				7	
Citharichthys stigmaeus	Speckled Sanddab				3	
Clupea harengus	Atlantic Herring	1				
Coregonus artedi	Cisco		1			
Coregonus clupeaformis	Lake Whitefish		35			
Cymatogaster aggregata	Shiner Perch				10	
Cynoscion arenarius	Sand Seatrout			7		
Cynoscion nebulosus	Spotted Seatrout	1				
Cynoscion regalis	Weakfish	7				
Cyprinus carpio	Common Carp		11			
Diplectrum formosum	Sand Perch			2		
Diplodus holbrooki	Spottail Pinfish			1		
Dorosoma cepedianum	Gizzard Shad		3			✓[B]
Elops saurus	Ladyfish	2		1		
Embiotoca lateralis	Striped Seaperch				2	
Esox lucius	Northern Pike		3			
*Eucinostomus gula**	Silver Jenny	1				
Fundulus majalis	Striped Killifish	1				✓[C]

Scientific Name	Common Name	Number Caught Per Region				Inappropriate for Human Health Objectives
		East Coast	Great Lakes	Gulf Coast	West Coast	
Genyonemus lineatus	White Croaker				6	
Haemulon plumierii	White Grunt	1		3		
Haemulon sciurus	Bluestriped Grunt			1		
Ictalurus punctatus	Channel Catfish	2	6			
Lagodon rhomboides	Pinfish	9		12		✓[C]
Leiostomus xanthurus	Spot	25		19		
Lepidopsetta bilineata	Rock Sole				3	
Lepisosteus osseus	Longnose Gar	1				
Lepomis gibbosus	Pumpkinseed		1			
Leptocottus armatus	Pacific Staghorn Sculpin				9	✓[B]
Limanda ferruginea	Yellowtail Flounder	1				
Lota lota	Burbot		2			
Lutjanus campechanus	Red Snapper			1		
Lutjanus griseus	Gray Snapper	1		5	1	
Lutjanus synagris	Lane Snapper			4		
Luxilus cornutus	Common Shiner		1			✓[C]
Menidia menidia	Atlantic Silverside	10				✓[C]
Menticirrhus americanus	Southern Kingfish	11				
Menticirrhus littoralis	Gulf Kingfish	1				
Menticirrhus saxatilis	Northern Kingfish	3				
Merluccius bilinearis	Silver Hake	1				
Micropogonias undulatus	Atlantic Croaker	13		18		
Micropterus dolomieu	Smallmouth Bass		35			
Micropterus salmoides	Largemouth Bass		3			
Morone americana	White Perch	20	9			
Morone chrysops	White Bass		7			
Morone saxatilis	Striped Bass	7				
Moxostoma anisurum	Silver Redhorse		1			
Moxostoma macrolepidotum	Shorthead Redhorse		11			
Mustelus canis	Smooth Dogfish	1				
Neogobius melanostomus	Round Goby		1			✓[C]
Oncorhynchus kisutch	Coho Salmon		2			

Scientific Name	Common Name	Number Caught Per Region				Inappropriate for Human Health Objectives
		East Coast	Great Lakes	Gulf Coast	West Coast	
Oncorhynchus mykiss	Rainbow Trout		1			
Opsanus tau	Oyster Toadfish	1				✓[B]
*Orthopristis chrysoptera**	Pigfish			2		
Osmerus mordax	Rainbow Smelt		1			
Paralabrax maculatofasciatus	Spotted Sand Bass				4	
Paralabrax nebulifer	Barred Sand Bass				4	
Paralichthys californicus	California Halibut				21	
Paralichthys dentatus	Summer Flounder	14				
Paralichthys lethostigma	Southern Flounder	1		2		
Perca flavescens	Yellow Perch		57			
Platichthys stellatus	Starry Flounder				3	
*Pleuronectes glacialis**	Arctic Flounder				1	
Pleuronichthys guttulatus	Diamond Turbot				1	
Pogonias cromis	Black Drum			2		
Pollachius virens	Pollock	1				
Pomatomus saltatrix	Bluefish	10				
Pomoxis nigromaculatus	Black Crappie		1			
Prionotus carolinus	Northern Searobin	2				
*Prionotus scitulus**	Leopard Searobin			1		
*Prosopium cylindraceum***	Round Whitefish		5			
Pseudopleuronectes americanus	Winter Flounder	26				
Salmo salar	Atlantic Salmon		1			
Salvelinus namaycush	Lake Trout		9			
Sander vitreus	Walleye		29			
Sciaenops ocellatus	Red Drum			2		
Scomber scombrus	Atlantic Mackerel	9				
Scophthalmus aquosus	Windowpane	1				
Stenotomus chrysops	Scup	45				
Tautogolabrus adspersus	Cunner	6				
Trinectes maculatus	Hogchoker	2				✓[C]
Urophycis chuss	Red Hake	1				
Zoarces americanus	Ocean Pout	1				

* Although small, may be eaten by humans.
**Although not typically targeted, may be eaten by humans.
[A] Used commercially for fish meal, fish oil, or bait; typically, not directly consumed by humans.
[B] Consumption by humans extremely rare.
[C] Small forage (and/or bait) species, not consumed by humans.

8.3 PCB ANALYSIS AND HUMAN HEALTH FISH TISSUE BENCHMARKS

Fish fillet tissue samples prepared from the 152 fish composite samples collected at Great Lakes nearshore sites were analyzed for PCBs using EPA Method 1668C (USEPA 2010). This method utilizes approximately 10 g of fillet tissue for analysis and provides results for the full set of 209 PCB congeners. The total PCB concentration for each sample was determined by summing the results for any of the 209 congeners that were detected, using zero for any congeners that were not detected in the sample.

EPA used a 49 ppb human health benchmark for total PCB noncancer effects and a 12 ppb human health benchmark for total PCB cancer effects to report 2015 Great Lakes Human Health Fish Fillet Tissue Study data in the NCCA 2015 Final Report. Both of these benchmarks were derived using a fish consumption rate of 32 g/day[21]. This nutrition-based fish consumption rate of 32 g/day better reflects the role and purpose of fish advisory programs because it does not include data for non-consumers and is also consistent with the rate used in fish advisory programs across the Great Lakes. EPA acknowledges this rate does not reflect "high frequency consumers" such as subsistence fishers or those who eat several meals of fish per week, which often includes individuals in underserved communities. In an effort to provide information to state, territorial, or tribal programs with populations of frequent fish consumers, EPA has provided **Table 8.3** that includes estimated benchmark exceedances for PCBs using fish consumption rates that are more typical of these populations. This table also includes results for the human health benchmarks based on a 32 g/day fish consumption rate for comparison.

Application of the PCB benchmarks representing average fish consumers for the Great Lakes area and two other sets of PCB benchmarks, described below for high frequency fish consumers, to the total PCB fillet data identifies the number and percentage of square miles in the Great Lakes nearshore sampled population containing fish with total PCB fillet concentrations that are above each PCB human health fish tissue benchmark. Data on exceedances of the PCB human health benchmarks are provided in **Table 8.3**. In addition to the benchmarks representing average fish consumers, the first set of PCB benchmarks for "high frequency consumers" is based on a fish consumption rate of 142 g/day, which is described in the EPA 2000 Human Health Methodology (USEPA 2000b). The second set of PCB benchmarks for "high frequency consumers" is based on a fish consumption rate of 175 g/day, which has been used by EPA and some states for high frequency consumers or subsistence fishers in the Pacific Northwest.

[21] Since EPA does not currently have a fish tissue-based water quality criterion for PCBs, EPA has selected to use the equations found in its *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories* (USEPA 2000) with updated body weights in EPA's *Exposure Factors Handbook* (USEPA 2011) and a nutritionally focused fish consumption rate consistent with the U.S. Department of Agriculture and Department of Health and Human Services' *Dietary Guidelines for Americans, 2020-2025* of 32 grams/day (equivalent to one eight-ounce meal of fish and shellfish per week).

Table 8.3 Percentages of Total PCB Human Health Fish Tissue Benchmark Exceedances

Chemical-Specific Human Health Fish Tissue (HH) Benchmarks	% Assessed Nearshore Area with Fish Fillet Concentrations Above Total PCB Noncancer HH Benchmarks	% Assessed Nearshore Area with Fish Fillet Concentrations Above Total PCB Cancer HH Benchmarks
Total PCB Noncancer 49 ppb HH Benchmark (32 g/day FCR*)	53%	
Total PCB Noncancer 11 ppb HH Benchmark (142 g/day FCR)	81%	
Total PCB Noncancer 9.1 ppb HH Benchmark (175 g/day FCR)	88%	
Total PCB Cancer 12 ppb HH Benchmark (32 g/day FCR)		79%
Total PCB Cancer 2.8 ppb HH Benchmark (142 g/day FCR)		100%
Total PCB Cancer 2.3 ppb HH Benchmark (175 g/day FCR)		100%

* FCR = Fish consumption rate

PCB concentration data from analysis of homogenized fish fillet samples are available to download from the NCCA Great Lakes Fish Tissue Studies webpage - https://www.epa.gov/fish-tech/national-coastal-condition-assessment-great-lakes-human-health-fish-tissue-studies.

8.4 PFAS Analysis And Human Health Fish Tissue Benchmark

Fish fillet tissue samples prepared from the 152 fish composite samples collected at Great Lakes nearshore sites were analyzed for 13 per- and polyfluoroalkyl substances (PFAS), including perfluorooctane sulfonate or PFOS, which is the most commonly detected PFAS in freshwater fish. There are no standard EPA methods for PFAS analysis of tissue samples, so the samples were analyzed by SGS AXYS Analytical Services, Ltd. using a proprietary procedure developed by their laboratory in Sidney, British Columbia, Canada. That procedure, which utilizes approximately 2 g of fillet tissue for analysis, uses high performance liquid chromatography with tandem mass spectrometry (HPLC-MS/MS) and applies the technique known as isotope dilution to determine the concentration of each of the 13 PFAS.

EPA used a 46 ppb human health benchmark for PFOS to report 2015 Great Lakes Human Health Fish Fillet Tissue Study data in the NCCA 2015 Final Report. This benchmark was derived using a fish

consumption rate of 32 g/day[22]. This nutrition-based fish consumption rate of 32 g/day better reflects the role and purpose of fish advisory programs because it does not include data for non-consumers and is also consistent with the rate used in fish advisory programs across the Great Lakes. EPA acknowledges this rate does not reflect "high frequency consumers" such as subsistence fishers or those who eat several meals of fish per week, which often includes individuals in underserved communities. In an effort to provide information to state, territorial, or tribal programs with populations of frequent fish consumers, EPA has provided **Table 8.4** that includes estimated benchmark exceedances for PFOS using fish consumption rates that are more typical of these populations. This table also includes results for the human health benchmark based on a 32 g/day fish consumption rate for comparison.

Application of the PFOS benchmark representing average fish consumers for the Great Lakes area and two other PFOS benchmarks, described below for frequent fish consumers, to the PFOS fillet data identifies the number and percentage of square miles in the Great Lakes nearshore sampled population containing fish with PFOS fillet concentrations that are above each PFOS human health fish tissue benchmark. Data on the exceedance of this human health benchmark for average fish consumers are provided in **Table 8.4**. In addition to the benchmark representing average fish consumers, the first PFOS benchmark for "high frequency consumers" is based on a fish consumption rate of 142 g/day, which is described in the EPA 2000 Human Health Methodology (USEPA 2000b). The second PFOS benchmark for "high frequency consumers" is based on a fish consumption rate of 175 g/day, which has been used by EPA and some states for high frequency consumers or subsistence fishers in the Pacific Northwest.

