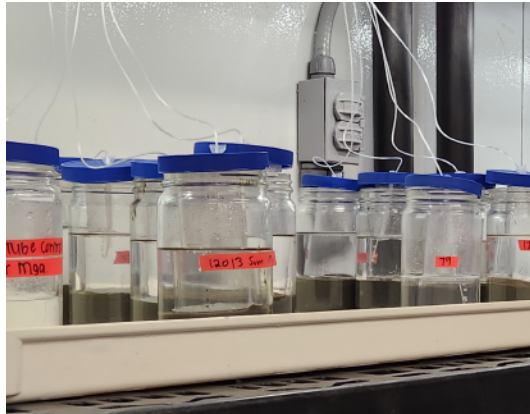




Sediment Chemistry

BIGHT '23



Southern California Bight
2023 Regional Monitoring Program
Volume II

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Southern California Bight 2023 Regional Monitoring Program: Volume II. Sediment Chemistry

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FOREWORD

The 2023 Southern California Bight Regional Monitoring Program (Bight '23) is a collaborative effort to provide an integrated large-scale assessment of the Southern California Bight (SCB). The Bight '23 survey is a continuation of previous regional assessments conducted in 1994, 1998, 2003, 2008, 2013, and 2018. The collaboration represents the joint efforts of over 100 organizations.

Bight '23 is organized into eight elements: (1) Sediment Quality (formerly Contaminant Impact Assessment/ Coastal Ecology); (2) Microbiology; (3) Water Quality/ Ocean Acidification and Hypoxia; (4) Harmful Algal Blooms; (5) Trash and Microplastics; (6) Estuaries; (7) Submerged Aquatic Vegetation; and (8) Bioaccumulation. This assessment report presents the results of the sediment chemistry portion of the survey, which is one component of the Sediment Quality element. Other Sediment Quality components include sediment toxicity and benthic infauna. Copies of this and other Bight '23 reports, as well as work plans and quality assurance plans, are available for download at www.sccwrp.org.

ACKNOWLEDGEMENTS

This report is a result of the dedication and hard work of many individuals who share a common goal of improving our understanding of the environmental quality of the Southern California Bight. The authors wish to thank the members of the Bight '23 Chemistry Technical Committee for their assistance with study design, sample analysis, data analysis, and report review. We also thank the Bight '23 Sediment Quality Planning Committee for their guidance and support of the application of sediment chemistry in regional monitoring. This study would not have been possible without the exceptional expertise of the field sampling personnel from the following organizations: Aquatic Bioassay and Consulting Laboratories, Channel Islands National Marine Sanctuary, City of Los Angeles, City of San Diego, Los Angeles County Sanitation Districts, MBC Aquatic Sciences, Orange County Sanitation District, Southern California Coastal Water Research Project, Weston Solutions, WSP, Anchor QEA, and U.S. Navy. We also wish to express our gratitude to Rich Gossett (Physis Laboratories) for data interpretation and discussion; Kelsey Cherland and Paul Smith (SCCWRP) for data submission; Abel Santana (SCCWRP) for map preparation; and Wayne Lao (SCCWRP) for preparation of field reference materials.

Sediment chemistry measurements were provided by the following laboratories: City of Los Angeles Environmental Monitoring Division, City of San Diego, Los Angeles County Sanitation Districts, Orange County Sanitation District, Physis Laboratories, and Weck Laboratories.

EXECUTIVE SUMMARY

Regional monitoring is an important mechanism for assessing the status and trends of human influences on coastal habitats across the Southern California Bight (SCB). The Southern California Bight 2023 Regional Monitoring Program (Bight '23) is the seventh in a series of regional marine monitoring efforts starting in 1994 and repeating in 1998, 2003, 2008, 2013, and 2018. Over 100 different organizations encompassing regulatory, regulated, academic, and non-governmental agencies collaborated in the Bight '23 Program.

The Sediment Chemistry element sought to address two primary questions:

1. What is the extent and magnitude of sediment chemistry exposure in the SCB?
2. How does the extent and magnitude of sediment chemistry exposure vary over time in the SCB?

Assessment of the toxicity, infauna, algal toxins (domoic acid), trash and microplastics, and extent and magnitude of bioaccumulation are reported at:

<https://www.sccwrp.org/about/research-areas/regional-monitoring/southern-california-bight-regional-monitoring-program/bight-program-documents/>. This URL contains all planning documents and assessments reports for all Bight cycles.

A stratified random sampling design was selected to ensure an unbiased sampling approach to areal assessments of environmental conditions. Ten strata were selected for this study, including three continental shelf depth strata (inner: 5-30 m, middle: 30-120 m, outer: 120-200 m), upper slope (200-500 m), lower slope (500-1000 m), Channel Islands (30-120 m), and four embayment strata (marinas, ports, bays, and estuaries). These strata were selected to represent a range of natural and potentially affected habitats, and are inclusive of most of the habitats sampled in previous Bight surveys. A total of 340 stations were sampled between July and October 2023.

Sediment samples were analyzed for grain size, total organic carbon, total nitrogen, 15 trace metals, polycyclic aromatic hydrocarbons (PAHs, sum of 24 compounds), polychlorinated biphenyls (PCBs, sum of 43 congeners), dichlorodiphenyltrichloroethane compounds (DDTs, sum of two isomers and 5 degradation products), and chlordanes (sum of 5 compounds). Five groups of organic constituents of emerging concern (CECs) were measured in Bight '23 including 13 polybrominated diphenyl ether (PBDE) flame retardant congeners, 8 pyrethroid compounds, 5 neonicotinoid insecticides, one tire-wear compound (6-*p*-phenylenediamine-quinone or 6-PPD-quinone), and two per- and polyfluorinated alkyl substances (PFAS) (perfluorooctanesulfonic acid or PFOS, and perfluorooctanoic acid or PFOA).

An estimated 83% of the sediment seafloor Bight-wide were considered minimal to low exposure to marine organisms according to California's Sediment Quality Objectives (SQO) assessment framework. Only 1% of SCB sediments in Bight '23 were in the worst category of high exposure. The relative extent of sediment contamination was generally greater in embayments than in offshore strata, due to greater proximity of contaminants originating from anthropogenic activities in embayments. DDT also had high concentrations offshore, due to historical discharges.

Over time, the areal extent of sediment condition that met SQO guidelines, defined as minimal or low chemical exposure using the SQO assessment framework, ranged from 61% to 83% over the five surveys from 2003 to 2023. Over the same period, the areal extent of high exposure to sediment contamination remained low at <3%. Apparent increases over time in sediment conditions meeting SQO guidelines were not statistically significant Bight-wide. No statistically significant areal changes in sediment conditions were observed for individual strata over time, except in the composite slope strata (upper and lower slope), which increased significantly from 53% in 2003 to 77% in 2023. This increase was driven by significant increases in the area of the lower slope meeting SQO guidelines, from 55% in 2003 to 83% in 2023. These observations suggest some modest improvement in sediment condition in the Bight, which has been subject to substantial increases in population and anthropogenic activity and stressors during the course of the Bight program while maintaining similar SQO conditions throughout.

New chemical analytes introduced in Bight '23 were PFAS, neonicotinoid insecticides, and tire-wear compounds. The major PFAS compounds PFOS and PFOA were generally found in embayments and along the coastline from Santa Monica into Ventura County, with mean and maximum concentrations Bight-wide of 1.2 ng/g and 68 ng/g, respectively, for PFOS, and 0.2 and 7.1 ng/g, respectively, for PFOA. The highest concentrations of PFAS were found at isolated spots in heavily impacted embayment sediments, although insufficient data and scientific consensus currently exist for adoption of any such sediment thresholds. Neonicotinoids were generally only detected at some embayment sites. Porewater in equilibrium with observed sediment concentrations may be below some current aquatic toxicity guidelines for neonicotinoids, but not others. The tire-wear degradate 6-PPD-quinone was sporadically detected, generally only at some embayment sites, with a maximum concentration of 18.6 ng/g. These concentrations are above some suggested freshwater thresholds for protection of extremely sensitive salmonids, such as coho salmon. Concentrations of all three classes of these CECs were comparable to sediment concentrations observed in heavily urbanized and industrialized sediments worldwide.

Several recommendations for regional monitoring arose from this study. First, the selection of chemical contaminants for future Bight surveys may consider a risk-based prioritization process

for CECs. This would provide a suitable focus on those chemicals of greatest interest and concern based on their occurrence, environmental fate and transport, and potential for human and ecosystem impacts. Such a process would allow for inclusion of new CECs over time, as well as off-boarding of chemicals, for example those that no longer rise in prioritization to warrant inclusion. The list of current CEC analytes can also be expanded or reduced based on existing observations of extent, magnitude, and potential for effects (or lack thereof). Finally, emphasis should be placed on understanding the extent and magnitude of chemical stressors in aqueous phase rather than just sediments. More polar toxic chemicals (e.g., tire-wear compounds, neonicotinoids, and PFAS) may be in higher concentrations in the water column and porewaters than sediment, and changes in dissolved-phase concentrations may change concentrations in the porewater and in the sediment (e.g., from concentration gradients that may develop).

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LIST OF COMMON ABBREVIATIONS

6-PPD-quinone	<i>N</i> -(1,3-dimethylbutyl)- <i>N'</i> -phenyl- <i>p</i> -phenylenediamine quinone
8:2 FTS	8:2 Fluorotelomer sulfonic acid
ACE	Acetamiprid
AFFF	Aqueous film-forming foam
AWM	Area weighted mean
CA LRM	California Logistic Regression Model
CEC	Constituent of Emerging Concern
CLAEMD	City of Los Angeles, Environmental Monitoring Division
CLO	Clothianidin
CRM	Certified Reference Material
CSD	City of San Diego
CSI	Chemical Score Index
DBP	Dichlorobenzophenone
DDA	2,2-bis(4-Chlorophenyl) acetic acid
DDD	Dichlorodiphenyldichloroethane
DDE	Dichlorodiphenyldichloroethylene
DDM	Dichlorodiphenylmethane
DDMU	1,1-bis(4-chlorophenyl)-2-chloroethylene
DDNU	1,1-bis(4-chlorophenyl)ethylene
DDT	Dichlorodiphenyltrichloroethane
DQO	Data quality objective
EPA	United States Environmental Protection Agency
GC	Gas chromatography

GC/ECD	Gas chromatography/electron capture detection
GC/MS	Gas chromatography/mass spectrometry
HDPE	High density polyethylene
HFPO-DA	Hexafluoropropylene oxide dimer acid
IMI	Imidacloprid
LACSD	Los Angeles County Sanitation Districts
LC50	Median lethal concentration
LC/MS/MS	Liquid chromatography/tandem mass spectrometry
MDL	Method detection limit
MRM	Multiple reaction monitoring
N/A	Not applicable
ND	Not detected
NPDES	National Pollutant Discharge Elimination System
OC San	Orange County Sanitation District
PAH	Polycyclic aromatic hydrocarbon
PBDE	Polybrominated diphenyl ether
PCB	Polychlorinated biphenyl
PFAS	Per- and polyfluorinated alkyl substance
PFHxS	Perfluorohexanesulfonic acid
PFNA	Perfluorononanoic acid
PFOA	Perfluorooctanoic acid
PFOS	Perfluorooctanesulfonic acid
OC	Organochlorine
QA	Quality assurance

QA/QC	Quality assurance/quality control
QC	Quality control
PT	Performance Testing standard
RL	Reporting limit
RPD	Relative percentage difference
SCB	Southern California Bight
SCCWRP	Southern California Coastal Water Research Project
SRM	Standard Reference Material
SQO	Sediment Quality Objective
SWAMP	Surface Water Ambient Monitoring Program
THA	Thiacloprid
THM	Thiamethoxam
TMDL	Total maximum daily load
TN	Total nitrogen
TOC	Total organic carbon

I. INTRODUCTION

Overview

The Southern California Bight (SCB) is an important and unique ecological area between Point Conception, California and the U.S.-Mexican border. The SCB is situated within an eastern boundary upwelling system, in which seasonal upwelling of nutrient-rich waters supports large-scale primary productivity along the coastline (Capone and Hutchins 2013; Chavez and Messié 2009). The topography of the SCB is complex, with offshore islands, submarine canyons, ridges, and basins (Dailey et al. 1993). This diversity of habitats, coupled with high productivity, sustains a biologically diverse coastal ocean environment (Dailey et al. 1993; Dawson 2001). In addition, the SCB is a major migration route, and has among the most diverse marine bird and mammal populations in north temperate waters.

The State of California has the nation's largest ocean-based economy, with a \$51 billion contribution to the state's gross domestic product (GDP) as of 2021 (NOAA 2025); of this amount, \$20 billion arises from the five SCB coastal counties (San Diego, Orange, Los Angeles, Ventura, and Santa Barbara), employing over 200,000 people. Los Angeles/Long Beach (LA/LB) Harbor is the largest commercial port in the U.S., and San Diego Bay is home to some of the largest naval facilities in the country. In 2013, commercial fishermen landed over 160 million kg of seafood in the SCB with an ex-vessel value (amount paid to fishermen) of nearly \$256 million (SeaGrant 2025). Tourism and recreation accounted for approximately \$9 billion of GDP in the five SCB coastal counties (NOAA 2015).

The coastal areas that form the SCB are also among the most densely populated regions in the country, with an estimated population of 17 million in 2025 (State of California 2025). This population density creates potential stresses upon the adjacent marine environment through the input of sediment, nutrients, pathogens, and toxic chemicals via runoff, treated wastewater effluent, impacts from shipping and ports, residential and commercial activities, and other activities resulting from the high population and density in the SCB. Population growth and economic activity have resulted in the conversion of open land into urban surfaces and the loss of historical estuarine habitat, e.g., 48% since 1850 (Stein et al. 2014). Many urban surfaces are not permeable, which can increase the rate of runoff. There are 17 major watersheds that discharge surface runoff from urban and agricultural land uses to the SCB (Schiff and Tiefenthaler 2011). In addition, ocean wastewater outfalls collectively discharge 4.1 billion L of treated effluent daily (Sutula et al. 2021).

Historically, monitoring for sediment quality had been focused on areas closest to regulated discharges associated with National Pollutant Discharge Elimination System (NPDES) permits. Such monitoring is primarily intended to assess end-of-pipe regulatory compliance (Schiff et al.