Table 8.4 Percentages of PFOS Human Health Fish Tissue Benchmark Exceedances

Chemical-Specific Human Health Fish Tissue (HH) Benchmarks	% Assessed Great Lakes Nearshore Area with Fish Fillet Concentrations Above PFOS HH Benchmarks
PFOS 46 ppb HH Benchmark (32 g/day FCR*)	5%
PFOS 11 ppb HH Benchmark (142 g/day FCR)	52%
PFOS 8.6 ppb HH Benchmark (175 g/day FCR)	67%

* FCR = Fish consumption rate

Summary statistics, including the number of detections for each of the 13 PFAS analyzed for the 2015 Great Lakes Human Health Fish Fillet Tissue Study, are provided in **Table 8.5**. PFAS concentration data from analysis of homogenized fish fillet samples are available to download from the NCCA Great Lakes Fish Tissue Studies webpage - https://www.epa.gov/fish-tech/national-coastal-condition-assessment-great-lakes-human-health-fish-tissue-studies.

[22] Since EPA does not currently have a fish tissue-based water quality criterion for PFOS, EPA has selected to use the equations found in its *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories* (USEPA 2000) with updated body weights in EPA's *Exposure Factors Handbook* (USEPA 2011) and a nutritionally focused fish consumption rate consistent with the U.S. Department of Agriculture and Department of Health and Human Services' *Dietary Guidelines for Americans, 2020-2025* of 32 grams/day (equivalent to one eight-ounce meal of fish and shellfish per week).

Table 8.5 2015 Great Lakes Human Health Fish Fillet Tissue Study PFAS Fillet Composite Data

Chemical	Number of Detections	Detection Frequency (%)	MDLs (ppb)	Measured Minimum Concentration (ppb) [a]	Weighted Median Concentration (ppb) [b]	Measured Maximum Concentration (ppb) [a]
PFBA	0	0	0.21	<MDL	<MDL	<MDL
PFBS	1	<1	0.36	<MDL	<MDL	0.36
PFPeA	2	1	0.24	<MDL	<MDL	1.13
PFHxA	0	0	0.13	<MDL	<MDL	<MDL
PFHxS	1	<1	0.63	<MDL	<MDL	0.96
PFHpA	0	0	0.13	<MDL	<MDL	<MDL
PFOA	22	14	0.23	<MDL	<MDL	1.93
PFOS	152	100	0.52	0.52	11.32	64.4
PFOSA	56	37	0.13	<MDL	<MDL	3.73
PFNA	119	78	0.12	<MDL	0.33	9.32
PFDA	134	88	0.16	<MDL	0.78	6.49
PFUnA	139	91	0.17	<MDL	1.05	15.00
PFDoA	123	81	0.09	<MDL	0.35	3.65

MDL = Method Detection Limit

[a] Observed data (minimum and maximum concentrations) measured in 152 Great Lakes fish fillet samples.

[b] Statistical estimates of the median fish fillet composite concentrations for the nearshore Great Lakes sampled population of 6,862 square miles.

8.5 REFERENCES

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9 ENTEROCOCCI INDICATOR

The EPA developed and validated a molecular testing method employing quantitative polymerase chain reaction (qPCR) as a rapid approach for the detection of enterococci in recreational water (USEPA 2015). NCCA used this method to estimate the presence and quantity of these fecal indicator bacteria in the nation's coastal area. The statistical benchmark value of 1280 calibrator cell equivalents (CCE)/100 mL from EPA's 2012 Recreational Water Quality Criteria document (RWQC) was then applied to the enterococci data to assess the recreational condition of coastal waters.

9.1 FIELD COLLECTION

To collect enterococci samples, field crews took a water sample with a gloved hand or a pole-dipper at a depth of 0.5 m using a sterile 250 mL bottle. In addition to collecting the sample, crews looked for signs of disturbance that would contribute to the presence of fecal contamination to the waterbody. Following collection, crews added sodium thiosulfate and placed the sample in a cooler on wet ice. Within 6 hours of collection, two 50 mL volumes were filtered and the filters were frozen and shipped to the lab on dry ice. A sterile phosphate buffer solution (PBS) blank was also filtered at revisit sites during visit 1 and visit 2.

9.2 LAB METHODS

The sample collections and the laboratory method followed EPA's Enterococcus qPCR Method 1609.1 (USEPA 2015; available on-line at https://www.epa.gov/cwa-methods/other-clean-water-act-test-methods-microbiological). Method 1609.1 describes a quantitative polymerase chain reaction (qPCR) procedure for the detection of DNA from enterococci bacteria in ambient water matrices based on the amplification and detection of a specific region of the large subunit ribosomal RNA gene (lsrRNA, 23S rRNA) from these organisms. This method uses an arithmetic formula (the comparative cycle benchmark (CT) method; Applied Biosystems, 1997) to calculate the ratio of Enterococcus lsrRNA gene target sequence copies (TSC) recovered in total DNA extracts from the water samples relative to those recovered from similarly prepared extracts of calibrator samples containing a consistent, pre-determined quantity of Enterococcus cells. Mean estimates of the absolute quantities of TSC recovered from the calibrator sample extracts were then used to determine the quantities of TSC in the water samples and then converted to CCE values as described in the section below. To normalize results for potential differences in DNA recovery, monitor signal inhibition or fluorescence quenching of the PCR analysis caused by a sample matrix component, or detect possible technical error, CT measurements of sample processing control (SPC) and internal amplification control (IAC) target sequences were performed as described in Method 1609.1. The qPCR method is appropriate for both marine and freshwater environments as described in the 2012 Recreational Water Quality Criteria guidelines.

9.2.1 *Calibration*

Estimates of absolute TSC recoveries from the calibrator samples were determined from standard curves using EPA-developed plasmid DNA standards of known TSC concentrations as described in Method 1609.1. Estimates of TSC recovered from the test samples were determined by the comparative cycle benchmark (CT) method, as also described in Method 1609.1. Before applying the EPA benchmarks to the qPCR data, it was necessary to convert the TSC estimates to CCE values. The standardized approach developed for this conversion is to assume 15 TSC/CCE (USEPA 2015). This approach allows the CCE values to be directly compared to the EPA RWQC values (Haugland et al. 2015).

9.3 ANALYSIS OF ENTEROCOCCI CONCENTRATIONS

For the data analysis of the enterococci measurements determined by Method 1609.1, EPA used benchmarks as defined and outlined in the 2012 RWQC document (USEPA 2012). The document contains the EPA's ambient water quality criteria recommendations for protecting human health in marine and freshwaters. Enterococci CCE/100 mL values were compared to the EPA statistical benchmark value of 1280 CCE/100 mL[23] (USEPA 2012). The enterococci concentration data are available to download from the NARS data webpage - https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys.

9.4 REFERENCES

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[23] Estimated Illness Rate (NGI): 32/1000 primary contact recreators. See USEPA 2012 for more information on additional NGI statistical threshold values for the qPCR method.

10 MICROCYSTINS

Microcystins comprise a group of toxins produced by various cyanobacteria, or blue-green algae. Microcystin exposure risk to humans is elevated when an overabundance of cyanobacteria occurs in recreational surface water, especially during algal bloom events. Human exposure to microcystins and associated cyanobacterial toxins may produce adverse symptoms ranging in severity from nausea, diarrhea, weakness, to liver and kidney failure, potentially cancer, and even death in severe cases (Chorus and Bartram 1999; Giannuzzi et al. 2011; Meneely and Elliott 2013). For NCCA, microcystin concentrations were evaluated against the EPA recommended swimming advisory level of 8 µg/L (USEPA 2019).

10.1 FIELD AND LABORATORY METHODS

Water samples were collected at a depth of 0.5 m using a water collection device (e.g., a Niskin bottle) for microcystin analysis from all estuarine and freshwater sites. Water was transferred to a 500 mL bottle, kept on ice, and then stored frozen until analysis.

Samples were lysed by three freeze-thaw cycles and filtered with 0.45-micron syringe filters, then analyzed using the Abraxis Microcystins-ADDA ELISA Kit. Brackish water (salinity greater than 3.5%) samples underwent further extraction to remove salts and eliminate adverse performance effects on the immunoassay (per the Abraxis Bulletin R041112: Microcystins in Brackish Water or Seawater Sample Preparation). For freshwater samples, the procedure's reporting limit is 0.15 µg/L, although, theoretically, the procedure can detect, but not quantify, microcystins concentrations as low as 0.10 µg/L. For brackish samples (samples with greater than 3.5 ppt salinity), the procedure's reporting limit is 0.263 µg/L, although, theoretically, the procedure can detect, but not quantify, microcystins concentrations as low as 0.175 µg/L.

10.2 ANALYSIS OF MICROCYSTIN CONCENTRATIONS

Microcystin concentrations were evaluated against the EPA recommended swimming advisory level of 8 µg/L (USEPA 2019). Microcystin concentration data are available to download from the NARS data webpage: https://www.epa.gov/national-aquatic-resource-surveys/data-national-aquatic-resource-surveys.

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11 FROM ANALYSES TO RESULTS

11.1 EXTENT ESTIMATION AND ASSESSMENT

A major goal of the National Aquatic Resource Surveys is to assess the condition of the nation's waters and changes over time. The following discussion describes the condition class assignments and calculations used in EPA's assessments. In the main report, results were calculated for the Great Lakes population and the Estuarine population separately.

11.1.1 *Condition Classes*

Condition classes were assigned to each site for each indicator based on the analysis described in the individual indicator chapters. The condition classes were then used to estimate the extent, change, and trend in condition across the sampled population. Only sites that were included in the probability design and were evaluated as "Target_Sampled" were used to calculate statistics. If sites were visited twice during NCCA 2015, only data from one site visit[24] were used to calculate condition estimates.

11.1.2 *Estimating the Extent for Each Condition*

The estimated extent $\hat{E}$ measures the prevalence of a particular condition k (good, fair, or poor). For each Y indicator, $\hat{E}$ provides an estimate of the square miles of coast in that condition.

The extent is estimated in two steps for each condition. The first step classifies each statistically selected site into one of the three conditions for each *Y*. The second step estimates the miles using the estimated survey weights $\hat{w}_i$ for each site *i*, classified into condition *k*. Applying weights to the data allows inferences to be made about all coastal areas in the target population, not just the sites from which physical samples were collected. Each sampled site is assigned an estimated weight for the number of square miles that it represents. For example, one site might represent 200 square miles of coastal area in the entire target population, and thus, its sample weight was $\hat{w}_{Yki} = 200$. Equation 11-1 (**below**) shows the estimation of extent ($\hat{E}_{Y_k}$) for condition class *k* for each *Y*.

$$\hat{E}_{Y_k} = \sum_i w_{Y_{k_i}} \qquad \textbf{Equation 11-1}$$

[24] For all but one or two sites, "Visit 1", which is denoted as VISIT_NO = "1" in the data files, was used to calculate condition estimates. If quality purposes required use of VISIT_NO = "2" data for condition estimates, that information is noted in the data files.

11.2 ANALYSES

Estimates of each condition category were computed using spsurvey (Kincaid and Olsen, 2016). The margin of error for national estimates was +/- 5% and for ecoregional estimates was +/- 15% with 95% confidence, meeting the objectives outlined in Section 3.1.

One of the objectives of the NCCA is to track changes over time. Previously, EPA and partners reported on the condition of coastal area in the NCCA 2010 and in the National Coastal Assessment (NCA) in 2005-2006 and 1991-2001. The 2015 report presents the difference in percentage points of coastal square miles in "good," "fair," and "poor" condition between the NCA 2005-2006 (data from 2005 and 2006 were combined into the 2005 time frame for the 2015 report), NCCA 2010 and NCCA 2015. Comparisons with earlier years can be viewed in the online data dashboard (https://coastalcondition.epa.gov/).

Benchmarks and analyses that were modified in NCCA 2015 (i.e. M-AMBI, Ecological Fish Tissue Contaminants) were applied to previous survey datasets in order for data to be directly comparable for the change analyses. Change analysis was not conducted for mercury in fish plugs, enterococci, or microcystin because these indicators were not included in the 2005 or 2010 surveys.

Change analysis was conducted through the use of the spsurvey 3.3 package in R (Kincaid and Olsen, 2016). Within the GRTS (Generalized Random Tessellation Stratified) survey design, change analysis can be conducted on continuous or categorical variables. When using categorical variables, change is estimated by the difference in category estimates from the two surveys. Category estimates were defined as the estimated proportion of values in each category (i.e. good, fair, and poor categories). Change between the two years was statistically significant when the resulting error bars around the change estimate did not cross zero.

11.3 TREND ANALYSIS

Trend estimates for "good" condition were calculated for the estuarine population using linear regressions. Values of 2005, 2010, and 2015 were used to represent the three design cycles, respectively, which provided an equally spaced set of values. Trend estimates for good condition from 2005 to 2015 can be viewed in online data dashboard (https://coastalcondition.epa.gov/).

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APPENDIX A. ECOLOGICAL FISH TISSUE CONTAMINANT INDEX BACKGROUND INFORMATION

The following sub-sections summarize the laboratory-based endpoints for each group of receptors chosen for each contaminant. The section describes the conversion factor applied if necessary as well as the body weight scaling for each group of receptors using the formula and scaling factors presented above. The laboratory-based endpoints presented below were those chosen to be used in the derivation of the fish tissue screening values. Tables A.1.1 through A.1.13 contain all laboratory endpoints extracted from the available scientific literature (search: 2011-2012) for each contaminant of concern measured for the NCCA.

A.1 LABORATORY ENDPOINTS FOR NCCA EFTCI CONTAMINANTS OF CONCERN

A.1.1 *Arsenic, Inorganic*

USFWS (1964) reported a chronic NOAEL and LOAEL for mallard mortality of 5.1 and 12.8 mg/kg-bw/d which were converted to wildlife avian NOAEL and LOAEL of 3.39 and 8.51 mg/kg-bw/d. A laboratory chronic NOAEL for mouse reproduction of 0.126 mg/kg-bw/d was reported (Sample et al., 1996). A conversion factor of 5 was applied to extrapolate a LOAEL of 0.63 mg/kg-bw/d. The wildlife mammalian TRVs were calculated using these values and resulted in a freshwater mammalian TRV NOAEL and LOAEL of 0.11 and 0.53 mg/kg-bw/d, respectively, and a marine mammalian NOAEL and LOAEL of 0.080 and 0.40 mg/kg-bw/d, respectively. Pedlar et al. (2002) reported a sub-chronic NOEC of 119.6 mg/kg food for growth of lake whitefish. Lake whitefish weighing 326 g, were fed 0.5% of their body weight or 1.63 g food. As reported, 1 kg of food contained 119.6 mg arsenic, each fish was fed 0.20 mg/0.326 kg-bw/d, or 0.60 mg/kg-bw/d. The fish were fed three times a week, so the daily dosage was 0.26 mg/kg-bw/d. The sub-chronic NOAEL was extrapolated to a chronic NOAEL by applying a conversion factor of 0.1. The NOAEL, 0.026 mg/kg-bw/d was used to extrapolate a LOAEL by applying a conversion factor of 5, resulting in a LOAEL of 0.13 mg/kg-bw/d. The laboratory TRVs were converted to wildlife TRVs of 0.027 and 0.14 mg/kg-bw/d for freshwater fish and 0.060 and 0.30 mg/kg-bw/d for marine species. See Table A.1.1.

A.1.2 *Cadmium*

A laboratory chronic NOAEL and LOAEL of 1.45 and 20 mg/kg-bw/d, respectively, were reported by Sample et al. (1996) for mallard reproduction. The laboratory TRVs were extrapolated to avian wildlife TRVs of 0.94 and 12.93 mg/kg-bw/d, respectively. ATSDR (2008) reported a cadmium NOAEL of 0.75 mg/kg-bw/d for reproduction in the dog. A conversion factor of 5 was applied to the NOAEL to extrapolated a LOAEL of 3.75 mg/kg-bw/d. These values were converted to mammalian wildlife TRVs of 0.89 mg/kg-bw/d and 4.46 mg/kg-bw/d

for freshwater mammals and 0.67 mg/kg-bw/d and 3.37 mg/kg-bw/d for marine mammals. Szczerbik et al. (2006) reported a chronic NOAEL and LOAEL of 1 and 10 mg cadmium/g food, respectively, for growth in the carp. As reported 0.56 g carp were fed 2% of their body weight per day, which is equal to 0.0112 g food containing the reported concentrations of cadmium. Therefore, the NOAEL was 20 mg/kg-bw/d and 200 mg/kg-bw/d was the LOAEL. The laboratory TRVs were converted to wildlife TRVs of 76.34 and 763.49 mg/kg-bw/d for freshwater fish and 168.0 and 1680.0 mg/kg-bw/d for marine species. See Table A.1.2.

A.1.3 *Chlordane, Total*

Wiemeyer (1996) reported a chronic NOAEL and LOAEL for mallard reproduction of 0.8 and 4.0 mg/kg-bw/d which were converted to wildlife avian NOAEL and LOAEL of 0.53 and 2.66 mg/kg-bw/d. A laboratory chronic NOAEL and LOAEL for mouse reproduction of 4.58 and 9.16 mg/kg-bw/d, respectively, were reported (Sample et al., 1996). The wildlife mammalian TRVs were calculated using these values and resulted in a freshwater mammalian TRV NOAEL and LOAEL of 3.85 and 7.69 mg/kg-bw/d, respectively, and a marine mammalian TRV NOAEL and LOAEL of 2.91 and 5.81 mg/kg-bw/d, respectively. Dietary exposure of fish to chlordane was not available in the literature and therefore represents an uncertainty. See Table A.1.3.

A.1.4 *DDT, Total*

A laboratory chronic NOAEL and LOAEL of 0.3 and 3.0 mg/kg-bw/d, respectively, were reported by USEPA (1995) for reproduction in the bald eagle. The laboratory TRVs were extrapolated to avian wildlife TRVs of 0.15 and 1.47 mg/kg-bw/d. Sample et al. (1996) reported a DDT NOAEL of 0.8 mg/kg-bw/d and a LOAEL of 4.0 mg/kg-bw/d for reproduction in the rat. These values were converted to mammalian wildlife TRVs of 0.78 mg/kg-bw/d and 3.89 mg/kg-bw/d for freshwater mammals and 0.59 mg/kg-bw/d and 2.94 mg/kg-bw/d for marine mammals. A chronic NOEC of 1 mg/kg-bw/week for the rainbow trout was reported (Macek et al., 1970) and converted to a daily dosage of 0.143 mg/kg-bw/d. A conversion factor of 5 was applied to derive the LOAEL, 0.715 mg/kg-bw/d. The laboratory NOAEL and LOAEL were converted to a freshwater and marine fish NOAEL and LOAEL of 0.28 and 1.42 mg/kg-bw/d, respectively, for freshwater fish and 0.62 and 3.12 mg/kg-bw/d, respectively, for marine fish, respectively. See Table A.1.4.

A.1.5 *Dieldrin*

Sample et al. (1996) reported a chronic NOAEL and LOAEL for the barn owl of 0.08 and 0.39 mg/kg-bw/d which were converted to wildlife avian NOAEL and LOAEL of 0.062 and 0.30 mg/kg-bw/d. A laboratory chronic LOAEL for the dog of 0.14 mg/kg-bw/d reported by ATSDR (2002b) was used to convert a chronic NOAEL by applying a conversion factor of 0.2, resulting in a laboratory mammalian NOAEL of 0.028 mg/kg-bw/d. The wildlife mammalian

TRVs were calculated using these values and resulted in a freshwater mammalian TRV NOAEL and LOAEL of 0.033 and 0.17 mg/kg-bw/d, respectively, and a marine mammalian TRV NOAEL and LOAEL of 0.025 and 0.13 mg/kg-bw/d, respectively. Argyle et al. (1975) reported a laboratory NOAEL of 0.8 μg Dieldrin/g food. As reported, 3.0 g fish were fed 4.2% of their body weight/day for a total of 0.12 kg food containing 0.8 μg Dieldrin/g. This is equivalent to 0.0336 mg/kg-bw/d. Because the fish were fed only 5 days a week, the laboratory chronic NOAEL was calculated as 0.024 mg/kg-bw/d. The reported LOAEL was 4 μg Dieldrin/g food which was also converted to 0.12 mg/kg-bw/d. Extrapolation of fish TRVs resulted in freshwater fish NOAEL and LOAEL of 0.065 and 0.33 mg/kg-bw/d, respectively, and marine fish NOAEL and LOAEL of 0.14 and 0.72 mg/kg-bw/d, respectively for wildlife species. See Table A.1.5.

A.1.6 *Endrin, Total*

Sample et al. (1996) reported a chronic NOAEL and LOAEL of 0.02 and 0.1 mg/kg/d for reproduction in the screech owl, respectively. The laboratory TRVs were converted to avian wildlife TRVs of 0.019 and 0.099 mg/kg/d. A chronic NOAEL and LOAEL of 0.18 and 0.92 mg/kg-bw/d were reported for reproduction in the mouse (Sample et al., 1996). These values were converted to mammalian wildlife TRVs of 0.15 and 0.77 mg/kg-bw/d for freshwater species. For marine mammals, wildlife TRVs were calculated to be 0.11 and 0.58 mg/kg-bw/d. A chronic NOAEL of 0.04 mg/kg-bw/d was reported by Argyle et al. (1973). A conversion factor of 5 was applied to calculate a chronic LOAEL of 0.2 mg/kg-bw/d. The calculated wildlife TRVs for freshwater fish were 0.16 and 0.78 mg/kg-bw/d. The calculated wildlife TRVs for marine species were 0.34 and 1.72 mg/kg-bw/d. See Table A.1.6.

A.1.7 *Endosulfan, Total*

A laboratory chronic NOAEL and LOAEL of 10 and 50 mg/kg-bw/d, respectively, were reported by Sample et al. (1996) for reproduction in the gray partridge. The laboratory TRVs were extrapolated to avian wildlife TRVs of 7.99 and 39.93 mg/kg-bw/d. ATSDR (2000) reported NOAEL of 1.0 mg/kg-bw/d and a LOAEL of 5.0 mg/kg-bw/d for systemic effects of endosulfan in dogs. These values were converted to mammalian wildlife TRVs of 1.19 mg/kg-bw/d NOAEL and 5.95 mg/kg-bw/d LOAEL for freshwater mammals and 0.90 mg/kg-bw/d NOAEL and 4.50 mg/kg-bw/d LOAEL for marine mammals. A chronic NOAEL of 0.24 μg/kg-bw/d and a chronic LOAEL of 0.5 μg/kg-bw/d for the Atlantic salmon was reported (Lundebye et al., 2010). The reported Atlantic salmon NOAEL/LOAEL were converted to a freshwater and marine fish NOAEL/LOAEL of 0.26 and 0.60 μg/kg-bw/d for freshwater fish and 0.60 and 1.31 μg/kg-bw/d for marine fish, respectively. See Table A.1.7.