2002), for the purposes of determining environmental impacts of a discharge on the surrounding receiving environment, and ascertaining what strategies might be further needed to protect it. This monitoring does not necessarily answer important questions regarding the health of coastal waters. However, the approach of monitoring discrete point sources was designed to evaluate the impacts of individual discharges, and only cover a small fraction of the total SCB area. Accordingly, they provide a non-representative and potentially biased perspective of coastal habitat health.

To address this issue, nearly 100 regulated, regulatory, nongovernmental, and academic organizations have combined resources to implement the SCB Regional Marine Monitoring Program. The Bight Program, as it is commonly called, is a probabilistic survey intended to assess the regional condition of SCB habitats to provide needed context for NPDES and Total Maximum Daily Load (TMDL) monitoring (Schiff et al. 2019). This regional monitoring program, which started in 1994 and has been conducted every five years thereafter (Table 1), allows assessment of large-scale reference conditions that cover a greater range of natural variability observed in the SCB, in contrast to comparing an individual discharge to a small number of local reference sites.

Beginning with the 2008 survey, the Sediment Quality component of the Bight Program evaluates potential impacts on marine benthic communities through multiple lines of evidence that are compatible with the California Sediment Quality Objectives Program (SQO): sediment chemistry, biological assemblages, and sediment toxicity (Bay and Weisberg 2008). Using the standardized SQO assessment methods allows for quantitative comparison of the Bight Sediment Quality results to other regions of the state, and provides a mechanism to detect and monitor changes over time.

The focus of this report is sediment chemistry, a key component of the overall assessment of sediment quality using the SQO tool. Chemical measurements provide much-needed information on the magnitude of contamination by specific chemical contaminants across the region. Substantial effort in developing analytical comparability was invested in previous surveys (Gossett et al. 2003), as a necessary part of having suitable data quality. Because all the regional programs were conducted in a collaborative fashion with multiple analytical laboratories participating, intercalibration studies were a focal point for trace metal and trace organic constituents. While all the laboratories were accredited by the State of California, there has been, at times, significant discrepancy for specific constituents. Therefore, iterative intercalibration exercises were performed until all the laboratories could meet prescribed data quality objectives for interlaboratory accuracy and precision. These intercalibrations remain one of the foundational elements of the regional monitoring quality assurance/quality control program and ensure a robust regional monitoring program. Other reports focus on sediment

toxicity, benthic infauna, bioaccumulation, trash and microplastics, harmful algal blooms, and ocean acidification.

Objectives of the 2023 Regional Monitoring Program

The sediment chemistry portion of the Bight '23 Sediment Quality Element aimed to address two questions:

1. What is the extent and magnitude of sediment chemistry exposure in the SCB?
2. How does the extent and magnitude of sediment chemistry exposure vary over time in the SCB?

Answering these questions addresses the management needs for assessing the overall environmental health of the SCB, describing regional reference conditions, and developing regional assessment tools. Given the probabilistic study design of the Bight Program, characterization of the breadth and depth of variability in sediment chemistry exposure of multiple habitats, and the region overall, will together provide needed context for local NPDES monitoring and to bridge areas beyond such efforts. Furthermore, because sediment chemistry exposure was evaluated in ten habitats, or strata, during Bight '23, relative habitat quality between habitats can also be described. The strategy to revisit a subset of sites during each Bight survey allows for characterization of site-specific trends in sediment chemistry exposure for the region. State and local agencies have made significant investments in improving water quality and treatment. Long-term monitoring, like the Bight Program, provides a means to document the impact of these management actions on regional sediment quality and the relative rate of those impacts.

This report is structured in eight chapters and includes sections on Methods (Section II), Quality Assurance/Quality Control (Section III), Descriptive Results (Section IV), Assessment Results (Section V), Discussion (Section VI), Conclusions (Section VII), and Recommendations (Section VIII). Appendices detail intercalibration results, maps of all analytes, and cumulative frequency distributions of analytes.

Table 1: Summary of participating organizations (e.g., providers of field crews and/or laboratory services), number of strata, and number of sampling stations for Bight Regional Survey Sediment Monitoring Programs.

Year	# of participants	# of strata	# of stations
1994	12	4	264
1998	60	10	404

Year	# of participants	# of strata	# of stations
2003	60	11	359
2008	90	10	383
2013	34	12	397
2018	46	11	376
2023	48	10	340

II. METHODS

Sampling Design

The Bight regional monitoring program uses a stratified random sampling design to ensure an unbiased sampling approach to provide areal assessments of environmental conditions (Stevens 1997). There were ten strata selected for this study. These include three continental shelf strata (inner shelf at 5-30 m depth, mid-shelf at 30-120 m, and outer shelf at 120-200 m), upper slope (200-500 m), lower slope (500-1000 m), Channel Islands, and the embayment strata of bays, estuaries, ports, and marinas (Figure 1). Brackish estuaries, introduced in Bight '18, were not incorporated as a separate stratum for Bight '23 although some participating organizations did sample within them. Stratification ensured that an appropriate number of stations and resultant samples (Table 2) were allocated to characterize each stratum with adequate precision. The goal was to allocate 30 sites to each stratum to yield a 90% confidence interval of ca. $\pm 10\%$ for estimates of areal extent, assuming a binomial probability distribution and $p = 0.2$. This level of precision was selected, as lower differences in response are unlikely to yield different management decisions. Enhancement of the sampling design was achieved through intensified sampling in targeted areas, driven by stakeholder needs, and area-weights were adjusted to account for intensification in certain areas (e.g., San Diego Bay, Newport Bay, Ports of Los Angeles and Long Beach). One stratum could not meet the target of 30 sites per stratum, namely Channel Islands. In this stratum, sediments were collected from 14 stations, all of which were revisits. To assess how extent and magnitude of site-specific sediment quality vary over time, approximately half of the Bight '23 sample sites were sampled in previous surveys. In all, sediment grabs were successful at 340 sites (Table 2). Six sites beyond this total were abandoned, with no replacement, for various logistical or technical reasons (1 upper slope, 1 outer shelf, 3 mid-shelf, and 1 Channel Islands).

Table 2: Sample size by stratum in Bight '23.

Stratum	New stations	Revisited stations	Total number of stations
Inner shelf	15	15	30
Mid-shelf	13	16	29
Outer shelf	13	14	27
Upper slope	14	16	30
Lower slope	14	15	29
Channel Islands	0	14	14
Bays	26	21	47
Estuaries	20	13	33
Marinas	30	16	46
Ports	38	17	55

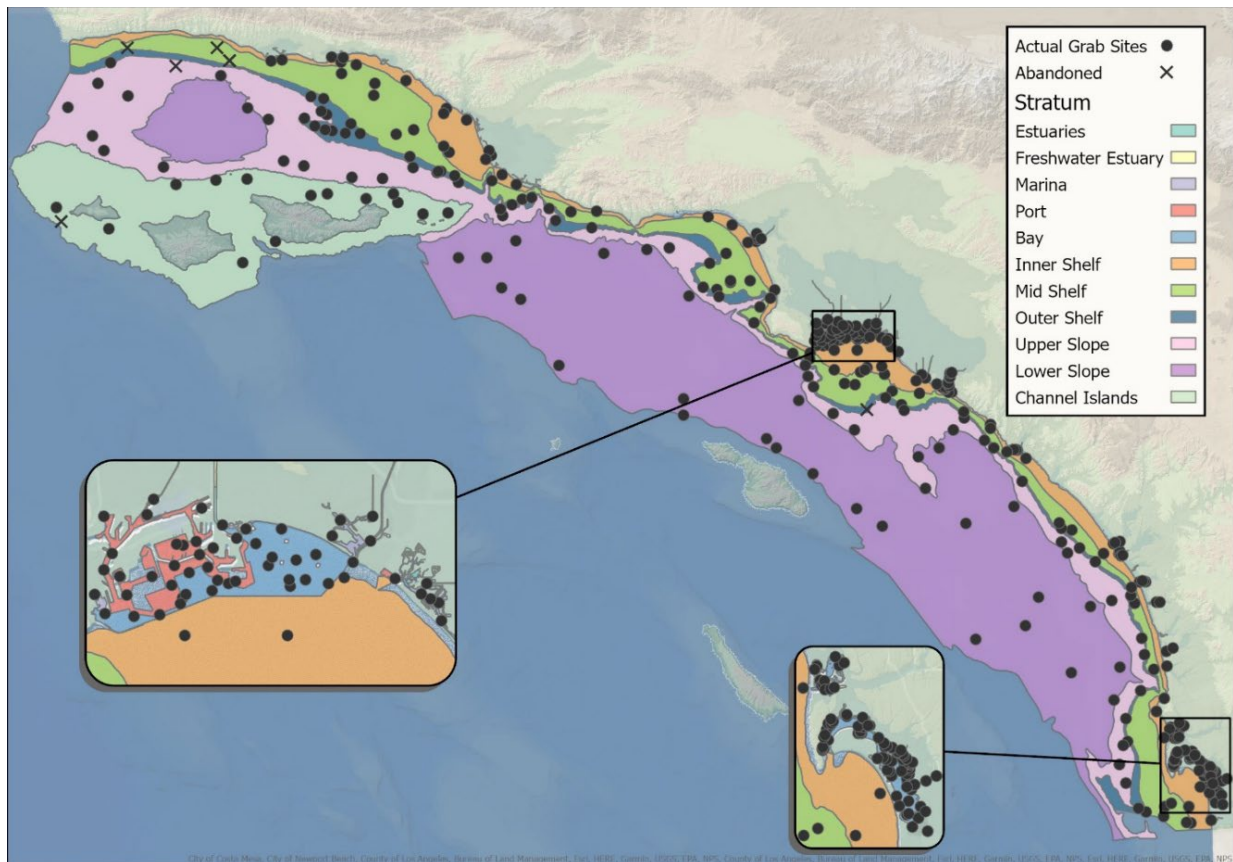


Figure 1: Strata and sediment grab sites in Bight '23.

Sample Collection

Sediment samples were collected using a 0.1 m² modified Van Veen grab sampler (Stubbs et al. 1987) or a 6-inch sediment push core at wadable depths (estuaries) within 100 m and 10% of the water depth of the location specified by the sampling design. Up to 6.0 L of sediment were collected for measurement of both sediment toxicity and sediment chemistry. A scoop was used to collect sediment from the top 2 cm (offshore stations) or top 5 cm (embayment stations) of the undisturbed surface material in the grab. This scoop was made of metal prerinse with alcohol and distilled water for the collection of sediment for PFAS, trace metals, and organics analysis. The scoop was made of plastic materials such as high-density polyethylene (HDPE), polycarbonate, or Teflon for TOC and grain size. Collection of sediments for PFAS analysis was done before that for other chemical analytes, and before allocation of sediment for toxicity testing, to minimize contamination from sampling materials for the latter that may contain PFAS (e.g., Teflon items). Contact with sediment within 1 cm of the sides of the grab was avoided in order to minimize cross-contamination. In most cases, multiple grabs were required to obtain enough sediment for both toxicity testing and chemical analysis. If more than one grab was required, sediment from each grab that was not allocated for PFAS

analysis was added to a Teflon bag and homogenized thoroughly using either a clean Teflon or plastic spoon, or by kneading the sample within the bag. After homogenization, sub-samples were aliquoted for non-PFAS chemical analyses. The remaining contents of the bag were used for toxicity testing. Homogenization of sediments prior to subsampling for non-PFAS chemistry and toxicity was required for all embayment stations. For offshore sites, the contents of multiple grabs could be homogenized as was done for the embayment sites. Alternatively, samples could be distributed directly to HDPE containers for toxicity and PFAS analysis, and to glass jars for other chemical analytes, by placing approximately equal aliquots of sediment from the surface of a grab sample into each container type.

Subsamples for chemical analysis were placed in appropriate containers for the subsequent analysis. For PFAS, vendor-certified PFAS-free HDPE containers (250 mL) were used, while glass containers with Teflon[®]-lined closures (250 mL) meeting Environmental Protection Agency (EPA) standards were used for all other analytes. Samples were stored frozen (-20 °C) until analyzed, except for those for grain size, which were stored at 4 °C. Further details on the sample collection procedures, including efforts to abate and evaluate potential contamination (e.g., for PFAS) used in this study, can be found in the Bight '23 Field Operations Manual (Bight '23 Sediment Quality Committee 2023). As soon as possible after collection, samples were distributed to the appropriate laboratories for analysis. A summary of the division of effort for the Bight '23 chemistry component as a function of parameter and laboratory is given in Table 3.

Table 3: Sediment chemistry laboratory effort for Bight '23. Laboratory name abbreviations: CLAEMD = City of Los Angeles, Environmental Monitoring Division; CSD = City of San Diego; LACSD = Los Angeles County Sanitation Districts, OC San = Orange County Sanitation District.

Parameter	CLAEMD	CSD	LACSD	OC San	Physis	Weck	Total sample number
Grain size	30	63	0	45	200	0	338
Total Organic Carbon (TOC)	30	63	33	42	169	0	337
Total nitrogen	29	63	0	40	170	33	335
Metals	30	63	33	45	169	0	340
Polycyclic aromatic hydrocarbons (PAHs)	30	61	33	45	171	0	340
Polychlorinated biphenyls (PCBs)	30	61	33	45	171	0	340
Organochlorine pesticides (OCs)	30	61	33	45	171	0	340
Polybrominated diphenyl ethers (PBDEs)	0	0	0	0	202	0	202
Pyrethroids	0	0	0	0	206	0	206
Neonicotinoids	0	0	16	0	140	0	156
Tire-wear compounds	0	0	16	0	89	68	168
Per- and polyfluorinated alkyl substances (PFAS)	0	0	16	0	134	68	218
Total number of samples	209	435	213	307	1992	169	3320

Analytical Methods

Analytical methods employed were at the discretion of the participating laboratories, contingent upon their ability to demonstrate acceptable analytical performance determined by strict adherence to common quality assurance/quality control (QA/QC) practices, routine analysis of certified reference materials (CRMs) and successful participation in an inter-laboratory calibration study to demonstrate compliance with specified performance criteria of

Bight '23 prior to any sample analysis. Each laboratory was required to demonstrate its capability to meet the stated data quality objectives (DQOs) for each of the target analytes. Initially, each laboratory established a method detection limit (MDL) for each target analyte following the MDL protocol cited in EPA 40 CFR Part 136 Appendix B, Revision 2 of 82 FR 40836 (dated 1 July 2021), and laboratories were required to meet the study's stated reporting levels (RLs). Analytical performance criteria and the DQOs for each quality control sample type can be found in the Bight '23 Survey Quality Assurance Plan (Bight '23 Sediment Quality Committee 2023a). Further details and assessment of quality assurance for Bight '23 chemistry are found in Section III.