A.1.8 *Heptachlor*

The LD_{50} for survival in the bobwhite quail was reported to be 125 mg/kg (USEPA, 1972). A conversion factor of 0.01 was applied to calculate a chronic NOAEL of 1.25 mg/kg-bw/d. A conversion factor of 5 was applied to the NOAEL to calculate a chronic LOAEL of 6.25 mg/kg-bw/d. The laboratory TRVs were converted to avian wildlife TRVs of 1.16 and 5.79 mg/kg-bw/d. Sample et al. (1996) reported a chronic NOAEL and LOAEL for reproduction in the mink of 0.2 and 1 mg/kg-bw/d, respectively. The laboratory TRVs were converted to mammalian wildlife TRVs of 0.21 and 1.037 mg/kg-bw/d for freshwater species. For marine mammals, wildlife TRVs were calculated to be 0.16 and 0.78 mg/kg-bw/d. Andrews et al. (1996) reported a laboratory NOAEL of 3.57 mg/kg-bw/d and a chronic LOAEL of 7.14 mg/kg-bw/d. Extrapolation of fish TRVs resulted in freshwater fish NOAEL and LOAEL of 8.09 and 16.2 mg/kg-bw/d, respectively, and marine fish NOAEL and LOAEL of 17.8 and 35.6 mg/kg-bw/d, respectively for wildlife species. See Table A.1.8.

A.1.9 *Hexachlorobenzene*

The chronic NOAEL and LOAEL for reproduction in the Japanese quail were reported as 0.11 and 0.57 mg/kg-bw/d (Coulston and Kolbye, 1994; Terretox, 2002). The laboratory TRVs were extrapolated to avian wildlife TRVs of 0.11 and 0.55 mg/kg-bw/d. Laboratory TRVs of 1 and 2 mg/kg-bw/d were reported by ATSDR (2002a) for reproduction in the rat. The calculated wildlife chronic NOAEL and LOAEL for freshwater mammals were 0.97 and 1.95 mg/kg-bw/d, respectively. For marine mammalian species, the calculated TRVs were 0.74 and 1.47 mg/kg-bw/d. Woodburn et al. (2008) reported a subchronic NOAEL of 327 ng HCB/g food for growth in the channel catfish. As reported, 4.0 g catfish were fed 2.1% of their body weight per day for a total of 0.084 g of food containing 0.000327 mg HCB per day. Therefore, the sub-chronic NOAEL was 0.00685 mg/kg/d. This value was converted to a chronic NOAEL of 0.00069 mg/kg/d using a conversion factor of 0.1. By applying a conversion factor of 5 to the NOAEL, the chronic LOAEL was calculated to be 0.0034 mg/kg/d. The laboratory TRVs were converted to wildlife TRVs of 0.0018 and 0.0088 mg/kg-bw/d for freshwater fish and 0.0039 and 0.019 mg/kg-bw/d for marine species. SeeTable A.1.9

A.1.10 *Lindane*

The chronic NOAEL and LOAEL for reproduction in the Japanese quail were reported as 0.56 and 2.25 mg/kg-bw/d, respectively (Sample et al., 1996). These values were converted to an avian wildlife chronic NOAEL and LOAEL of 0.54 and 2.19 mg/kg-bw/d, respectively. Sample et al. (1996) reported chronic endpoints of 8 and 40 mg/kg/d for reproduction in the rat. These values were converted to a mammalian wildlife chronic NOAEL and LOAEL of 7.79 and 38.93 mg/kg-bw/d, respectively, for freshwater species. The calculated wildlife TRVs for marine mammals were 5.88 and 29.41 mg/kg-bw/d. Cossarini-Dunier et al. (1987) reported chronic NOAEL of 1.0 g lindane/kg food for immune response in the carp. As reported, 60 g carp were

fed 1% of their body weight each day for a total of 0.0006 kg food containing 1.0 mg lindane/kg. Therefore, the endpoint was 0.6 mg/0.06 kg-bw/d, resulting in a calculated laboratory chronic NOAEL of 10 mg/kg-bw/d. A conversion factor of 5 was applied to extrapolate a chronic LOAEL of 50 mg/kg-bw/d. The laboratory TRVs were converted to wildlife TRVs of 14.99 and 74.95 mg/kg-bw/d for freshwater fish, and 32.98 and 164.91 mg/kg-bw/d for marine fish. See Table A.1.10.

A.1.11 *Mercury (Methylmercury)*

Heinz and Locke (1976) reported a chronic NOAEL and LOAEL for mallard reproduction of 0.03 and 0.18 mg/kg-bw/d which were converted to wildlife avian NOAEL and LOAEL of 0.020 and 0.12 mg/kg-bw/d. A laboratory chronic NOAEL and LOAEL for rat reproduction of 0.032 and 0.16 mg/kg-bw/d were reported (Sample et al., 1996). The wildlife mammalian TRVs were calculated using these values and resulted in a freshwater mammalian TRV NOAEL and LOAEL of 0.031 and 0.16 mg/kg-bw/d, respectively, and a marine mammalian NOAEL and LOAEL of 0.024 and 0.12 mg/kg-bw/d, respectively. Berntssen et al. (2003) reported a chronic NOAEL and LOAEL of 4.23 and 8.31 mg methylmercury chloride/kg food for brain pathology in the Atlantic salmon. As reported, 10.8g Atlantic salmon were fed 1.6% of their body weight per day. Therefore, the endpoint was 0.068 mg/kg/d NOAEL and 0.13 mg/kg-bw/d LOAEL. The laboratory TRVs were converted to wildlife TRVs of 0.14 and 0.28 mg/kg-bw/d for freshwater fish and 0.31 and 0.62 mg/kg-bw/d for marine species. See Table A.1.11.

A.1.12 *Mirex*

Hyde et al. (1973) reported chronic NOEC and LOEC of 1 and 100 mg mirex/kg food for reproduction in the mallard. A reference body weight of 1 kg and a reference food ingestion rate of 100 g/d (Sample et al., 1996) were used to convert the dietary concentrations to units of mg/kg-bw/d. Therefore, the chronic NOAEL and LOAEL for reproduction in the mallard duck are 0.01 and 1 mg/kg-bw/d, respectively. The laboratory TRVs were converted to avian wildlife TRVs of 0.0066 and 0.66 mg/kg-bw/d. A mammalian chronic NOAEL and LOAEL of 0.07 and 0.7 mg/kg-bw/d, respectively, were reported by NTP (1990) for liver and thyroid effects in the rat. These laboratory TRVs were converted to freshwater mammalian wildlife TRVs of 0.064 and 0.64 mg/kg-bw/d and marine mammalian wildlife TRVs of 0.048 and 0.48 mg/kg-bw/d. A chronic NOAEL of 0.3 mg/kg-bw/d for growth in the brook trout was reported by Skea et al. (1981). A conversion factor of 5 was applied to this value to obtain a chronic LOAEL of 1.5 mg/kg-bw/d. These values were converted to wildlife NOAELs and LOAELs of 0.40 and 1.98 mg/kg-bw/d for freshwater fish, respectively, and 0.87 and 4.35 mg/kg-bw/d, respectively for marine fish. See Table A.1.12.

A.1.13 *Polychlorinated Biphenyls (PCBs), Total*

Polychlorinated biphenyls typically exist as conglomerates of multiple aroclors (i.e., Aroclor 1242, -1248, -1254, -1260, etc.). The aroclor number with respect to PCBs is an indication of the percent of chlorination (i.e., Aroclor-1254 has 54% chlorination). Using the toxic effects of Aroclor-1254 as a surrogate for PCBs should yield a conservative estimate because toxic effects are thought to be related to the degree of chlorination (Exponent, 2010). A laboratory chronic NOAEL and LOAEL of 0.18 and 1.8 mg/kg-bw/d, respectively, were reported by Sample et al. (1996) for Aroclor-1254 and ring-necked pheasant reproduction. The laboratory TRVs were extrapolated to avian wildlife TRVs of NOAEL = 0.12 and LOAEL= 1.20 mg/kg/d. Sample et al. (1996) reported an aroclor-1254 NOAEL of 0.068 mg/kg-bw/d and a LOAEL of 0.68 mg/kg-bw/d for reproduction in the oldfield mouse. These values were converted to mammalian wildlife TRVs of 0.055 mg/kg-bw/d NOAEL and 0.55 mg/kg-bw/d LOAEL for freshwater mammals and 0.041 mg/kg-bw/d NOAEL and 0.41 mg/kg-bw/d LOAEL for marine mammals. Leatherland and Sonstegard (1980) reported a subchronic LOEC of 50 mg/kg food for liver and thyroid effects in rainbow trout. As reported, 50 g trout were fed 2% of their body weight per day, or a dosage of 1 mg/kg-dw/d. The sub-chronic LOAEL was converted to a chronic NOAEL by applying a conversion factor of 0.05 for NOAEL = 0.05 mg/kg-bw/d. A conversion factor of 5 was applied to get a chronic LOAEL of 0.25 mg/kg-bw/d. The laboratory TRVs were converted to wildlife TRVs of 0.078 and 0.39 mg/kg-bw/d for freshwater fish and 0.17 and 0.86 mg/kg-bw/d for marine species. See Table A.1.13.

A.1.14 *Selenium*

A laboratory chronic NOAEL and LOAEL of 0.4 and 0.8 mg/kg-bw/d, respectively, were reported by Sample et al. (1996) for mallard reproduction. The laboratory TRVs were extrapolated to avian wildlife TRVs of NOAEL = 0.27 and LOAEL= 0.53 mg/kg-bw/d. Sample et al. (1996) reported a selenium NOAEL of 0.2 mg/kg-bw/d and a LOAEL of 0.33 mg/kg-bw/d for reproduction in the rat. These values were converted to mammalian wildlife TRVs of 0.19 mg/kg-bw/d NOAEL and 0.32 mg/kg-bw/d LOAEL for freshwater mammals and 0.15 mg/kg-bw/d NOAEL and 0.24 mg/kg-bw/d LOAEL for marine mammals. A chronic NOAEL of 0.91 and LOAEL of 1.22 mg/kg-bw/d for the fathead minnow were reported (Ogle and Knight, 1989). The reported NOAEL and LOAEL were converted to a freshwater and marine fish NOAEL and LOAEL of 5.02 and 6.70 mg/kg-bw/d for freshwater fish and 11.04 and 14.75 mg/kg-bw/d for marine fish, respectively.

Table A.1.1 Summary of literature values for arsenic, inorganic

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-d
USFWS 1964	Avian	Mallard	1	Mortality	128 days	Chronic NOAEL	5.1			5.1		
USFWS 1964	Avian	Mallard	1	Mortality	128 days	Chronic LOAEL	12.8					12.8
Pedlar et al. 2002	Fish	Lake whitefish	0.326	Growth	64 days	Subchronic NOEC	0.2563[a]	mg/kg/d	0.1	0.02563	5	0.12815
USEPA 2005	Mammal	Dog	10.1	Biochemical	8 weeks	Chronic NOAEL	1.04			1.04		
USEPA 2005	Mammal	Dog	10.1	Biochemical	8 weeks	Chronic LOAEL	1.66					1.66
ATSDR 1993d	Mammal	Dog	10	Systemic	2 years	Chronic NOAEL	1.2	mg/kg/d		1.2	5	6
Sample et al. 1996	Mammal	Mouse	0.03	Reproduction	3 generations	Chronic NOAEL	0.126	mg/kg/d		0.126	5	0.63

a - NOEC was 119.6 ug/g for 3 day/week feeding at 0.5% BW/tank with 6 326g fish/tank. Converted to a daily dosage.

Table A.1.2 Summary of literature values for cadmium

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Mallard	1.153	Reproduction	90 days	Chronic NOAEL	1.45	mg/kg/d		1.45		
Sample et al. 1996	Avian	Mallard	1.153	Reproduction	90 days	Chronic LOAEL	20	mg/kg/d				20
Chowdhury et al. 2004	Fish	Rainbow Trout	0.1654	Survival	45 days	Subchronic NOEC	6.9	mg/kg/d	0.1	0.69	5	3.45
Szczerbik et al. 2006	Fish	Prussian Carp	0.00056	Growth	3 years	Chronic NOEC	20[a]	mg/kg/d		20		
Szczerbik et al. 2006	Fish	Prussian Carp	0.00056	Growth	3 years	Chronic LOEC	200[b]	mg/kg/d				200
Sample et al. 1996	Mammal	Rat	0.303	Reproduction	6 weeks w/gestation	Chronic NOAEL	1	mg/kg/d		1		
Sample et al. 1996	Mammal	Rat	0.303	Reproduction	6 weeks w/gestation	Chronic LOAEL	10	mg/kg/d				10
ATSDR 2008	Mammal	Dog	10	Reproduction	3 months	Chronic NOAEL	0.75	mg/kg/d		0.75	5	3.75

a - Carp at 2% BW/d, or 0.0112g. NOEC conc. Was 1 mg cd/g food, so daily dose was 20 mg/kg/d.
b - Carp at 2% BW/d, or 0.0112g. LOEC conc. Was 10 mg cd/g food, so daily dose was 200 mg/kg/d.

Table A.1.3 Summary of literature values for chlordane, total

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Red-winged blackbird	0.06	Survival	84 days	Chronic NOAEL	2.14	mg/kg/d		2.14		
Sample et al. 1996	Avian	Red-winged blackbird	0.06	Survival	84 days	Chronic LOAEL	10.7	mg/kg/d				10.7
Wiemeyer 1996	Avian	Northern bobwhite	0.19	Reproduction	Not Specified	Chronic NOAEL	1.19	mg/kg/d		1.19		
Wiemeyer 1996	Avian	Northern bobwhite	0.19	Reproduction	Not Specified	Chronic LOAEL	5.95	mg/kg/d				5.95
Wiemeyer 1996	Avian	Mallard	1	Reproduction	Not Specified	Chronic NOAEL	0.8	mg/kg/d		0.8		
Wiemeyer 1996	Avian	Mallard	1	Reproduction	Not Specified	Chronic LOAEL	4	mg/kg/d				4
Hudson et al. 1984	Avian	Mallard	1 [a]	Survival	14 days	LD_{50}	1200	mg/kg	0.01	12	5	60
Hudson et al. 1984	Avian	California quail	1 [b]	Survival	14 days	LD_{50}	14.1	mg/kg	0.01	1.41	5	7.05
Hudson et al. 1984	Avian	Pheasant	1 [b]	Survival	14 days	LD_{50}	24	mg/kg	0.01	0.24	5	1.2
Sample et al. 1996	Mammal	Mouse	0.03	Reproduction	6 generations	Chronic NOAEL	4.58	mg/kg/d		4.58		
Sample et al. 1996	Mammal	Mouse	0.03	Reproduction	6 generations	Chronic LOAEL	9.16	mg/kg/d				9.16
USEPA 1976	Mammal	Rat	0.35 [a]	Survival	Not Specified	LD_{50}	335	mg/kg	0.01	3.35	5	16.75

a - Reference BW from Sample et al., 1996.
b - Extrapolated from LD50 because the unit is mg chemical/kg body weight.

Table A.1.4 Summary of literature values for DDT, total

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
USEPA 1995	Avian	Japanese quail	0.11	Reproduction	3 generations	Chronic NOAEL	0.5	mg/kg/d		0.5		
USEPA 1995	Avian	Japanese quail	0.11	Reproduction	3 generations	Chronic LOAEL	5	mg/kg/d				5
USEPA 1995	Avian	Mallard	1	Reproduction	2 years	Chronic NOAEL	0.6	mg/kg/d		0.6		
USEPA 1995	Avian	Mallard	1	Reproduction	2 years	Chronic LOAEL	1.5	mg/kg/d				1.5
USEPA 1995	Avian	Bald Eagle	4.6	Reproduction	112 days	Chronic NOAEL	0.3	mg/kg/d		0.3		
USEPA 1995	Avian	Bald Eagle	4.6	Reproduction	112 days	Chronic LOAEL	3	mg/kg/d				3
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	>2240	mg/kg	0.01	>22.4	5	>112
Hudson et al. 1984	Avian	California quail	1 [a]	Survival	14 days	LD_{50}	595	mg/kg	0.01	5.95	5	29.75
Hudson et al. 1984	Avian	Japanese quail	0.15 [b]	Survival	14 days	LD_{50}	841	mg/kg	0.01	8.41	5	42.05
Hudson et al. 1984	Avian	Pheasant	1 [a]	Survival	14 days	LD_{50}	1334	mg/kg	0.01	13.34	5	66.7
Hudson et al. 1984	Avian	Sandhill crane	1 [a]	Survival	14 days	LD_{50}	>1200	mg/kg	0.01	>12	5	>60
Hudson et al. 1984	Avian	Rock dove	1 [a]	Survival	14 days	LD_{50}	>4000	mg/kg	0.01	>40	5	>200
Hudson et al. 1984	Avian	Mallard	1 [b]	Survival	30 days	EMLD (empirical minimum lethal dosage)	50	md/kg/d				
Macek et al. 1970	Fish	Rainbow trout	0.0147	Growth	140 days	Chronic NOEC	1	mg/kg/w		0.143	5	0.715
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	2 years	Chronic NOAEL	0.8	mg/kg/d		0.8		
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	2 years	Chronic LOAEL	4	mg/kg/d				4

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
ATSDR 2002c	Mammal	Dog	10	Reproduction	2 generations	Chronic NOAEL	1	mg/kg/d		1	5	5
USEPA 1976	Mammal	Rat	0.35 [b]	Survival	Not Specified	LD_{50}	113	mg/kg	0.01	1.13	5	5.65
EXTOXNET 1996	Mammal	Rat	0.35 [b]	Reproduction	15-19 days	Chronic NOAEL	38	mg/kg/d		38	5	190
EXTOXNET 1996	Mammal	Monkey/ Hamster	1 [a]	Unspecified	3.5-7 years	Chronic NOAEL	8 to 20	mg/kg/d		8 to 20	5	40-100
Macek 1968	Fish	Brook trout	0.162	Growth	156 days	Chronic NOEC	2	mg/kg/w		0.286	5	1.43

a - Extrapolated from LD50 because the unit is mg chemical/kg body weight.
b - Reference BW from Sample et al., 1996.

Table A.1.5 Summary of literature values for dieldrin

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Barn Owl	0.47	Reproduction	2 years	Chronic NOAEL	0.08	mg/kg/d		0.08		
Sample et al. 1996	Avian	Barn Owl	0.47	Reproduction	2 years	Chronic LOAEL	0.39	mg/kg/d				0.39
Hudson et al. 1984	Avian	Canada Goose	1[a]	Survival	14 days	LD_{50}	<141	mg/kg	0.01	<1.41	5	<7.05
Hudson et al. 1984	Avian	Mallard	1[b]	Survival	14 days	LD_{50}	381	mg/kg	0.01	3.81	5	19.05
Hudson et al. 1984	Avian	California quail	1 [a]	Survival	14 days	LD_{50}	8.78	mg/kg	0.01	0.0878	5	0.439
Hudson et al. 1984	Avian	Japanese quail	0.15[b]	Survival	14 days	LD_{50}	69.7	mg/kg	0.01	0.697	5	3.485
Hudson et al. 1984	Avian	Pheasant	1 [a]	Survival	14 days	LD_{50}	79	mg/kg	0.01	0.79	5	3.95
Hudson et al. 1984	Avian	Chukar	1 [a]	Survival	14 days	LD_{50}	25.3	mg/kg	0.01	0.253	5	1.265
Hudson et al. 1984	Avian	Rock dove	1 [a]	Survival	14 days	LD_{50}	26.6	mg/kg	0.01	0.266	5	1.33
Hudson et al. 1984	Avian	House sparrow	1 [a]	Survival	14 days	LD_{50}	47.6	mg/kg	0.01	0.476	5	2.38
Hudson et al. 1984	Avian	Fulvous whistling duck	1 [a]	Survival	14 days	LD_{50}	100	mg/kg	0.01	1	5	5
Hudson et al. 1984	Avian	Mallard	1 [b]	Survival	30 days	EMLD (empirical minimum lethal dosage)	5	mg/kg/d				
Hudson et al. 1984	Avian	Fulvous whistling duck	1 [a]	Survival	30 days	EMLD	2.5	mg/kg/d				
Hudson et al. 1984	Avian	Gray partridge	1 [a]	Survival	30 days	EMLD	1.25	mg/kg/d				
Argyle et al. 1975	Fish	Channel Catfish	0.003	Growth	210 days	Chronic NOEC	0.024[c]	mg/kg/d		0.024		
Macek et al.	Fish	Rainbow	0.0147	Growth	140 days	Chronic	1	mg/kg/w		0.143	5	0.715

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
1970		trout				NOEC		eek				
Argyle et al. 1975	Fish	Channel Catfish	0.003	Growth	210 days	Chronic LOEC	0.12[d]	mg/kg/d				0.12
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic NOAEL	0.04	mg/kg/d		0.04		
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic LOAEL	0.2	mg/kg/d				0.2
ASTDR 2002b	Mammal	Dog	10	Systemic	15.7 months	Chronic LOAEL	0.14	mg/kg/d	0.2	0.028		0.14
Hudson et al. 1984	Mammal	Mule deer	1 [a]	Survival	14 days	LD_{50}	75	mg/kg	0.01	0.75	5	3.75
Hudson et al. 1984	Mammal	Domestic goat	1 [a]	Survival	14 days	LD_{50}	100	mg/kg	0.01	1	5	5
USEPA 1976	Mammal	Rat	0.35	Survival	Not Specified	LD_{50}	46	mg/kg	0.01	0.46	5	2.3

a - Extrapolated from LD50 because the unit is mg chemical/kg body weight.
b - Reference BW from Sample et al., 1996.
c - Treatment of 0.8 ug/g food, 3g BW, 4.2%BW feeding rate, 5 days a week was converted to mg/kg/d.
d - Treatment of 4 ug/g food, 3g BW, 4.2%BW feeding rate, 5 days a week was converted to mg/kg/d.