Target Analytes

The analytes for Bight'23 (Table 4) are similar to those measured in Bight '18, with several additions and one deletion. The 15 inorganic analytes (hereafter referred to as "metals") were selected from those commonly monitored in the SCB, such as by NPDES permittees. The 24 PAHs include the 16 EPA priority pollutant PAHs (marked by asterisks in Table 4), as well as 8 additional compounds, including 5 methylated PAHs. Due to cost and capability constraints, not all polychlorinated biphenyl congeners (PCBs) were targeted. The 43 congeners analyzed for Bight '23 were selected based on their potential toxicity and occurrence in the commonly used and subsequently discharged mixtures: Aroclors 1242, 1248, 1254, and 1260. These congeners represent the most prevalent and toxic congeners of the total 209 PCB congeners. Many of these targeted congeners are also associated with assessment (Bay et al. 2021) of Sediment Quality Objectives (SQO). The 12 organochlorine (OC) pesticides were selected based on known abundance and impacts in previous Bight surveys. Fipronil, analyzed during Bight '18, was not analyzed in Bight '23, given low occurrence and limited potential for toxicity (Du et al. 2022). Five groups of emerging contaminants were measured in Bight '23. These included 13 polybrominated diphenyl ether (PBDE) flame retardant congeners, 8 pyrethroid pesticides, 5 neonicotinoid insecticides, one tire-wear compound, and two per- and polyfluorinated alkyl substance (PFAS) compounds. The PBDEs and pyrethroids were first measured in Bight '08 special studies. Bight '23 is the first SCB regional monitoring program to measure neonicotinoids, tire-wear compounds, and PFAS. It should be noted that while only PFOS and PFOA were included as PFAS analytes for Bight '23, other PFAS analytes were included in the Bight '23 database if laboratories had QA/QC for such additional analytes that were equivalent to that for PFOS and PFOA. These other PFAS analytes were also analyzed by the accredited methods used for PFOS and PFOA (e.g., EPA Method 1633), but will not be discussed further in this report unless otherwise noted.

Table 4: Sediment chemistry target analytes for Bight '23. *denotes EPA priority polycyclic aromatic hydrocarbon (PAH).

Target analyte group	Number of analytes	Target analytes
Metals	15	Aluminum (Al), Antimony (Sb), Arsenic (As), Barium (Ba), Beryllium (Be), Cadmium (Cd), Chromium (Cr), Copper (Cu), Iron (Fe), Lead (Pb), Mercury (Hg), Nickel (Ni), Selenium (Se), Silver (Ag), Zinc (Zn)
Polycyclic aromatic hydrocarbons (PAHs)	24	1,6,7-trimethylnaphthalene, 1-methylnaphthalene, 1-methylphenanthrene, 2,6-dimethylnaphthalene, 2-methylnaphthalene, *acenaphthene, *acenaphthylene, *anthracene, *benz[a]anthracene, *benzo[a]pyrene, *benzo[b]fluoranthene, benzo[e]pyrene, *benzo[g,h,i]perylene, *benzo[k]fluoranthene, biphenyl, *chrysene, *dibenz[a,h]anthracene, *fluoranthene, *fluorene, *indeno[1,2,3-c,d]pyrene, *naphthalene, perylene, *phenanthrene, *pyrene
Polychlorinated biphenyls (PCBs)	43	PCB congeners 8, 18, 28, 37, 44, 49, 52, 66, 70, 74, 77, 81, 87, 99, 101, 105, 110, 114, 118, 119, 123, 126, 128, 138, 149, 151, 153, 156, 157, 158, 167, 168, 169, 170, 177, 180, 183, 187, 189, 194, 195, 201, 206
Organochlorine pesticides (OCs)	12	4,4'-DDT, 2,4'-DDT, 4,4'-DDD, 2,4'-DDD, 4,4'-DDE, 2,4'-DDE, 4,4'-DDMU, alpha-chlordane, gamma-chlordane, <i>cis</i> -nonachlor, <i>trans</i> -nonachlor, oxychlordane
Polybrominated diphenyl ethers (PBDEs)	15	PBDE congeners 17, 28, 47, 49, 66, 85, 99, 100, 138, 153, 154, 183, 190
Pyrethroid insecticides	8	bifenthrin, cyfluthrin (total), cypermethrin (total), I-cyhalothrin (total), <i>cis</i> -permethrin, <i>trans</i> -permethrin, deltamethrin, esfenvalerate
Neonicotinoid insecticides	5	acetamiprid, clothianidin, imidacloprid, thiacloprid, thiamethoxam
Tire-wear compounds	1	6-PPD-quinone
Per- and polyfluorinated alkyl substances (PFAS)	2	perfluorooctane sulfonic acid (PFOS), perfluorooctanoic acid (PFOA)
Other parameters	4	grain size, total organic carbon (TOC), total nitrogen (TN), moisture content

Grain Size

All grain size samples were screened through 1000 and 2000 μm sieves prior to analysis to remove methodological interferences and bias. The sample fraction greater than 2000 μm was designated as gravel. The sample fraction smaller than 1000 μm (i.e., ϕ of 0 to 14) was analyzed using a Horiba or Micromeritics Saturn II Digisizer. Size distributions were computed by summarizing the set of numbers for each size classification, which were represented by each channel detector. All categories < 63 μm were considered fine-grained material (silts + clays) and are referenced herein as percent fines.

Total Organic Carbon and Total Nitrogen

Total organic carbon (TOC) and total nitrogen (TN) analyses were performed using an elemental analyzer, in which samples were combusted at high temperature (>1000 °C) and the carbon or nitrogen content measured in the evolved gas. Frozen sediments were first thawed to room temperature and homogenized before being dried in an air oven at 60 °C overnight. The dried samples were then exposed to concentrated hydrochloric acid vapors in a closed container to remove inorganic carbon. The acid-treated samples were again dried and weighed, and crimped in a tin or silver capsule prior to analysis as described above e.g., EPA Method 9060.

Metals

Samples, not including those for mercury, were digested in strong acid (e.g., EPA Methods 200.7, 200.8, 6010B, 6020, and 7471B). The resulting digestates were diluted to a specific volume with deionized water and subsequently analyzed by one or more of the following instrumental methods, depending on the laboratory: inductively coupled plasma mass spectrometry, inductively coupled plasma emission spectroscopy, flame atomic absorption, or graphite furnace atomic absorption. Samples for mercury were prepared as per EPA Methods 245.5 or 7471B. All laboratories analyzed mercury using cold vapor atomic absorption spectroscopy.

Trace Organics

Trace organic chemistry samples were solvent-extracted using one of the following methods: accelerated solvent extraction, Soxhlet, sonication, or liquid solvent extraction with solid-phase extraction cleanup. The extracts obtained were subjected to each laboratory's own clean-up procedures and were analyzed by an appropriate gas chromatographic method. PCB congeners and organochlorine pesticides were analyzed using either dual-column gas chromatography (GC) with electron capture detection (GC/ECD), mass spectrometry (GC/MS) in the selected ion monitoring (SIM) mode, or tandem mass spectrometry (GC/MS/MS) using multiple reaction monitoring (MRM). All laboratories analyzed PAHs and PBDEs by GC/MS. Pyrethroids were analyzed by GC/MS (EPA Method 8270). Neonicotinoids and tire-wear compounds were

analyzed by either GC/MS (EPA 8270) or by liquid chromatography-tandem mass spectrometry (LC/MS/MS) using MRM. PFAS analytes were analyzed by LC/MS/MS.

Data Analysis

The sediment chemistry data from Bight '23 was analyzed to determine descriptive statistics of sediment contamination, and to assess the extent and magnitude of sediment contamination. Specific aims are described as follows.

Descriptive statistics had two specific areas of focus. The first consisted of distributions and central tendencies of parameter values, including the area-weighted mean (AWM) and confidence interval for each of the strata of interest and the SCB as a whole (Schiff et al. 2016). The second consisted of geographical distributions, including thematic maps of sediment concentrations by parameter.

Assessment of extent and magnitude had three areas of focus. The first was an estimation of the proportion of contaminant mass for each constituent relative to the amount of area occupied for individual strata. The second was an evaluation of sediment concentrations using chemistry indices, part of the SQO assessment framework established by the State of California (SWRCB 2018). The third was a comparison of the extent of sediment contamination to the results from previous surveys.

Data below a laboratory's MDL values were treated as zero when data were reported, except as indicated. One laboratory submitted values equal to its reporting limit for PFAS and tire-wear compounds. The same approach was adopted for the determination of descriptive statistics, including summation of analyte classes (e.g., the concentration of a non-detected PCB congener was considered zero in calculating total PCBs). A sensitivity analysis was performed in Section V in which data below MDLs were treated as either one-half the method detection limit or the full method detection limit to evaluate how method detection limit discrepancies affected SQO chemistry indices. Very limited changes were observed in this exercise, suggesting that the data set is robust enough that arbitrary selection of less-than values, which can be problematic (Helsel, 2006) did not much change the key points of this study. For the determination of chemical indices, data preparation followed the California Sediment Quality Objectives Technical Manual (Bay et al. 2021). For example, if all components of a sum were non-detected, then the highest reporting level of any one compound in the group should be used to represent the sum value.

Quantitative spatial analysis was performed using R (R Development Core Team 2015) and the Spatial Survey package (Diaz-Ramos et al. 1996; Kincaid and Olsen 2015). This function estimated the AWM concentrations, area weighted chemical index scores, and the

corresponding confidence intervals. The 95% confidence intervals about the mean were calculated as 1.96 times the standard error.

Evaluation of Chemical Exposure

Following the procedure first used in Bight '08, the SQO chemistry indices for bays and estuaries were used to assess chemical exposure. SQO chemistry is one of the three lines of evidence, along with sediment toxicity and benthic infauna community structure, used for the objective of benthic community protection (Bay et al., 2021).

For each line of evidence, an evaluation of condition is made, then the three lines of evidence are combined for a final site assessment. In the case of sediment chemistry, concentrations of selected chemical constituents of those measured (Table 4) were evaluated using two chemistry indices: the Chemical Scoring Index (CSI) and California Logistic Regression Model (CA LRM). Results from the two indices were combined to determine the chemical exposure category.

The four chemistry exposure categories are:

1. Minimal Exposure - Sediment-associated contamination may be present, but exposure is unlikely to result in effects.
2. Low Exposure - Small increase in contaminant exposure that may be associated with increased effects, but the magnitude or frequency of occurrence of biological impacts is low.
3. Moderate Exposure - Clear evidence of sediment contaminant exposure at concentrations that are likely to result in biological effects.
4. High Exposure - Contaminant exposure is highly likely to result in substantial biological effects.

Sediments scoring minimal or low chemical exposure are deemed by the SQO assessment framework as meeting guidelines for the chemical line of evidence (Bay et al. 2021).

The analytes required to calculate the chemical indices are a subset of those measured in the Bight survey: cadmium, copper, lead, mercury, zinc, alpha-chlordane, gamma-chlordane, *trans*-nonachlor, 4,4'-DDT, Σ HPAH (high molecular weight PAHs, consisting of benzo[*a*]anthracene, benzo[*a*]pyrene, benzo[*e*]pyrene, chrysene, dibenz[*a,h*]anthracene, fluoranthene, perylene, and pyrene), Σ LPAH (low molecular weight PAHs, consisting of acenaphthene, anthracene, phenanthrene, biphenyl, naphthalene, 2,6-dimethylnaphthalene, fluorene, 1-methylnaphthalene, 2-methylnaphthalene, and 1-methylphenanthrene), Σ DDD (2,4'-DDD and

4,4'-DDD), Σ DDE (2,4'-DDE and 4,4'-DDE), Σ DDT (2,4'-DDT and 4,4'-DDT), and Σ PCB (PCBs 8, 198, 28, 44, 52, 66, 101, 105, 110, 118, 128, 138, 153, 180, 187, and 195). Dieldrin is a required analyte for SQO chemical index, but has not been analyzed since Bight '08 because it was rarely detected. As in previous Bight assessments, we assume that non-detectable quantities of dieldrin would not increase chemical exposure category. The methods for determining the compound class sums, handling non-detects, applying correction factors (e.g., for total PCBs for consistency with other groups) and calculating the indices are described in Bay et al. (2021). While other analytes may exert potential toxicity, these are not currently included in the two chemistry indices.

There are two assumptions in evaluating sediment condition based on chemical exposure. First, we only apply the sediment chemistry line of evidence portion of the SQO assessment framework because sediment benthic infaunal data are not yet available. In order to comply with the complete protocol, the remaining two lines of evidence must be applied. This will be discussed in a synthesis report for Bight '23, forthcoming after all assessments for this Bight cycle are complete: toxicity, infauna, algal toxins (domoic acid), trash and microplastics, and extent and magnitude of bioaccumulation.

Our second assumption was that applying the SQO chemistry indices is applicable to sediments on the continental shelf, slope, and basin. The SQO chemistry indices were developed specifically for bays and estuaries of the state, and this is the only habitat in which the full SQO assessment is appropriate. However, no other California-specific sediment chemistry assessment tool currently exists for these offshore habitats. While not intended for the shelf and slope, our assessment in these strata using the SQO chemistry indices would provide some useful understanding of the impacts of chemical contamination there.