Table A.1.6 Summary of literature values for endrin, total

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Mallard	1.15	Reproduction	>200 days	Chronic NOAEL	0.3	mg/kg/d		0.3		
Sample et al. 1996	Avian	Mallard	1.15	Reproduction	>200 days	Chronic LOAEL	1.5	mg/kg/d				1.5
Sample et al. 1996	Avian	Screech Owl	0.18	Reproduction	>83 days	Chronic NOAEL	0.02	mg/kg/d		0.02		
Sample et al. 1996	Avian	Screech Owl	0.18	Reproduction	>83 days	Chronic LOAEL	0.1	mg/kg/d				0.1
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	5.64	mg/kg	0.01	0.0564	5	0.282
Hudson et al. 1984	Avian	Sharp-tailed grouse	1 [b]	Survival	14 days	LD_{50}	1.06	mg/kg	0.01	0.0106	5	0.053
Hudson et al. 1984	Avian	California quail	1 [b]	Survival	14 days	LD_{50}	1.19	mg/kg	0.01	0.0119	5	0.0595
Hudson et al. 1984	Avian	Pheasant	1 [b]	Survival	14 days	LD_{50}	1.78	mg/kg	0.01	0.0178	5	0.089
Hudson et al. 1984	Avian	Rock dove	1 [b]	Survival	14 days	LD_{50}	2	mg/kg	0.01	0.02	5	0.1
Hudson et al. 1984	Avian	Mallard	1 [a]	Survival	30 days	EMLD (empirical minimum lethal dosage)	0.25	mg/kg/d				
IPCS 1992	Avian	Pigeon	1 [b]	Survival	Not specified	LD_{50}	2	mg/kg	0.01	0.02	5	0.1
IPCS 1992	Avian	Redwinged blackbird	1 [b]	Survival	Not specified	LD_{50}	2.37	mg/kg	0.01	0.0237	5	0.1185
IPCS 1992	Avian	Quail	1 [b]	Survival	Not specified	LD_{50}	4.22	mg/kg	0.01	0.0422	5	0.211
Grant and Mehrle 1970	Fish	Goldfish	0.0152	Growth	157 days	Chronic NOEC	0.143	mg/kg/d		0.143		
Grant and Mehrle 1970	Fish	Goldfish	0.0137	Growth	157 days	Chronic LOEC	0.43	mg/kg/d				0.43
Grant and Mehrle 1973	Fish	Rainbow trout	0.129	Growth	163 days	Chronic NOEC	0.043	mg/kg/d		0.043		

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Grant and Mehrle 1973	Fish	Rainbow trout	0.134	Growth	163 days	Chronic LOEC	0.145	mg/kg/d				0.145
Argyle et al. 1973	Fish	Channel catfish	0.0005	Growth	198 days	Chronic NOEC	0.04	mg/kg/d		0.04	5	0.2
Sample et al. 1996	Mammal	Mouse	0.03	Reproduction	120 Days	Chronic NOAEL	0.18	mg/kg/d		0.18		
Sample et al. 1996	Mammal	Mouse	0.03	Reproduction	120 Days	Chronic LOAEL	0.92	mg/kg/d				0.92
Hudson et al. 1984	Mammal	Mule deer	1 [b]	Survival	14 days	LD_{50}	6.25	mg/kg	0.01	0.0625	5	0.3125
Hudson et al. 1984	Mammal	Domestic goat	1 [b]	Survival	14 days	LD_{50}	25	mg/kg	0.01	0.25	5	1.25
USEPA 1976	Mammal	Rat	0.35 [a]	Survival	Not specified	LD_{50}	8	mg/kg	0.01	0.08	5	0.4
IPCS 1992	Mammal	Big brown bat	1 [b]	Survival	Not specified	LD_{50}	5	mg/kg	0.01	0.05	5	0.25

a - Reference BW from Sample et al., 1996.
b - Extrapolated from LD50 because the unit is mg chemical/kg body weight.

Table A.1.7 Summary of literature values for endosulfan, total

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Gray Partridge	0.4	Reproduction	28 days	Chronic NOAEL	10	mg/kg/d		10		
Sample et al. 1996	Avian	Gray Partridge	0.4	Reproduction	28 days	Chronic LOAEL	50	mg/kg/d				50
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	33	mg/kg	0.01	0.33	5	1.65
Hudson et al. 1984	Avian	Mallard	1 [a]	Survival	14 days	LD_{50}	45	mg/kg	0.01	0.45	5	2.25
Hudson et al. 1984	Avian	Mallard	1 [a]	Survival	14 days	LD_{50}	31.2	mg/kg	0.01	0.312	5	1.56
Hudson et al. 1984	Avian	Pheasant	1[b]	Survival	14 days	LD_{50}	80	mg/kg	0.01	0.8	5	4
Hudson et al. 1984	Avian	Pheasant	1 [b]	Survival	14 days	LD_{50}	190	mg/kg	0.01	1.9	5	9.5
Hudson et al. 1984	Avian	Pheasant	1 [b]	Survival	14 days	LD_{50}	>320	mg/kg	0.01	>3.2	5	>16
Lundebye et al. 2010	Fish	Atlantic Salmon	0.25	Lipid digestibility	95 days	Chronic NOEC	0.0002393	mg/kg/d		0.0002393		
Lundebye et al. 2010	Fish	Atlantic Salmon	0.25	Lipid digestibility	95 days	Chronic LOEC	0.0005286	mg/kg/d				0.0005286
Petri et al. 2006	Fish	Atlantic Salmon	0.0387	Condition Factor	49 days	Subchronic NOEC	0.000758	mg/kg/d	0.1	0.0000758	5	0.000379
Petri et al. 2006	Fish	Atlantic Salmon	0.0387	Condition Factor	49 days	Subchronic LOEC	0.010621	mg/kg/d	0.05	0.00053105	5	0.00265525
Berntssen et al. 2008	Fish	Atlantic Salmon	0.148	Growth	92 days	Chronic NOEC	0.005792	mg/kg/d		0.005792	5	0.02896
Coimbra et al. 2007	Fish	Nile Tilapia	0.09105	Liver Pathology	35 days	Subchronic LOEC	0.0000197	mg/kg/d	0.05	0.000000985	5	0.000004925
Sample et al. 1996	Mammal	Rat	0.35	Fertility	30 days	Chronic NOAEL	1.5	mg/kg/d		1.5		
Sample et al. 1996	Mammal	Rat	0.35	Fertility	30 days	Chronic LOAEL	7.5	mg/kg/d				7.5
ASTDR 2000	Mammal	Dog	10	Systemic	2 years	Chronic NOAEL	1	mg/kg/d		1		

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
ASTDR 2000	Mammal	Dog	10	Systemic	2 years	Chronic LOAEL	5	mg/kg/d				5
EXTOXNET 1996	Mammal	Rat	0.35 [a]	Reproduction	Three generations	Chronic NOAEL	2.5	mg/kg/d		2.5	5	12.5

a - Reference BW from Sample et al., 1996.
b - Extrapolated from LD50 because the unit is mg chemical/kg body weight.

Table A.1.8 Summary of literature values for hexachlorobenzene

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Coulston and Kolbye 1994; Terretox 2002	Avian	Japanese Quail	0.15	Reproduction	90 days	Chronic NOAEL	0.11	mg/kg/d		0.11		
Coulston and Kolbye 1994; Terretox 2002	Avian	Japanese Quail	0.15	Reproduction	90 days	Chronic LOAEL	0.57	mg/kg/d				0.57
EXTOXNET 1996	Avian	Bobwhite	0.19[a]	Survival	Not Specified	LD_{50}	575	mg/kg	0.01	5.75	5	28.75
EXTOXNET 1996	Avian	Mallard	1[b]	Survival	Not Specified	LD_{50}	1450	mg/kg	0.01	14.5	5	
Niimi and Cho 1980	Fish	Rainbow Trout	0.09	Growth	57 days	Subchronic NOEL	0.0234[d]	mg/kg/d		0.0234	5	0.117
Woodburn et al. 2008	Fish	Channel catfish	0.004	Growth	28 days	Subchronic NOEL	0.00685[e]	mg/kg/d	0.1	0.000685	5	0.003425
ATSDR 2002a	Mammal	Rat	0.35	Reproduction	4 generations	Chronic NOAEL	1	mg/kg/d		1		
ATSDR 2002a	Mammal	Rat	0.35	Reproduction	4 generations	Chronic LOAEL	2	mg/kg/d				2
ATSDR 2002a	Mammal	Dog	10	Systemic	1 Year	Chronic NOAEL	1.2	mg/kg/d		1.2		
ATSDR 2002a	Mammal	Dog	10	Systemic	1 Year	Chronic LOAEL	12	mg/kg/d				12
EXTOXNET 1996	Mammal	Rat	0.35[b]	Survival	Not Specified	LD_{50}	3500	mg/kg	0.01	35	5	175
EXTOXNET 1996	Mammal	Mouse	0.03[b]	Survival	Not Specified	LD_{50}	4000	mg/kg	0.01	40	5	200
EXTOXNET 1996	Mammal	Rabbit	1[c]	Survival	Not Specified	LD_{50}	2600	mg/kg	0.01	26	5	130
EXTOXNET 1996	Mammal	Cat	1[c]	Survival	Not Specified	LD_{50}	1700	mg/kg	0.01	17	5	85
Arnold et al. 1985	Mammal	Rat	0.35[b]	Liver Effects	130 weeks	Chronic NOAEL	0.08	mg/kg/d		0.08		
Arnold et al. 1985	Mammal	Rat	0.35[b]	Liver Effects	130 weeks	Chronic LOAEL	0.29	mg/kg/d				0.29

a - Reference BW from Wildlife Exposure Factors Handbook.

b - Reference BW from Sample et al., 1996.
c - Extrapolated from LD50 because the unit is mg chemical/kg body weight.
d - 90g trout consumed 3% BW/d, which is .0027 kg food. Highest conc. Was 780 ug HCB/kg food. 0.002106 mg HCB/ .09 kg BW/ d = 0.0234 mg HCB/kg BW/d
e - 4g fish were fed 327 ng HCB/g food and ate 2.1% BW/d = 0.084g food/d. 0.084g food x 0.000327mg HCB/g food = 0.0000274 mg/HCB/4g BW/d. 0.0000274 mg HCB/0.004kg BW = 0.00685 mg/kg/d.

Table A.1.9 Summary of literature values for lindane

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Mallard	1	Reproduction	8 weeks	Chronic NOAEL	4	mg/kg/d		4		
Sample et al. 1996	Avian	Mallard	1	Reproduction	8 weeks	Chronic LOAEL	20	mg/kg/d				20
Sample et al. 1996	Avian	Japanese quail	0.15	Reproduction	90 days	Chronic NOAEL	0.56	mg/kg/d		0.56		
Sample et al. 1996	Avian	Japanese quail	0.15	Reproduction	90 days	Chronic LOAEL	2.25	mg/kg/d				2.25
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	>2000	mg/kg	0.01	>20		
Hudson et al. 1984	Avian	Mallard	1 [a]	Survival	30 days	EMLD (empirical minimum lethal dosage)	30	mg/kg/d				
Cossarini-Dunier et al 1987	Fish	Carp	0.06	Immune Response	109 days	Chronic NOEC	10[c]	mg/kg/d		10	5	50
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic NOAEL	8	mg/kg/d		8		
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic LOAEL	40	mg/kg/d				40
EXTOXNET 1996	Mammal	Rat	0.35 [a]	Survival	Not Specified	LD_{50}	88	mg/kg	0.01	0.88	5	4.4
EXTOXNET 1996	Mammal	Mouse	0.03 [a]	Survival	Not Specified	LD_{50}	59	mg/kg	0.01	0.59	5	2.95
EXTOXNET 1996	Mammal	Guinea pig	1[b]	Survival	Not Specified	LD_{50}	100	mg/kg	0.01	1	5	5
EXTOXNET 1996	Mammal	Rabbit	1 [b]	Survival	Not Specified	LD_{50}	200	mg/kg	0.01	2	5	10
EXTOXNET 1996	Mammal	Mice, Rats, Dogs	1 [b]	Chronic	2 years	Chronic NOAEL	1.25	mg/kg/d		1.25	5	6.25
EXTOXNET 1996	Mammal	Rat	0.35 [a]	Reproduction	138 days	Chronic NOAEL	5	mg/kg/d		5	5	25

a - Reference BW from Sample et al., 1996.
b - Extrapolated from LD50 because the unit is mg chemical/kg body weight.
c - 60g carp fed 1% BW/d = 0.6g food/d. 1000mg lindane/kg food = 0.6mg lindane/0.06kg BW/d = 10 mg/kg/d.