Comparisons of SQO Chemical Index Scores in this report are based on a unified dataset created since the Bight '18 report. This unified dataset contains all sediment chemistry data from all Bight cycles (1994-2023). Accordingly, scores as well as resultant data based on scores (e.g., areal extents) for all Bight cycles in this report are calculated according to current protocols (Bay et al. 2021), and may thus differ slightly from those reported in reports for sediment chemistry from those previous cycles in which SQOs were determined i.e., 2008 onwards. For example, our calculations round analyte concentrations to four decimal places if the measured value was under unity, and to two decimal places otherwise. This is a change from the past, in which concentrations were universally rounded to two decimal places. This change was introduced in part because the older calculation method rounded concentrations of mercury that were low but were clearly detections (i.e., $< 0.005 \mu\text{g/g}$) down to zero. Mercury is measured with greater sensitivity than in the past, so this rounding would avoid classifying low but clearly detectable values of mercury, as commonly measured now, as non-detects.

III. QUALITY ASSURANCE/QUALITY CONTROL (QA/QC)

The primary goal of the QA/QC effort was to ensure that sediment chemistry data generated among the many study participating organizations was comparable and complete. Therefore, a performance-based approach to QA/QC was adopted, as used in previous regional surveys (Gossett et al. 2003). This approach, carried out in accordance with the Bight '23 Quality Assurance Manual (Bight '23 Sediment Quality Committee 2023), meets common DQOs for criteria with regards to sensitivity, accuracy, and precision, yet allows for flexibility by each participating laboratory to utilize its own protocols. The QA/QC effort for Bight '23 sediment chemistry is summarized below, with details on the intercalibration exercise in Appendix A, and on QA/QC for analysis of Bight '23 sediment samples in Appendix B.

Interlaboratory Comparison Exercise

Prior to analysis of field samples, reference sediment samples were selected and prepared, and analyzed by all participating labs to assess the inter-laboratory comparability of analytical results. Metals, organic measurements, and some ancillary analytes (e.g., grain size, TOC) were each evaluated using two types of reference materials: a certified reference material with assigned certified or reference values, and reference materials generated from Bight sediment with regionally relevant matrices and ranges of expected target analyte concentrations. Reference materials were analyzed in triplicate. As noted below, PBDEs, pyrethroids, neonicotinoids, tire-wear compounds, and PFAS were measured and assessed only using field reference materials due to the unavailability of certified reference materials for these analyte classes. Intercalibration results for these analytes are for informational purposes only. The full set of participating laboratories in Appendix A, which details the intercalibration exercise and its results, included some that did not analyze field samples; they participated on a volunteer basis.

Reporting Limits

Quality Control (QC) goals are described in detail in the Bight '23 Quality Assurance Manual (Bight '23 Sediment Quality Committee 2023), summarized in Tables 5-7, and discussed in detail in Appendix B. In brief summary, completeness, defined as the proportion of the expected data that was collected in the measurement process, was 98% for core analytes (metals, organochlorine pesticides, PAHs, and PCBs), and was 92% or greater for other analyte classes. Holding times were compliant with QC goals for nearly all analytes, except for mercury, grain size, and PFAS. These holding time exceedances are unlikely to affect data quality, as analyses of mercury were initially done within the holding time requirements albeit not with a Bight QA sample, and neither grain size nor PFAS measurements are likely to be affected by holding time (Appendix B). The success frequency of running method blank and reference material QC

samples was 100%, except for TOC (97%) and PFAS (88%). The accuracy and precision success of the QC samples was typically 70% to 100%, except for trace metals with sample duplicates (46%), for PCBs with matrix spike duplicate precision, and for PFAS with matrix spike parameters. The QC of the dataset was addressed by the Sediment Chemistry Technical Committee, as detailed in Appendix B, and data from the affected sample batches were accordingly accepted. Overall, most of the QC criteria were met. Deviations did not impart additional, meaningful uncertainty in measurements. Consequently, no data was removed, excluded, or rejected from the study and its database. However, deviations from QC criteria were flagged by the analyzing laboratory in the study database for users to make their own decisions regarding data quality (Appendix B).

Table 5: Summary of performance-based sediment quality control (QA) criteria and project success for metals.

Quality control parameter	Data quality objective (DQO)	Success (%)
Completeness	90%	98
Method blank frequency	1/batch	100
Method blank limit	<MDL or <5% of measured concentration in samples	97
Reference material frequency	1/batch	100
Reference material accuracy	Within PT performance acceptance limits of certified values for all 15 analytes	99
Blank spike and duplicate frequency	1/batch	78
Blank spike and duplicate accuracy	Within $\pm 15\%$ true value for at least one blank spike per batch	86
Blank spike duplicate precision	RPD <25% (ICP-AES), <15% RPD for over 10 \times MDL (ICP-MS, AA, GFAA, hydride generation, cold vapor)	99
Sample duplicate frequency	10%	84
Sample duplicate precision	RPD < 25% (<30% AA, GFAA, hydride generation, cold vapor)	46

All QA/QC data for pyrethroids, neonicotinoids, tire-wear compounds, and PFAS are for informational purposes only. All PFAS compounds in all types of blanks (laboratory, equipment, and field) were below the reporting limits of all laboratories, indicating that the procedures to

control contamination from PFAS-bearing materials during sample collection, as described in the Bight '23 Field Manual, were effective.

Table 6: Summary of performance-based sediment quality control (QA) criteria and project success for polycyclic aromatic hydrocarbons (PAHs).

Quality control parameter	Data quality objective (DQO)	Success (%)
Completeness	90%	98
Method blank frequency	1/batch	100
Method blank limit	<10×MDL for all analytes	100
Reference material frequency	1/batch	100
Reference material accuracy	Within ±40% of specified value for >80% analytes	85
Matrix spike and duplicate frequency	1/batch	100
Matrix spike and duplicate accuracy	60-140% recovery of spiked amount for >80% of analytes	92
Matrix spike duplicate precision	RPD <40% for >70% of analytes	97

Table 7: Summary of performance-based sediment quality control (QA) criteria and project success (%) for other classes of organic compounds: organochlorine (OC) pesticides, polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), pyrethroids insecticides, neonicotinoid insecticides (neonics), tire-wear compounds, and per- and polyfluorinated alkyl substances (PFAS). N/A = not applicable.

QC parameter	Data quality objective (DQO)	OC pesticides	PCBs	PBDEs	Pyrethroids	Neonics	Tire-wear	PFAS
Completeness	90%	98	98	97	99	100	99	100
Method blank frequency	1/batch	100	100	100	100	100	100	88
Method blank limit	<10×MDL for all analytes	100	100	100	100	100	100	88
Reference material frequency	1/batch	100	100	N/A	N/A	N/A	N/A	N/A
Reference material accuracy	Within ±40% of the specified value for >70% analytes	81	100	N/A	N/A	N/A	N/A	N/A
Matrix spike and duplicate frequency	1/batch	100	88	100	100	97	97	54
Matrix spike and duplicate accuracy	60-140% recovery of spiked amount for >70% of analytes	85	83	100	100	94	90	59
Matrix spike duplicate precision	RPD <40% for >70% of analytes	87	70	100	99	97	97	54

IV. DESCRIPTIVE RESULTS

Unless otherwise specified, all concentrations are on a dry weight basis. In addition, only core Bight analytes (metals, PCBs, organochlorine insecticides, and PAHs) were measured at all sites. The remaining analytes were analyzed in embayment strata, but offshore only at selected sites.

Bight-wide Results

Area-weighted mean (AWM) concentrations for each analyte as well as summary statistics (95% confidence interval or CI, minimum, 10th percentile, median, 90th percentile, maximum concentration, and percent of sites detected) are summarized in Table 8.

Grain size was very coarse (0.6% fines) to very fine (100% silt and clay), averaging $70 \pm 2.7\%$ fines overall. The TOC measurements varied from non-detect to 6.6% TOC, with an average of $1.6 \pm 0.2\%$, and an average 12.5:1 TOC/TN ratio.

All fifteen metals were detected in all samples. Metal AWM ($\pm 95\%$ CI) concentrations ranged from 0.068 ± 0.012 $\mu\text{g/g}$ for mercury, which is present at trace levels in the SCB, to $22,000 \pm 1,700$ $\mu\text{g/g}$ for iron, a constituent of the mineral component of sediment particles. With regards to organic compounds, every site had detectable levels (i.e., at least one concentration above analysis laboratory's RL for at least one analyte within the analyte group) of total chlordanes, total DDT, total PAH, and total PCBs, while total neonicotinoids, total tire-wear compounds, total PBDEs, and total PFAS were detected in 46%, 51%, 59%, and 61% respectively. For organic analyte classes, AWM ranged from 0.21 ± 0.1 ng/g for chlordanes, to 101 ± 17 ng/g for total PAH.

The AWM concentrations and variances observed in Bight '23 (Table 8) are similar to those observed in the previous cycle, Bight '18 (Du et al. 2020). For example, AWM concentrations for mercury, the metal with the lowest values for AWM, were 0.07 ± 0.01 $\mu\text{g/g}$ in 2018, while those for iron were $22,000 \pm 1600$ $\mu\text{g/g}$. The same is true for organic compounds, although means exhibited greater differences than for metals for many compounds e.g., total chlordanes had the lowest AWM concentrations Bight-wide at 0.21 ± 0.1 ng/g (Table 8) compared to a value of 0.08 ± 0.06 ng/g in Bight '18 (Du et al. 2020). The greatest AWM concentrations for organics was for PAHs at and total PAHs at 100 ± 17 ng/g (Table 8), which compared well with the Bight '18 corresponding values of 100 ± 24 ng/g (Du et al. 2020). Some organic compounds, such as DDT and PCBs, had concentrations quite variable across the entire Bight, resulting in large variances similar in magnitude as those previously observed. For example, PCBs had AWM concentrations of 22 ± 16 ng/g (Table 8), compared to 13 ± 8.8 ng/g in Bight '18 (Du et al. 2020). Bight-wide AWM concentrations of the analytes new to Bight '23 were all low (Table 8).

Subpopulation Comparisons

Strata-specific AWM concentrations and their corresponding 95% CI (Table 9) show three different patterns in general. First, the embayment strata (marinas, ports, bays, and estuaries) had greater concentrations of most metals and organic contaminants compared to shelf and slope strata. For example, copper (Figure 2) ranged offshore from $1.0 \pm 0.8 \mu\text{g/g}$ in the outer shelf to $29 \pm 1.5 \mu\text{g/g}$ on the lower slope, and from $38 \pm 7.1 \mu\text{g/g}$ in estuaries to $140 \pm 17 \mu\text{g/g}$ in marinas. Second, a different trend was present for DDT (Figure 2) in sediments on the continental shelf and slope (maximum of $1400 \pm 1300 \text{ ng/g}$), which had higher concentrations of DDT compared to embayments (maximum of $46 \pm 9.6 \text{ ng/g}$) from historical discharges of DDT wastes through wastewater effluent and through direct dumping in offshore strata (Kivenson et al. 2019). Third, analytes with natural origins, such as aluminum (Figure 2) and iron had enrichments both nearshore and offshore. Also, an enrichment of sediment fines and TOC was found at deeper locations. Fines on the inner shelf (5-30 m) were $31 \pm 4.3\%$ compared to $86 \pm 3.3\%$ on the lower slope (500-1000m). The increase in finer, more organic-rich sediments is consistent with the observation of similar increases in concentrations of many contaminants with depth.

Marinas and ports consistently exhibited the greatest AWM concentrations of trace metals and legacy organic contaminants (Table 9). For example, copper and mercury were greatest in marinas, while total PFAS and total pyrethroids were greatest in ports. Within embayments, ports also had the greatest concentrations of total PAHs and total PCBs. The latter is consistent with contributions from Aroclors 1254 and 1260 from the building and boat construction industries (Phan and Otim 2025). Other metals with a history of anthropogenic inputs (i.e. zinc and lead) followed similar trends. In some cases, contaminants of the lower slope had AWM concentrations at levels similar to those in embayments (Table 9). Sediments from the Channel Islands consistently had the lowest concentrations of most trace metals and organic constituents, except for neonicotinoid insecticides.

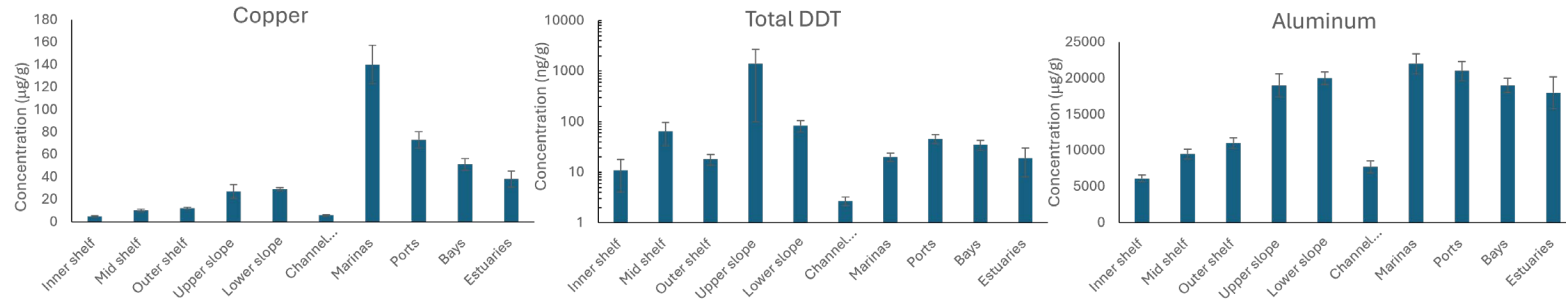


Figure 2: Area-weighted mean concentrations and 95% confidence limits of copper (left), total DDT (center, note log scale), and aluminum (right) by stratum.

Table 8: Bight-wide area-weighted mean (AWM) concentrations and selected summary statistics for sediment contaminants. Metal concentrations are in ug/g dry weight, organic contaminant concentrations are in ng/g dry weight. MDL = method detection limit. TOC = total organic carbon, TN = total nitrogen. N/A = not applicable.

Analyte	AWM	95% CI	Minimum	10 th percentile	Median	90 th percentile	Maximum	% detection
% fines	70	2.7	0.6	18	66	92	100	N/A
% TOC	1.65	0.18	0	0	0.68	2.46	6.59	99
% TN	0.13	0.013	0	0	0.03	0.2	0.78	99
Aluminum	16000	700	930	5500	16000	30000	49000	100
Antimony	0.56	0.086	0	0.04	0.38	1.2	5.0	90
Arsenic	6.1	0.34	0	2.7	6.0	12	36	99
Barium	240	32	2.9	39	105	260	2000	100
Beryllium	0.51	0.033	0	0.08	0.4	0.74	2.0	93
Cadmium	0.99	0.18	0	0.08	0.34	1.7	30	99
Chromium	52	3.7	2.8	13	34	65	350	100
Copper	21	1.6	0.74	4.6	28	150	520	100
Iron	26000	1700	3200	9200	23000	39000	99000	100
Lead	10	1.1	0.77	3.4	12	45	170	100
Mercury	0.068	0.012	0	0.01	0.06	0.46	2.6	99
Nickel	26	1.6	0.8	5.7	16	33	75	100

Analyte	AWM	95% CI	Minimum	10th percentile	Median	90th percentile	Maximum	% detection
Selenium	2.5	0.25	0	0.05	0.47	2.9	9.2	91
Silver	0.26	0.05	0	0	0.18	1.2	8.1	71
Zinc	72	4.6	4.9	25	85	234	680	100
Total chlordanes	0.21	0.1	0	0	0	3.4	59	35
Total DDT	290	240	0	0	4.6	95	40000	86
Total neonics	0.013	0.0042	0	0	0	0	72	4
Total PAH	100	17	0	6.7	160	1400	17000	97
Total PBDE	0.31	0.072	0	0	1.3	3.8	68	47
Total PCBs	22	16	0	0	4.0	49	2700	75
Total PFAS	0.033	0.0081	0	0	0.04	1.1	68	31
Total pyrethroids	0.22	0.089	0	0	1.4	40	1300	44
Total tire wear	0.0059	0.0021	0	0	0	0.84	19	6

Table 9: Area-weighted mean (AWM) concentrations and 95% confidence intervals (Cis) of sediment contaminants in geographic subpopulations (strata) of the Bight. Metal concentrations are in ug/g dry weight, organic contaminant concentrations are in ng/g dry weight. MDL = method detection limit. TOC = total organic carbon. N/A = not applicable (i.e., not measured in this stratum).

Analyte	Inner shelf mean	Inner shelf 95% CI	Mid shelf mean	Mid shelf 95% CI	Outer shelf mean	Outer shelf 95% CI	Upper slope mean	Upper slope 95% CI	Lower slope mean	Lower slope 95% CI	Channel Islands mean	Channel Islands 95% CI	Marinas Mean	Marinas 95% CI	Ports Mean	Ports 95% CI	Bays Mean	Bays 95% CI	Estuaries Mean	Estuaries 95% CI
% fines	31	4.3	54	4.5	58	3.3	82	3.0	86	3.3	35	5.3	71	3.8	68	2.8	63	3.5	43	5.2
% TOC	0.27	0.04	0.77	0.08	0.88	0.10	1.9	0.16	2.4	0.3	0.37	0.12	1.1	0.12	0.86	0.2	0.65	0.09	1.0	0.25
% TN	0.030	0.0031	0.066	0.007	0.080	0.011	0.15	0.012	0.20	0.022	0.0014	0.0014	0.060	0.016	0.041	0.0094	0.053	0.017	0.038	0.017
Aluminum	6100	460	9500	680	11000	730	19000	1600	20000	850	7700	830	22000	1400	21000	1300	19000	970	18000	2200
Antimony	0.33	0.069	0.50	0.088	0.45	0.064	0.46	0.074	0.74	0.18	0.25	0.027	0.67	0.081	0.72	0.15	0.32	0.046	0.57	0.13
Arsenic	4.2	0.38	4.9	0.56	6.3	0.72	8.1	1.2	6.6	0.48	3.4	0.26	9.0	0.54	8.2	0.48	6.9	0.44	5.7	0.75
Barium	66	9.4	160	37	140	28	210	32	360	61	81	19	110	8.1	150	12	110	9.4	78	8.8
Beryllium	0.18	0.025	0.31	0.038	0.52	0.061	0.63	0.076	0.64	0.053	0.23	0.029	0.54	0.039	0.33	0.048	0.30	0.031	0.39	0.054
Cadmium	0.34	0.083	0.48	0.15	0.34	0.025	2.1	0.95	0.95	0.086	0.57	0.091	0.55	0.14	1.2	0.26	1.1	0.21	0.35	0.067
Chromium	16	1.2	29	3.8	35	5.7	67	11	66	5.4	26	2.6	43	3.3	42	2.0	36	2.2	25	2.7
Copper	4.9	0.60	10	1.2	12	1.0	27	6.0	29	1.5	5.8	0.59	140	17	73	7.4	51	5.2	38	7.1
Iron	11000	730	16000	1200	25000	2800	30000	2700	33000	2900	11000	1100	28000	1600	27000	1300	24000	1200	21000	2300
Lead	4.4	0.42	6.8	0.80	7.4	0.89	15	5.1	12	1.1	4.1	0.45	40	5.3	28	2.7	24	1.8	16	2.6
Mercury	0.03	0.015	0.051	0.012	0.034	0.005	0.12	0.061	0.070	0.012	0.012	0.0012	0.41	0.081	0.29	0.030	0.20	0.022	0.037	0.007
Nickel	9.7	1.2	13	1.5	15	1.0	29	2.6	35	2.3	11	1.3	20	1.6	22	1.3	17	1.4	12	1.4
Selenium	0.64	0.15	0.87	0.15	1.1	0.17	1.9	0.27	4.1	0.39	0.41	0.051	1.2	0.23	0.29	0.042	0.31	0.051	0.40	0.072
Silver	0.10	0.093	0.16	0.064	0.092	0.033	0.37	0.21	0.31	0.054	0.088	0.013	0.68	0.15	0.86	0.13	0.81	0.17	0.36	0.058
Zinc	29	2.6	43	3.6	52	3.2	94	20	90	3.5	25	2.6	200	16	150	17	129	7.2	110	19
Total chlordanes	0.11	0.05	0.24	0.10	0.26	0.25	0.55	0.54	0.08	0.05	0	0	3.6	0.7	3.9	2.3	2.0	0.8	5.1	2.1
Total DDT	11	7.0	65	32	18	4.2	1400	1300	83	21	2.7	0.52	20	3.9	46	9.6	35	7.8	19	11
Total neonics	0	0	0	0	0	0	0	0	0	0	0	0	0.84	0.75	4.1	1.4	0.75	0.52	2.6	2.2
Total PAH	68	23	74	15	72	20	92	33	125	34	22	6.6	830	160	2100	640	520	64	250	65

Analyte	Inner shelf mean	Inner shelf 95% CI	Mid shelf mean	Mid shelf 95% CI	Outer shelf mean	Outer shelf 95% CI	Upper slope mean	Upper slope 95% CI	Lower slope mean	Lower slope 95% CI	Channel Islands mean	Channel Islands 95% CI	Marinas Mean	Marinas 95% CI	Ports Mean	Ports 95% CI	Bays Mean	Bays 95% CI	Estuaries Mean	Estuaries 95% CI
Total PBDE	N/A	N/A	N/A	N/A	0.020	0.020	0.55	0.21	0.11	0.10	1.2	0.20	1.3	0.32	3.4	1.2	1.7	0.30	4.2	2.0
Total PCBs	2.1	0.91	8.3	4.4	2.6	1.0	95	89	7.9	2.5	0.010	0.01	31	7.1	43	6.8	32	7.0	5.6	1.9
Total PFAS	0.097	0.051	0.010	0.005	0.019	0.006	0.027	0.013	N/A	N/A	N/A	N/A	0.76	0.29	7.3	3.2	0.46	0.14	0.87	0.52
Total pyrethroids	N/A	N/A	N/A	N/A	0	0	0	0	0	0	0.017	0.017	27	7.3	61	47	11	4.6	45	21
Total tire-wear	0	0	0.011	0.011	0.036	0.036	0	0	N/A	N/A	N/A	N/A	0.74	0.26	0	0	0.26	0.19	1.9	0.7

Geographic Distribution of Sediment Parameters

The spatial distribution of chemical concentrations (Figures 3-5, and Figures 24-49 in Appendix C) illustrates differing sources and likely fates within the SCB. As with AWM concentrations, there were generally three different patterns for chemical geographic distributions observed: higher levels inshore from anthropogenic activity as exemplified by copper (Figures 3, 31), increased levels offshore specifically from anthropogenic inputs as exemplified by total DDT (Figures 4, 40), and increased concentrations at some locations offshore from probable natural contributions as exemplified by aluminum (Figures 5, 24).

For the first pattern, it is clear that most chemicals had sediment concentrations that were greater in embayments than offshore, although for different reasons. The increased concentrations of copper in embayments are likely due to its use in anti-fouling paints on recreational and commercial vessels (Phan and Otim 2025), while those for PAHs and PBDEs are more likely due to land-based runoff (Du et al. 2020). Embayments generally had greater concentrations of most insecticide classes observed, as well as for PFAS and for tire-wear compounds. Indeed, non-detected levels were generally observed offshore for legacy chlordanes, as well as for current-use pyrethroids, neonicotinoids, and tire-wear compounds, bearing in mind limited analysis for these analytes at offshore sites. The latter two classes of compounds were detected only at scattered locales within embayments.

The second pattern is that of DDT, for which concentrations were greatest near Palos Verdes and Los Angeles Harbor (Figures 4, 40), due to historical discharges from the LACSD ocean outfall (Kivenson et al. 2019, USEPA 2024). These concentrations declined moving northward through Santa Monica Bay and the Santa Barbara Channel, in the net direction of ocean currents. Concentrations were also high further offshore Palos Verdes, between the shore and Catalina Island (Otim 2020), consistent with past ocean dumping of DDT and other wastes (Kivenson et al. 2019).

For the third pattern, metals generally had greater concentrations in embayments, as exemplified above for copper (Figures 3, 31). Elevated levels offshore were observed for some metals, such as aluminum, arsenic, barium, beryllium, iron, nickel, and selenium. These increased concentrations, some of which are at sites not in proximity to known or suspected contamination sources, may be from natural background distributions (Schiff and Gossett 1998, Noblet et al. 2003). For example, iron levels and distributions (Figure 32) have been previously used to determine background levels of metals in southern California coastal surficial sediments (Schiff and Weisberg, 1999, Noblet et al. 2003).

In general, irrespective of contamination pattern, the concentration distributions of chemicals in surficial sediments of Bight '23 were similar to those observed in previous Bight cycles, such

as Bight '13 and Bight '18. This can be illustrated by copper (Figure 6) and DDT (Figures 7), two sets of analytes common across these cycles, and taking into mind that different sites within each strata were sampled in each cycle given the probabilistic study design of the Bight program. The concentration ranges for each bin in these maps (Figures 6-7), taken by determining percentiles from the combined concentration data of all three Bight cycles, were similar to those observed for Bight '23. Accordingly, these observations all suggest that processes that can affect surficial sediment concentrations of analytes occur slowly, over decadal time scales at most. These processes include substantial inputs (e.g., from wastewater or stormwater), exports (e.g., remobilization further offshore), and diagenesis (e.g., resuspension, burial, and transformation).

Episodic events also have potential to influence sediment concentrations. For example, an oil spill from a pipeline leak was discovered offshore Huntington Beach in October 2021. Oil residues from this spill impacted the shorelines of Huntington Beach and Newport Beach. However, there were no corresponding elevated levels of PAHs offshore (Figure 42), and the higher concentrations of PAHs in Newport Bay were found within the protected areas of the bay, consistent with PAH distributions in Bight '18 (Du et al. 2020). There is no evidence that this oil spill contaminated surficial sediments, at least at the Bight sites. The potential impacts of a second episodic event, Hurricane Hilary, will be discussed later in this report.

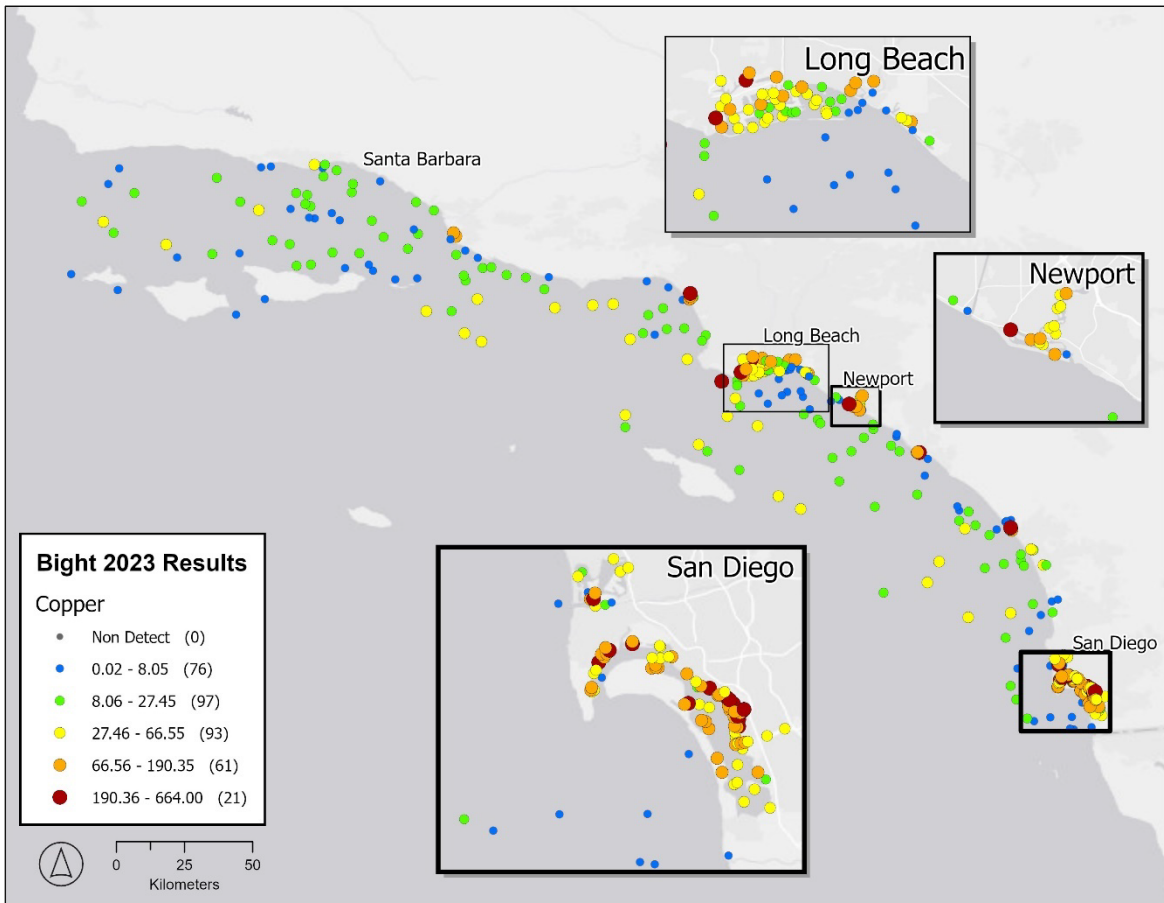


Figure 3: Map of copper concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments, categorized as non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum. Numbers in parentheses after the concentration ranges are the number of samples in each bin.