Table A.1.10 Summary of literature values for mercury (methylmercury)

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Heinz and Locke 1976	Avian	Mallard	1	Reproduction		Chronic NOAEL	0.03			0.03		
Heinz and Locke 1976	Avian	Mallard	1		1.5 years	Chronic LOAEL	0.18					0.18
Sample et al. 1996	Avian	Japanese quail	0.15	Reproduction	1 year	Chronic NOAEL	0.45	mg/kg/d		0.45		
Sample et al. 1996	Avian	Japanese quail	0.15	Reproduction	1 year	Chronic LOAEL	0.9	mg/kg/d				0.9
USEPA 1995	Avian	Red-tailed Hawk	1.1	Survival/ Neurological	12 weeks	Chronic NOAEL	0.49	mg/kg/d		0.49		
USEPA 1995	Avian	Red-tailed Hawk	1.1	Survival/ Neurological	12 weeks	Chronic LOAEL	1.2	mg/kg/d				1.2
USEPA 1997	Avian	Mallard	1	Reproduction	3 generations	Chronic NOAEL	0.026	mg/kg/d		0.026		
USEPA 1997	Avian	Mallard	1	Reproduction	3 generations	Chronic LOAEL	0.078	mg/kg/d				0.078
Lee et al. 2011	Fish	Green Sturgeon	0.028	Survival and Growth	8 weeks	Subchronic NOEC	0.625[a]	mg/kg/d	0.1	0.0625		
Lee et al. 2011	Fish	Green Sturgeon	0.028	Survival and Growth	8 weeks	Subchronic LOEC	1.25[b]	mg/kg/d	0.05	0.0625	5	0.3125
Lee et al. 2011	Fish	White Sturgeon	0.028	Survival and Growth	8 weeks	Subchronic NOEC	1.25 [b]	mg/kg/d	0.1	0.125		
Lee et al. 2011	Fish	White Sturgeon	0.028	Survival and Growth	8 weeks	Subchronic LOEC	2.5 [c]	mg/kg/d	0.05	0.125	5	0.625
Berntssen et al. 2003	Fish	Atlantic Salmon	0.0105	Brain Pathology	4 months	Chronic NOEC	0.13776 [d]	mg/kg/d		0.13776		
Berntssen et al. 2003	Fish	Atlantic Salmon	0.0105	Brain Pathology	4 months	Chronic LOEC	1.59456 [e]	mg/kg/d				1.59456
Berntssen et al. 2003	Fish	Atlantic Salmon	0.0108	Brain Pathology	4 months	Chronic NOEC	0.06768 [f]	mg/kg/d		0.06768		
Berntssen et al. 2003	Fish	Atlantic Salmon	0.0108	Brain Pathology	4 months	Chronic LOEC	0.13296 [g]	mg/kg/d				0.13296
Fuyuta et al. 1978	Mammal	Rat	0.428	Development		Chronic LOAEL	4		0.2	0.8		4

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Khera and Tabacova 1973	Mammal	R	0.1875	Reproduction	122 days	Chronic NOAEL	0.25			0.25	5	1.25
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic NOAEL	0.032	mg/kg/d		0.032		
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	3 generations	Chronic LOAEL	0.16	mg/kg/d				0.16
Sample et al. 1996	Mammal	Mink	1	Survival/ Weight loss	93 days	Chronic NOAEL	0.25	mg/kg/d		0.25		
Sample et al. 1996	Mammal	Mink	1	Survival/ Weight loss	93 days	Chronic LOAEL	0.15	mg/kg/d				0.15

a - Avg. daily ration of 2.5%. Treatment was 25 mg/kg food, converted to a daily dose.
b - Avg. daily ration of 2.5%. Treatment was 50 mg/kg food, converted to a daily dose.
c - Avg. daily ration of 2.5%. Treatment was 100 mg/kg food, converted to a daily dose.
d - Ration was 1.6% BW/d, treatment was 8.61 mg/kg feed converted to a daily dose.
e - Ration was 1.6% BW/d, treatment was 99.66 mg/kg feed converted to a daily dose.
f - Ration was 1.6% BW/d, treatment was 4.23 mg/kg feed converted to a daily dose.
g - Ration was 1.6% BW/d, treatment was 8.31 mg/kg feed converted to a daily dose.

Table A.1.11 Summary of literature values for mirex

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	>2400	mg/kg	0.01	>24	5	>120
Hudson et al. 1984	Avian	Pheasant	1[b]	Survival	14 days	LD_{50}	>2000	mg/kg	0.01	>20	5	>100
Hyde et al. 1973	Avian	Mallard	1	Reproduction	25 weeks	Chronic NOEC	0.01[c]	mg/kg/d		0.01		
Hyde et al. 1973	Avian	Mallard	1	Reproduction	25 weeks	Chronic LOEC	1[d]	mg/kg/d				1
USEPA 1986	Fish	Bluegill	1[b]	Growth	168 days	Chronic NOAEL	3	mg/kg		3	5	15
Van Valin et al. 1968	Fish	Bluegill	0.0132	Growth	168 days	Chronic NOEL	2.14[e]	mg/kg/d		2.14		
Van Valin et al. 1968	Fish	Bluegill	0.0136	Growth	168 days	Chronic LOEL	3.57[f]	mg/kg/d				3.57
Skea et al. 1981	Fish	Brook Trout	0.1145	Growth	104 days	Chronic NOEC	0.3[g]	mg/kg/d		0.3	5	1.5
Leatherland & Sonstegard 1980	Fish	Rainbow trout	0.05	Liver and Thyroid Effects	1 month	Subchronic NOEC	1[h]	mg/kg/d	0.1	0.1	5	0.5
WHO 1984	Mammal	Rat	0.35[a]	Survival	Not Specified	LD_{50}	600	mg/kg	0.01	6	5	30
WHO 1984	Mammal	Rat	0.35[a]	Survival	Not Specified	LD_{50}	365	mg/kg	0.01	3.65	5	18.25
WHO 1984	Mammal	Hamster	1[b]	Survival	Not Specified	LD_{50}	125	mg/kg	0.01	1.25	5	6.25
WHO 1984	Mammal	Dog	1[b]	Survival	Not Specified	LD_{50}	1000	mg/kg	0.01	10	5	50
USEPA 1986	Mammal	Rat	0.35[a]	Chronic	90 days	Chronic LOAEL	6.2	mg/kg/d	0.2	1.24		
NTP 1990	Mammal	Rat	0.12	Liver and Thyroid Effects	104 weeks	Chronic NOAEL	0.07	mg/kg/d		0.07		
NTP 1990	Mammal	Rat	0.12	Liver and Thyroid Effects	104 weeks	Chronic LOAEL	0.7	mg/kg/d				0.7

a - Reference BW from Sample et al., 1996.
b - Extrapolated from LD50 because the unit is mg chemical/kg body weight.
c - Value reported was 1 ppm. Used reference values of 1 kg BW and 100 g/d food intake from Sample et al to convert to mg/kg/d.
d - Value reported was 100 ppm. Used reference values of 1 kg BW and 100 g/d food intake from Sample et al to convert to mg/kg/d.
e - Treatment of 3 mg/kg 5 days a week converted to a daily dosage of 2.14 mg/kg.
f - Treatment of 5 mg/kg five days a week converted to a daily dosage of 3.57 mg/kg.
g - Treatment of .7 mg/kg three times a week converted to daily dosage of .3 mg/kg.
h - 50g trout at 2% BW/d at 50mg/kg food = 0.001kg food/d x 50mg Mirex/kg food = 0.05mg/0.05kg/d = 1 mg/kg/d.

Table A.1.12 Summary of literature values for polychlorinated biphenyls (PCBs)

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-d
Hudson et al. 1984	Avian	Mallard	1[a]	Survival	14 days	LD_{50}	>2000	mg/kg	0.01	>20	5	100
Hudson et al. 1984	Avian	Bobwhite	0.19 [b]	Survival	14 days	LD_{50}	>2000	mg/kg	0.01	>20	5	100
Biessmann 1982	Avian	Japanese quail	0.072 [a]	Reproduction	3 weeks	Chronic LOEC	7.29[d]	mg/kg/d	0.2	1.458		7.29
Sample et al. 1996	Avian	Ring-necked pheasant	1	Reproduction	17 weeks	Chronic NOAEL	0.18	mg/kg/d		0.18		
Sample et al. 1996	Avian	Ring-necked Pheasant	1	Reproduction	17 weeks	Chronic LOAEL	1.8	mg/kg/d				1.8
Leatherland and Sonstegard 1980	Fish	Rainbow Trout	0.05	Liver and Thyroid Effects	1 month	Subchronic LOEC	1[e]	mg/kg/d	0.05	0.05	5	0.25
Nakayama 2004	Fish	Japanese medaka	0.0003 [c]	Reproduction	3 weeks	Subchronic NOEC	1	mg/kg/d	0.1	0.1	5	0.5
Hudson et al. 1984	Mammal	Albino rat	0.35 [a]	Survival	14 days	LD_{50}	841	mg/kg	0.01	8.41	5	42.05
Hudson et al. 1984	Mammal	Albino rat	0.35 [a]	Survival	14 days	LD_{50}	2000	mg/kg	0.01	20	5	100
Sample et al. 1996	Mammal	Oldfield mouse	0.014	Reproduction	12 months	Chronic NOAEL	0.068	mg/kg/d		0.068		
Sample et al. 1996	Mammal	Oldfield mouse	0.014	Reproduction	12 months	Chronic LOAEL	0.68	mg/kg/d				0.68
Sample et al. 1996	Mammal	Mink	1	Reproduction	4.5 months	Chronic NOAEL	0.14	mg/kg/d		0.14		
Sample et al. 1996	Mammal	Mink	1	Reproduction	4.5 months	Chronic LOAEL	0.69	mg/kg/d				0.69

a - Reference BW from Sample et al., 1996.
b - Reference BW from Wildlife Exposure Factors Handbook.
c - This was the BW before a 1 month acclimation prior to test initiation (Leatherland and Sonstegard, 1980).
d - Delayed egg laying was observed at 50 ppm, ingestion rate was 0.1458 kg food/kg bw/d according to Nagy equation, so daily dose was 7.29 mg/kg/d
e - 50g trout at 2% BW/d at 50mg/kg food = 0.001kg food/d x 50mg PCB/kg food = 0.05mg/0.05kg/d = 1 mg/kg/d.