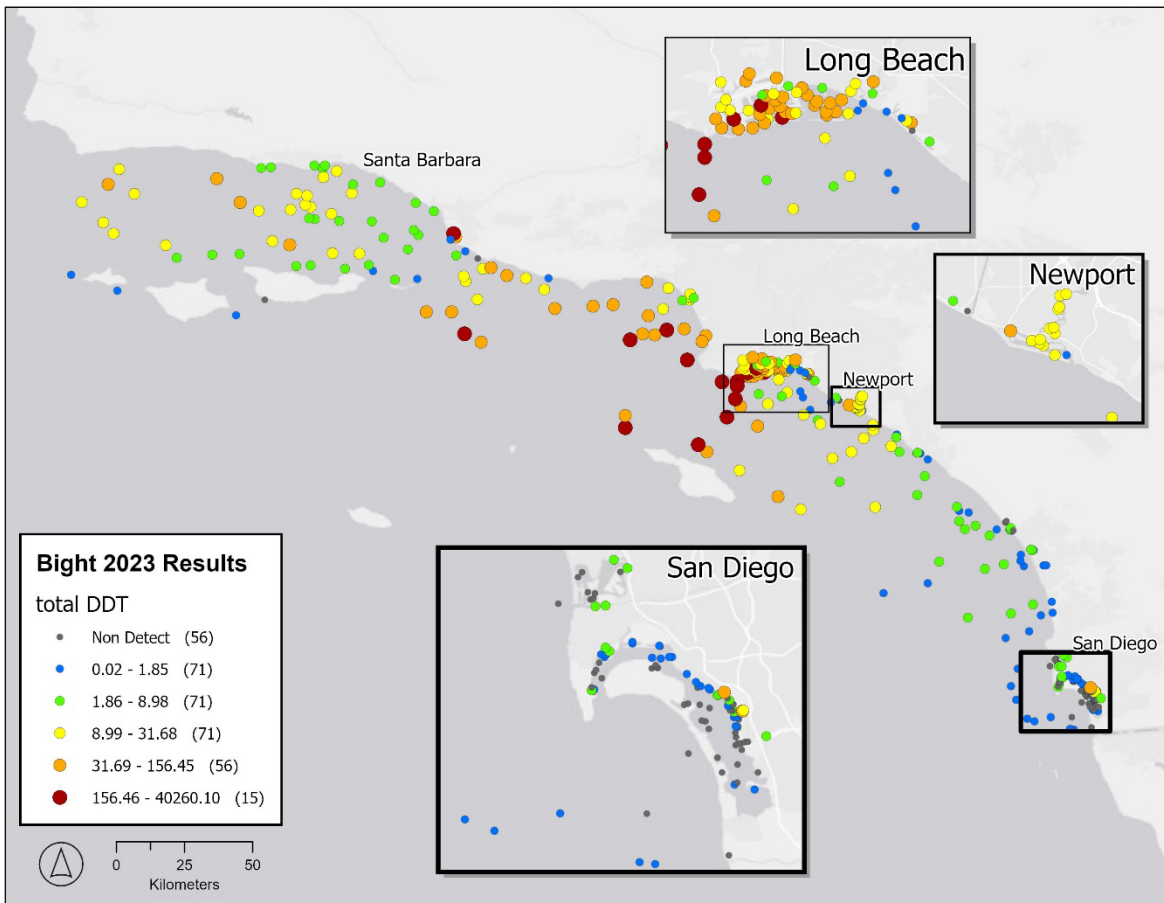


Figure 4: Map of total DDT concentrations in ng/g dry weight in Bight '23 sediments, categorized as non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum. Numbers in parentheses after the concentration ranges are the number of samples in each bin. Non-detected values are treated as a concentration of zero in summation of DDT analytes to calculate total DDT.

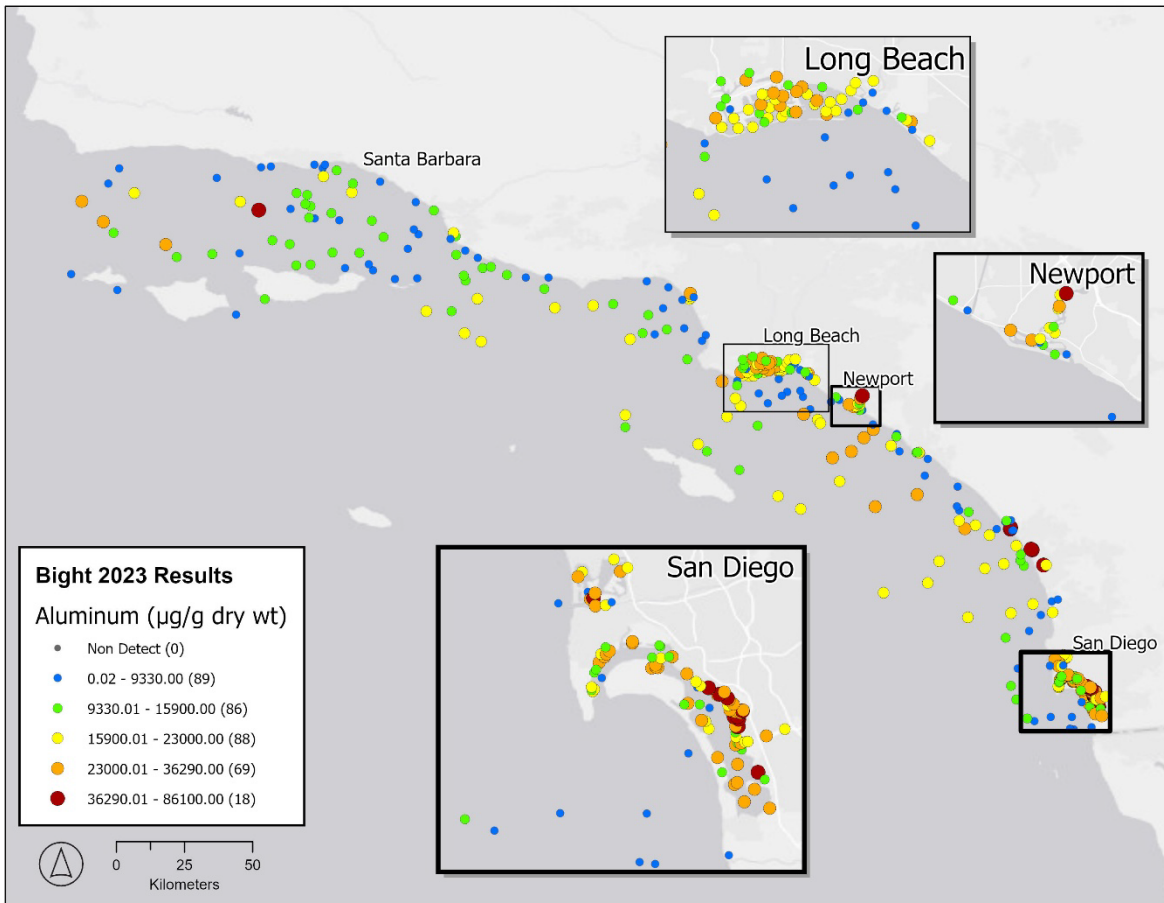


Figure 5: Map of aluminum concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments, categorized as non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum. Numbers in parentheses after the concentration ranges are the number of samples in each bin.

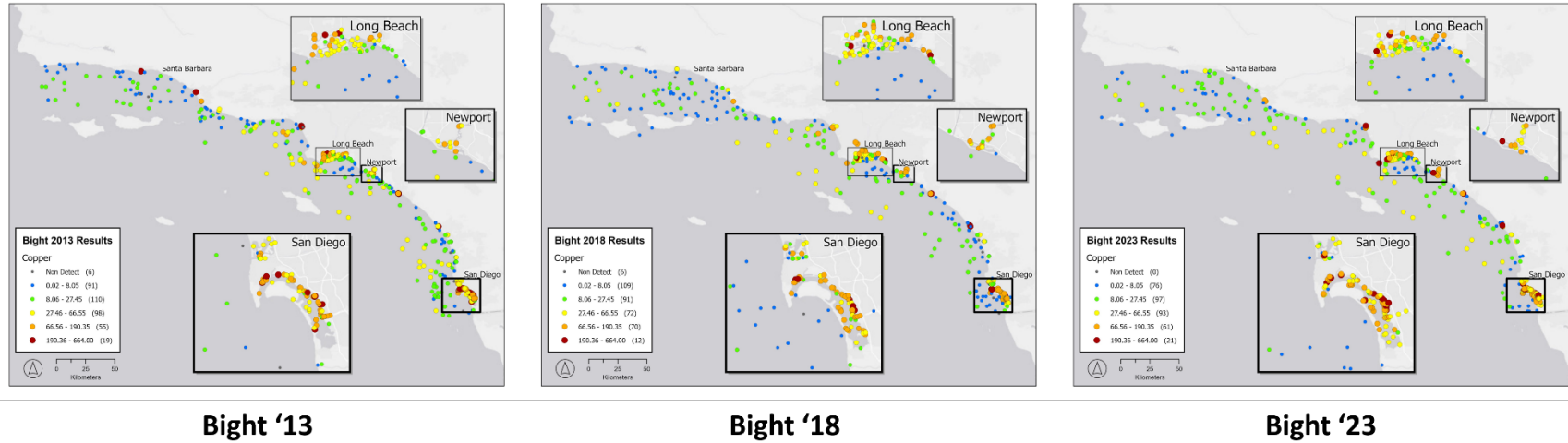
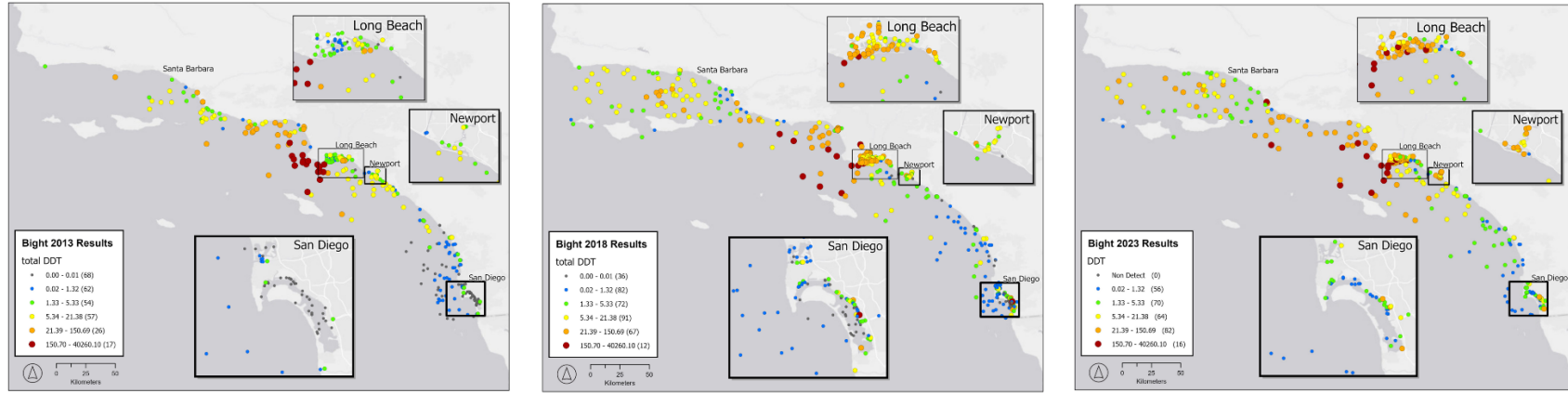


Figure 6: Map of copper concentrations in $\mu\text{g/g}$ dry weight in sediments of Bight '13 (left), Bight '18 (center), and Bight '23 (right), categorized as non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum. Concentration categorization bins use combined data from all three Bight cycles. Numbers in parentheses after the concentration ranges are the number of samples in each bin.



Bight '13

Bight '18

Bight '23

Figure 7: Map of total DDT concentrations in ng/g dry weight in sediments of Bight '13 (left), Bight '18 (center), and Bight '23 (right), categorized as non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum. Concentration categorization bins use combined data from all three Bight cycles. Numbers in parentheses after the concentration ranges are the number of samples in each bin. Non-detected values are treated as a concentration of zero in the summation of DDT analytes to calculate total DDT.

V. ASSESSMENT RESULTS

There are four categories for SQO Chemical Index Scores, derived from the State's sediment quality objectives framework (Bay and Weisberg 2008; SWRCB 2018), and are as follows: Category 1 is minimal exposure, Category 2 is low exposure, Category 3 is moderate exposure, and Category 4 is high exposure. All error estimates represent 95% confidence intervals unless otherwise indicated, with error estimates propagated from those of individual categories. Sediments in categories 1 and 2 meet SQO requirements (Bay et al. 2021) for chemical exposure evaluation.

All assessments must take into account four important aspects of evaluation. First, this assessment is based on composite scoring indices that aggregate many chemicals, so individual chemical assessments are unknown. Related to this point is the fact that this assessment does not take into account those chemicals that are not part of the SQO framework that may also exhibit aquatic toxicity (e.g., PBDEs, pyrethroids, PFAS, tire-wear compounds, neonicotinoid insecticides), nor does it account for chemicals that were not measured but may be present in sediments. Second, the SQO site assessment also relies on biological and toxicological lines of evidence, which are the focus of separate reports for Bight '23. Third, the SQO framework only applies to marine embayments; that having been said, we use the tool here for offshore strata for comparison purposes. Finally, SQO assessments are most meaningful when performed with enough sites to provide statistical confidence, i.e., Bight-wide, or within individual strata, which was done here except as noted.

Spatial Extent of Chemical Index Scores

Approximately $83 \pm 9.3\%$ of SCB sediments met SQO requirements (Figure 8). Of the remainder, approximately $16 \pm 5.6\%$ of SCB sediments were in the moderate exposure category, with the remaining $0.65 \pm 1.0\%$ of the SCB in the high chemistry exposure category.

Among strata, the areal extent of conditions meeting SQO requirements varied from 32% to 100% (Figure 9). Lower proportions of SQO-compliant conditions were observed in most embayment strata (bays at $53 \pm 12\%$, marinas at $32 \pm 13\%$, and ports at $40 \pm 15\%$) compared to offshore strata ($60 \pm 17\%$ for upper slope, which was also the only non-embayment stratum to have chemical exposure in the high category, to 100% for Channel Islands). Of the embayment strata, estuaries had the greatest extent of sediments meeting SQO requirements, at $79 \pm 20\%$.

The spatial distribution of exposure categories (Figure 10) shows that the majority of sites with chemical exposure conditions that did not meet SQO requirements were those in embayments, such as sites in Marina del Rey, Los Angeles/Long Beach Harbor, Newport Bay, and San Diego Bay. Approximately $47.0 \pm 8.4\%$ of the combined area of all embayments (bays, estuaries, ports, marinas) had sediment exposure scores meeting SQO guidelines. There were a number of sites

in the moderate exposure category offshore, generally in Santa Monica Bay and further offshore, between Santa Barbara and the Channel Islands and to their west, and offshore of Palos Verdes. The slope strata had $76.5\pm 13\%$ of its area meeting SQO guidelines, while the corresponding value for shelf strata as $93\pm 10\%$.

As previously mentioned, episodic events may impact sediment contaminant concentrations, as well as sediment quality. The second such event that occurred since the previous Bight survey (Bight '18) was the Category 4 Hurricane Hilary (August 16-21, 2023), the remnants of which made landfall off southern California as a tropical storm. Clearly, this storm had significant potential to resuspend and redistribute sediments and their contaminant loads as well as pollutant loads from coastal watersheds, and thus affect organisms that could be exposed. However, there was no obvious pattern in SQO Integrated Exposure scores for sediment samples taken before and after the storm (Figure 11), including in Mission Bay and San Diego Bay, the parts of the SCB most heavily hit by the storm. This observation suggests that overall, Hilary had no significant effect on sediment chemical contaminant distributions regionally, as exemplified by SQO exposure scores. However, it must be noted that there were very few samples immediately before or after the storm, and particularly in San Diego, to evaluate specifically the effect of the storm. The Bight program is not currently designed to evaluate sudden and unexpected episodic events.

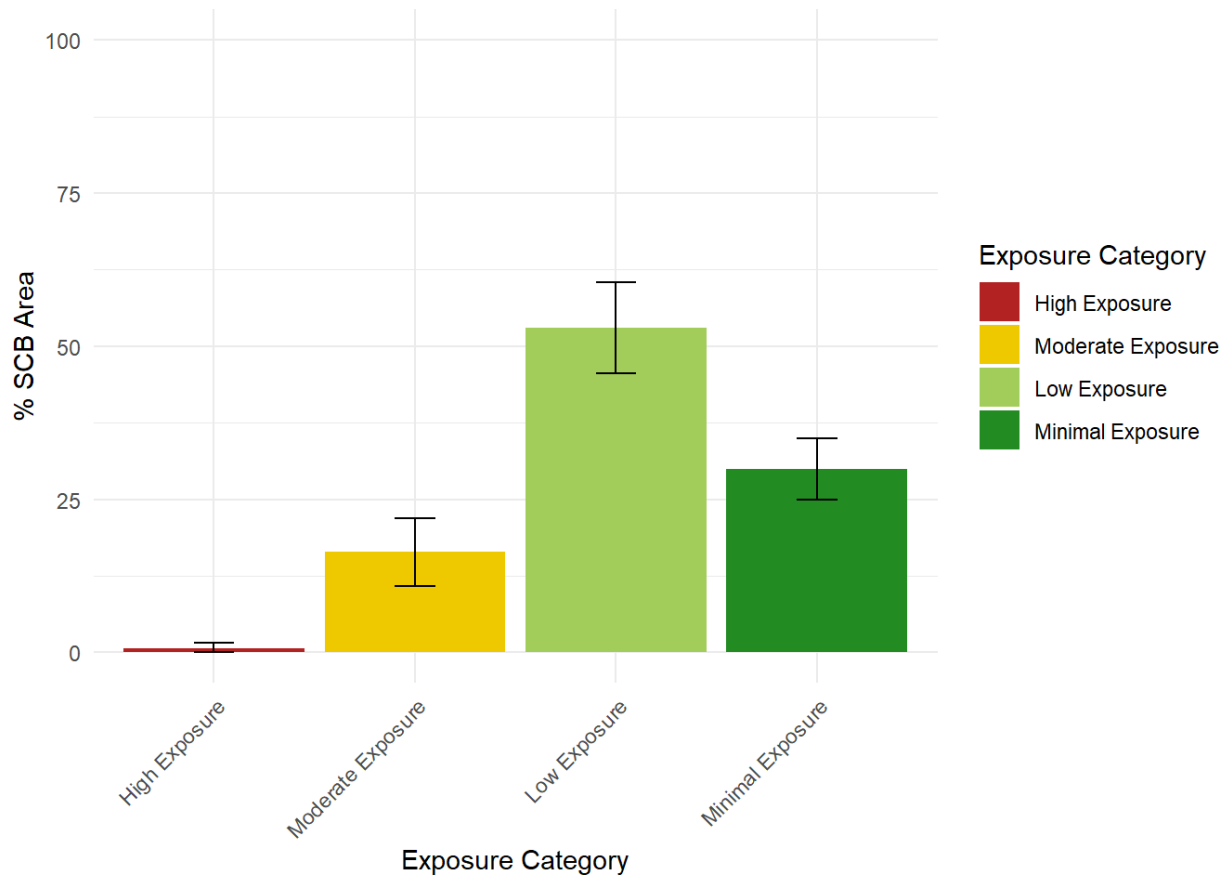


Figure 8: Areal extent of chemistry exposure Sediment Quality Objectives (SQOs) across the Southern California Bight. Minimal and low exposure are in compliance for the SQO sediment chemistry line of evidence.

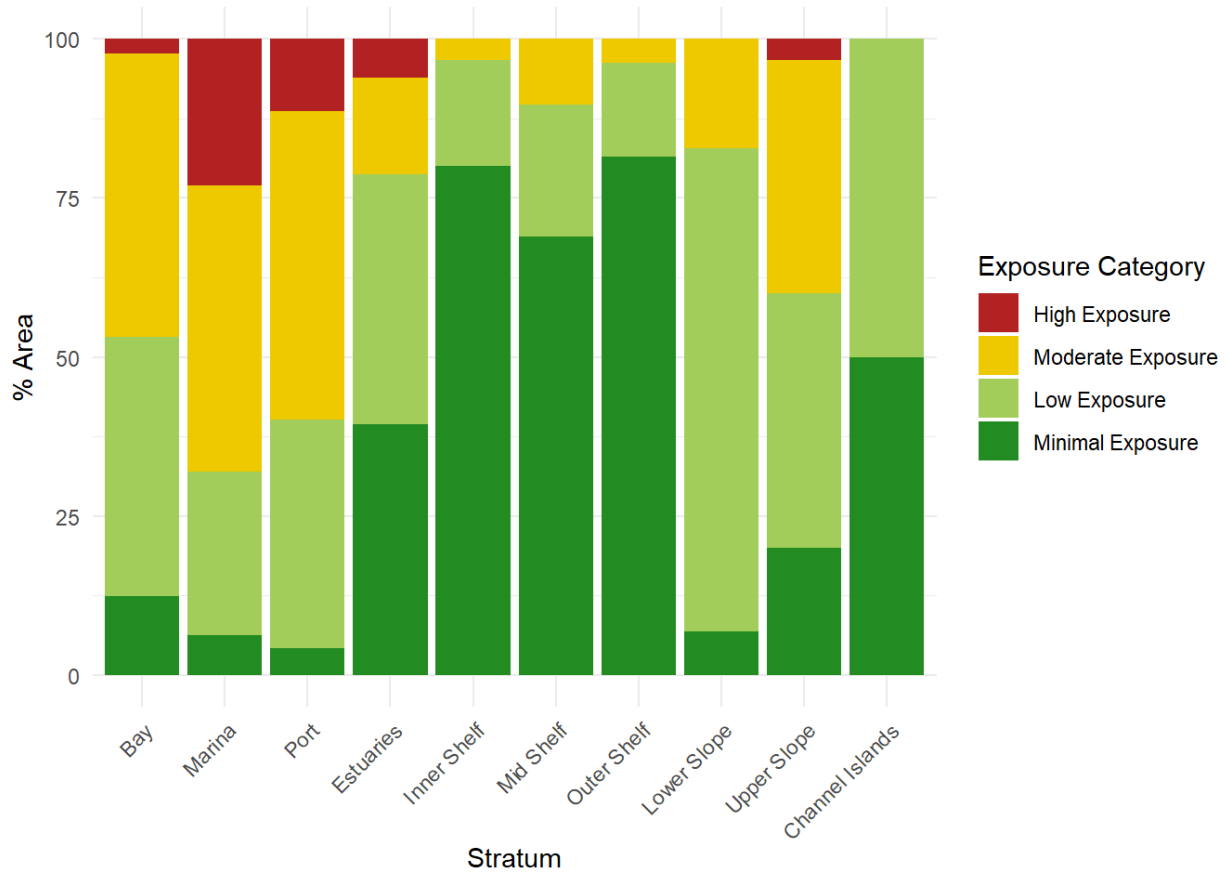


Figure 9: Areal extent of chemistry exposure Sediment Quality Objectives (SQOs) across Bight by stratum.

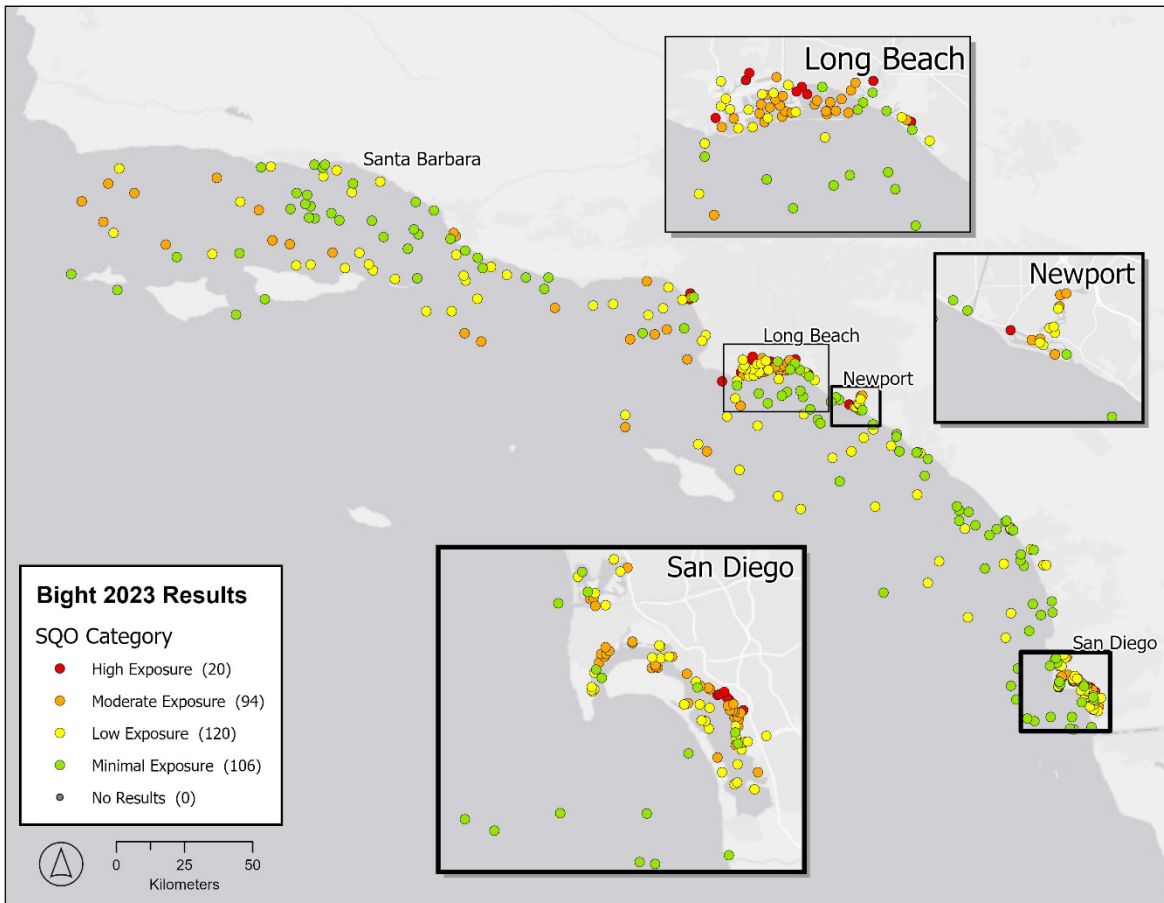


Figure 10: Spatial distribution of categories for Sediment Quality Objectives (SQOs). Number of samples in each category is in parentheses in the legend.