Table A.1.13 Summary of literature values for selenium

Source (Author, Year)	ROC	Effects Endpoint										
		Study Species	Study Species Body Weight (kg)	Study Endpoint	Study Duration	Study Endpoint Type	Reported Endpoint	Units	UF	Chronic NOAEL/ NOEC mg/kg-day	UF	Chronic LOAEL/ LOEC mg/kg-day
Sample et al. 1996	Avian	Mallard	1	Reproduction	100 days	Chronic LOAEL	0.8	mg/kg/d				0.8
Sample et al. 1996	Avian	Mallard	1	Reproduction	100 days	Chronic NOAEL	0.4	mg/kg/d		0.4		
Sample et al. 1996	Avian	Black-crowned night-heron	0.88	Reproduction	94 days	Chronic NOAEL	1.8	mg/kg/d		1.8		
Sample et al. 1996	Avian	Black-crowned night-heron	0.88	Reproduction	94 days	Chronic LOAEL	9	mg/kg/d				9
Sample et al. 1996	Avian	Screech owl	0.2	Reproduction	13.7 weeks	Chronic NOAEL	0.44	mg/kg/d		0.44		
Sample et al. 1996	Avian	Screech owl	0.2	Reproduction	13.7 weeks	Chronic LOAEL	1.45	mg/kg/d				1.45
Wang et al. 2007	Fish	Crucian carp	0.01367	Survival	30 days	Subchronic NOEC	0.0165[a]	mg/kg/d	0.1	0.00165	5	0.00825
Ogle and Knight 1989	Fish	Fathead minnow	0.00009	Growth	98 days	Chronic NOEC	0.912	mg/kg/d		0.912		
Ogle and Knight 1989	Fish	Fathead minnow	0.00009	Growth	98 days	Chronic LOEC	1.218	mg/kg/d				1.218
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	1 year	Chronic NOAEL	0.2	mg/kg/d		0.2		
Sample et al. 1996	Mammal	Rat	0.35	Reproduction	1 year	Chronic LOAEL	0.33	mg/kg/d				0.33

a - Fed 3% BW/d of 0.55 mg selenium/kg diet, converted to 0.0165 mg/kg/d

A.2 UNCERTAINTIES/LIMITATIONS

A.2.1 *Body Weight*

The use of minimum adult body weights may overestimate the risk to the receptor population that are typically heavier than the minimum reported weight. The use of the minimum body weight may also under-estimate the risk to juveniles within each population. The use of the minimum body weight is a typical conservative assumption in risk estimate (USEPA, 1997).

A.2.2 *High Food Ingestion Rate*

The formulae presented in Nagy (1987) calculate food ingestion rate based on body weight. Because the food ingestion (birds and mammals) and daily ration (fish) are based on metabolism of the receptor, the smaller individuals generally consume more food than larger receptors based on body weight. This uncertainty may over- or under-estimate the calculated fish tissue concentration depending on whether a receptors food ingestion rate is higher or lower than what is calculated.

A.2.3 *Ingestion TRVs*

Data on the toxicity of many of the contaminants to wildlife receptors were sparse or lacking, requiring the extrapolation of data from laboratory studies with non-wildlife species. This is a typical extrapolation for ecological risk assessments because, so few wildlife species have been tested directly for most constituents. The uncertainties associated with toxicity extrapolation were minimized through the selection of the most appropriate test species for which suitable toxicity data were available. The factors considered in selecting a test species to represent a receptor group included taxonomic relatedness, trophic level, and available dietary toxicity data.

A.2.4 *Contaminant Exposure*

The screening fish tissue concentration calculated accounts for the risk to upper trophic level receptors from each contaminant due to the uptake through the diet only. Receptors are not only exposed to contaminants through diet but may be exposed through incidental uptake of inorganic media (i.e., surface water, sediment, or soil), dermal contact, and via respiration. These additional exposure pathways are typically evaluated in ecological risk assessments but were not in the calculation of the screening fish tissue concentrations. Therefore, the risk to upper trophic level receptors based on the fish tissue screening value may under-estimated the overall risk to each receptor group from the contaminants of concern.

A.2.5 *Constituent Mixtures*

Information on the ecotoxicological effects of constituent interactions is generally lacking, although it is required (as is standard for ERAs) that the constituents be evaluated on a constituent-by-constituent basis in comparison to TRVs. This could result in an underestimation of risk (if there are additive or synergistic effects among constituents) or an overestimation of risks (if there are antagonistic effects among constituents).

A.2.6 *Chlordane Dietary Exposure to Fish*

Toxicity data for dietary exposure of chlordane to fish was not available in the scientific literature and represents an uncertainty.

A.3 GENERALIZED RECEPTOR OF CONCERN GROUPINGS

Generalized receptor of concern ROC groupings used as endpoints for the NCCA ecological fish tissue contaminant index. The most sensitive receptor in each group was determined by the highest food ingestion rate per body weight (highlighted in yellow in the following table). American mink and muskellunge were selected, respectively as the generalized mammalian and fish ROCs.

Table A.3.1 Minimum and maximum body weights and derived food ingestion rates for select receptors of concern commonly used in ecological risk assessments.

Group	Receptors	Body Weight (kg)		Ref.	Food Ingestion Rate (kg food/kg BW/d)	
		Min/Ave	Max		Min/Ave BW	Max BW
Avian[1]	Great Blue Heron	1.47	2.99	a	0.051	0.040
	Western Osprey	1.22	1.95		0.054	0.046
	Bald Eagle	3.00	4.50		0.040	0.034
	Herring Gull	0.83	1.62		0.062	0.049
	Belted Kingfisher	0.13	0.22		0.120	0.100
	Brown Pelican	3.00	3.50	b	0.040	0.038
Freshwater Mammals[1]	River Otter	5.00	15.00	a	0.052	0.042
	American Mink	0.55	2.08		0.076	0.060
Marine Mammals[1]	Harbor Seal	58.80	124.00		0.033	0.029
	Bottlenose Dolphin	150.00	490.00	c	0.028	0.023
	Atlantic Walrus	900.00	1400.00	d	0.020	0.019
Marine Fish[2]	Bluefin Tuna	32.00	219.00	e	0.044	0.016
	Yellowfin Tuna	23.42	52.45	f	0.023	0.010
	Shortfin Mako	63.50		g	0.046	
	Sandbar Shark	34.00		h	0.009	
	Mackerel Tuna	34.55		i	0.022	
	Swordfish	58.00		j	0.016	
Freshwater Fish[2]	Brown Trout	0.91	3.63	k	0.0095	
	Muskellunge	0.34	31.64	l	0.064	
	Largemouth Bass	0.45	4.50	m	0.024	

[1] Avian and mammalian food ingestion rates were calculated using equations derived from Nagy (1987).

[2] Food ingestion rates for fish were calculated based on daily rations. Daily rations were converted from percent body weight/day to kg food/ kg body weight/day in order to estimate food ingestion rates that are comparable to the avian and mammalian values. Data for the shortfin mako, sandbar shark, mackerel tuna, and swordfish are based on average body weight and daily ration as opposed to minimum and maximum body weight.

a – USEPA 1993 b – Schreiber, 1976 c – Kastelein et al., 2002 d – Born et al., 2003

e – Aguado-Gimenez and Garcia-Garcia, 2005 f – Maldeniya, 1996 g – Wood et al., 2009

h – Stillwell and Kohler, 1993 i – Giffiths et al., 2009 j – Stillwell and Kohler, 1985

k – Becker, 1983 l – Carlander, 1969 m – Carlander, 1977

A.4 FISH SPECIES ANALYZED FOR CONTAMINANTS

Table A.4.1 Fish species analyzed for contaminants from estuarine sites. Number of sites from which each species was submitted, by NCCA region.

		NCCA Region			
Genus	**Species**	**Gulf**	**Northeast**	**Southeast**	**West**
Anguilla	*rostrata*		2		
Ariopsis	*felis*	94		9	
Bagre	*marinus*	44			
Bairdiella	*chrysoura*		1	4	
Brevoortia	*smithi*			4	
Brevoortia	*tyrannus*		2	1	
Caranx	*hippos*			2	
Centropristis	*striata*	1	10		
Cheilotrema	*saturnum*				1
Chriodorus	*atherinoides*			1	
Citharichthys	*sordidus*				6
Citharichthys	*stigmaeus*				3
Clupea	*harengus*		1		
Cymatogaster	*aggregata*				15
Cynoscion	*arenarius*	4			
Cynoscion	*nebulosus*			1	
Cynoscion	*regalis*		9	1	
Diplectrum	*formosum*	2			
Diplodus	*holbrookii*	1			
Elops	*saurus*	1			
Embiotoca	*lateralis*				1
Eucinostomus	*gula*			1	
Fundulus	*heteroclitus*		1		
Fundulus	*majalis*		1		
Genyonemus	*lineatus*				6
Haemulon	*plumierii*	3		1	
Haemulon	*sciurus*	1			
Hypsopsetta	*guttulata*				1
Ictalurus	*punctatus*		2	1	
Lagodon	*rhomboides*	16		10	
Leiostomus	*xanthurus*	17	7	16	
Lepidopsetta	*bilineata*				3
Leptocottus	*armatus*				27
Limanda	*ferruginea*		1		
Lutjanus	*campechanus*	1			

		NCCA Region			
Genus	**Species**	**Gulf**	**Northeast**	**Southeast**	**West**
Lutjanus	*griseus*	5		1	
Lutjanus	*synagris*	4			
Menidia	*menidia*		12		
Menticirrhus	*americanus*		4	9	
Menticirrhus	*littoralis*			1	
Menticirrhus	*saxatilis*		2		
Merluccius	*bilinearis*		1		
Micropogonias	*undulatus*	34	7	6	
Morone	*americana*		18	2	
Morone	*saxatilis*		6		
Mustelus	*canis*		4		
Opsanus	*tau*		2		
Orthopristis	*chrysoptera*	2			
Paralabrax	*clathratus*				1
Paralabrax	*maculatofasciatus*				4
Paralabrax	*nebulifer*				3
Paralichthys	*californicus*				22
Paralichthys	*dentatus*		17		
Peprilus	*triacanthus*		1		
Platichthys	*stellatus*				3
Pollachius	*virens*		1		
Pomatomus	*saltatrix*		5	2	
Prionotus	*carolinus*		2		
Prionotus	*evolans*		1		
Prionotus	*scitulus*	1			
Pseudopleuronectes	*americanus*		25		
Sciaenops	*ocellatus*	1			
Scomber	*scombrus*		9		
Scophthalmus	*aquosus*		1		
Sphoeroides	*maculatus*		1		
Stenotomus	*chrysops*		49		
Tautogolabrus	*adspersus*		6		
Urophycis	*chuss*		1		
Zoarces	*americanus*		1		

Table A.4.2 Fish species analyzed for contaminants from the Great Lakes.

Genus	species	Great Lakes
Alosa	*pseudoharengus*	6
Ambloplites	*rupestris*	2
Ameiurus	*nebulosus*	1
Aplodinotus	*grunniens*	22
Catostomus	*catostomus*	24
Catostomus	*commersonii*	22
Coregonus	*artedi*	1
Coregonus	*clupeaformis*	54
Cyprinus	*carpio*	4
Dorosoma	*cepedianum*	8
Esox	*lucius*	2
Ictalurus	*punctatus*	6
Lepomis	*gibbosus*	1
Lota	*lota*	3
Luxilus	*cornutus*	1
Micropterus	*dolomieu*	24
Micropterus	*salmoides*	3
Morone	*americana*	6
Morone	*chrysops*	5
Moxostoma	*carinatum*	1
Moxostoma	*macrolepidotum*	1
Neogobius	*melanostomus*	22
Oncorhynchus	*kisutch*	2
Oncorhynchus	*mykiss*	1
Osmerus	*mordax*	1
Perca	*flavescens*	50
Pomoxis	*nigromaculatus*	1
Prosopium	*cylindraceum*	5
Salvelinus	*namaycush*	5
Sander	*vitreus*	23
Species not reported		1

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