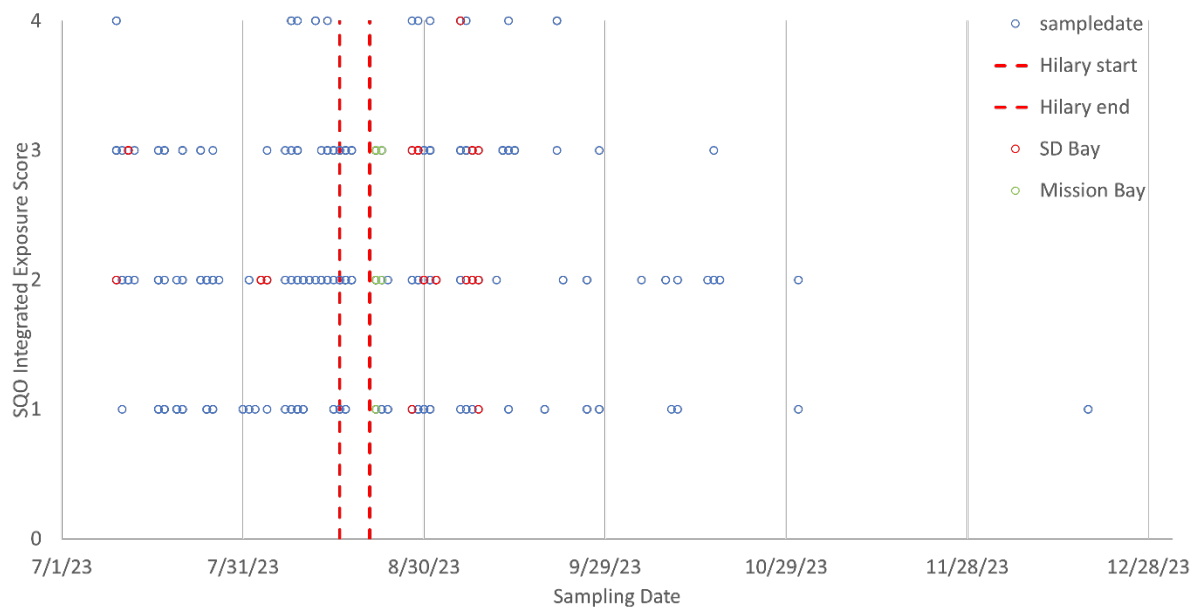


Figure 11: Sediment Quality Objective (SQO) Integrated Exposure scores for sediment samples collected before and after Hurricane Hilary.

Temporal Extent of Chemical Index Scores

There are no significant changes Bight-wide in sediment quality, based on chemistry SQO, from 1998 to 2023. The areal extent of condition meeting SQO guidelines across the entire SCB based on SQO Chemical Index Scores changed from $96 \pm 5.3\%$ in 1998 to $83 \pm 9.3\%$ in 2023 (Figure 12). There is an apparent overall gradual increase in the areal extent of conditions meeting SQO guidelines since 2003 ($66 \pm 6.4\%$), with 2008 ($77 \pm 6.6\%$) being greater than those five years previous, 2013 being lower ($68 \pm 11\%$) than in 2008, and 2018 ($79 \pm 8.7\%$) being greater than in 2013. That said, the increase in areal extent was not statistically significant assuming $\alpha = 0.05$, as indicated with a linear regression (Figure 13) across the survey years (2003 to 2023) against the overall areal percentage of the overall SCB meeting SQO guidelines ($r^2 = 0.60$, slope $p = 0.13$). The areal extent of high sediment exposure was small, irrespective of survey year, with a maximum of $2.2 \pm 2.6\%$ of the overall SCB area in 2008 and a minimum of $0.07 \pm 0.07\%$ in 2003. Between the 1998 and 2023 surveys, the range of reporting levels was similar (i.e., within the same order of magnitude), indicating that changes in reporting levels were not a factor in any changes in the areal extent of SQO-compliant conditions.

The apparent drop in Bight-wide sediment quality meeting SQO guidelines between 1998 and subsequent cycles (Figure 12) is an artifact of sampling during Bight '98, as strata for the outer shelf and the slopes were not yet included at that time. Offshore sites, which generally have lower chemical scores, also typically have greater area weights than those inshore. As a result, a greater proportion of the area in Bight '98 fell under sediment conditions meeting SQO

guidelines (Figure 14). For a more even comparison, the areal extent for SQOs was tallied across all six Bight cycles (1998-2023) using only data from those Bight '98 strata also used in subsequent cycles: ports, marinas, bays, and the inner- and mid-shelf strata. The canyon stratum, unique to Bight '98, was excluded, as were Bight '98 sites associated with the Channel Islands for which only grain size data is available. With this more direct comparison, all Bight cycles from 1998 to now have comparable areal extent of sediment quality meeting SQO guidelines of 80-98%, which is also not significant statistically ($r^2 = 0.27$, slope $p = 0.29$).

Sediment conditions meeting SQO guidelines in embayment strata were, at best, unchanged (Figure 15). Within the combined port, marina, and bay strata, conditions that met SQO guidelines changed from $52 \pm 15\%$ in 2003 to $75 \pm 8.5\%$ in 2013, then were at 2003 levels (e.g., $47 \pm 8.4\%$ in 2023). This change was not significant, based on regressions of year against percent SQO-compliant area of these combined strata (Figure 16), irrespective of whether 1998 was included ($r^2 = 0.25$, slope $p = 0.31$) or not ($r^2 = 0.04$, slope $p = 0.76$). The 2013 assessment was driven by greater areal extent of chemical exposure meeting SQO guidelines in bays and in ports, and not in marinas (Figure 15). As previously noted (Du et al. 2020), the 2013 survey results may be anomalous relative to the other survey years. With two exceptions, there were generally no apparent temporal patterns in the areal extent of chemical exposure conditions meeting SQO guidelines for individual embayment strata (Figure 15), including estuaries, nor were there any temporal patterns observed for areal extent of high chemical exposure there. Those two exceptions are as follows. First, ports had only $0.52 \pm 0.44\%$ of their area within the high chemical exposure category in 2008, but subsequent surveys found increasing areal extent through 2018 ($10 \pm 10\%$) and 2023 ($11 \pm 9.2\%$). Second, the areal extent of sediment chemical exposure conditions meeting SQO guidelines in estuaries increased from 2003 ($61 \pm 17\%$) to 2023 ($79 \pm 20\%$). However, neither exception was statistically significant, based on regression of years (2003-2023) against the relevant areas (Figure 16, $r^2 = 0.11$, slope $p = 0.59$; and $r^2 = 0.02$, slope $p = 0.81$, respectively), as may be expected by the many factors that may skew data collected over a long period of time (Otim 2025).

Offshore, the percentage of conditions meeting SQO guidelines within the shelf composite stratum has been similar since 2003 (Figure 15), but the area extent of minimal exposure increased with each cycle since then (Figure 17) albeit not significantly ($r^2 = 0.12$, slope $p = 0.56$), from $57 \pm 7.3\%$ in 2003 to $75 \pm 7.2\%$ in 2023, aside from a decrease in 2013 ($53 \pm 7.1\%$) compared to 2008 ($77 \pm 6.8\%$). This increase was driven by conditions in the outer shelf, which changed albeit also not significantly ($r^2 = 0.53$, slope $p = 0.16$) from $39 \pm 14\%$ in 2003 to $81 \pm 12\%$ in 2023 (Figure 17). For the slope composite strata (Figure 15), chemistry exposure conditions meeting SQO guidelines have significantly increased from $53 \pm 8.6\%$ in 2003 to $77 \pm 13\%$ in the current cycle (Figure 16, $r^2 = 0.90$, slope $p = 0.01$). This increase was due to statistically

significant increases in the lower slope, from $55 \pm 8.8\%$ in 2003 to $83 \pm 16\%$ in 2023 ($r^2 = 0.95$, slope $p = 0.004$, Figure 17). The areal extent of sediment condition meeting SQO guidelines in the Channel Island National Marine Sanctuary increased, but not significantly ($r^2 = 0.75$, slope $p = 0.13$), from $87 \pm 16\%$ in 2003 to $100 \pm 28\%$ in both 2018 and 2023 (Figure 15).

There is a total of 157 sites in Bight '23 (Table 2) revisited from previous Bight surveys, for which data is available for SQO calculations across both this survey and previous ones. Of these revisit sites, only a single site (B23-12136 in Los Angeles/Long Beach Harbor) had chemical exposure conditions that no longer met SQO guidelines in 2023, but did in both 2018 and in 2003, the first Bight cycle in which common strata were studied from that point forward. This site also had chemical exposure conditions meeting SQO guidelines in 2008 and 2013. As noted previously, focusing assessment on revisit site results does not result in an equal spatial resolution of the strata in which the revisit sites are located (Du et al. 2020), and thus comparisons would have poor statistical confidence. Bight is designed to capture regionwide trends, not trends at specific sites or in specific waterbodies for which evaluations are best done with site- or waterbody-specific monitoring programs that do exist. Moreover, chemical exposure is only one of three lines of evidence for SQO site assessment, along with biological and toxicological lines.

It must also be noted, over the decades that the Bight program has existed, there have been considerable increases in population and anthropogenic activity (such as shipping in ports), and thus environmental stressors including chemical contamination. The fact that overall sediment quality has maintained a consistent level during this period suggests that environmental laws, management efforts, and consequent actions (e.g., best management practices or BMPs) are providing protection from keeping sediment quality from getting progressively worse over time. Indeed, as noted above, there are some modest improvements in sediment quality in specific strata, which may reflect such management actions and efforts.

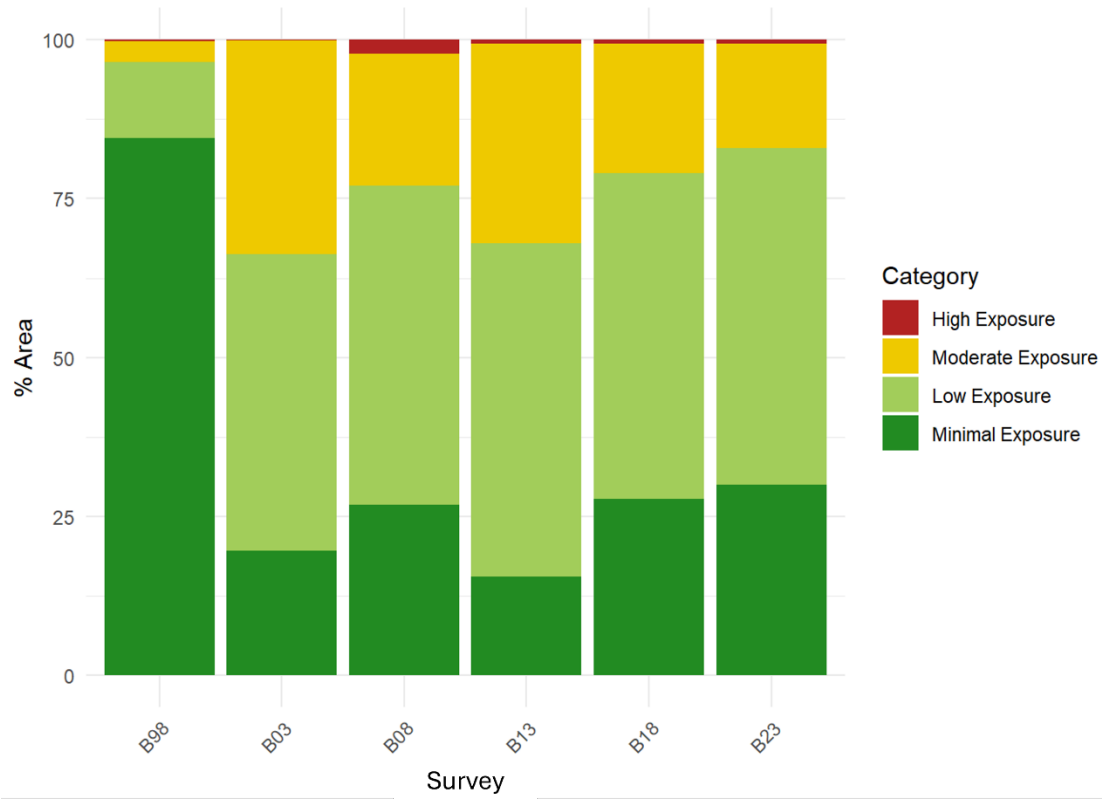


Figure 12: Bight-wide areal extent of sediment chemical contamination exposure by survey year.

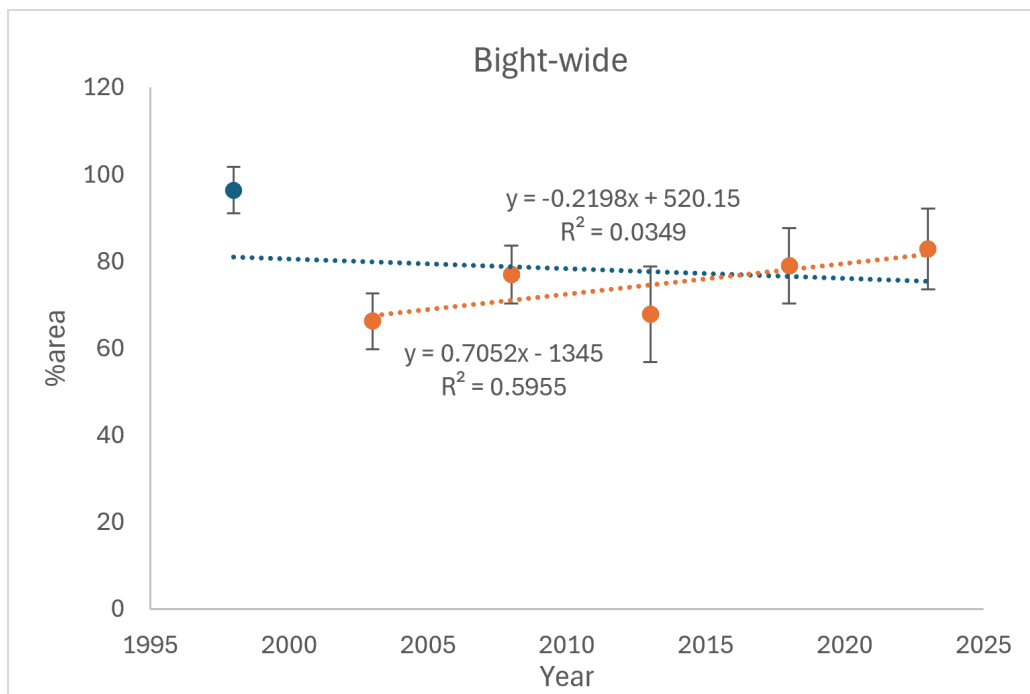


Figure 13: Temporal trends in Bight-wide area meeting sediment chemistry objective (SQO) guidelines (minimal + low exposure) from 1998-2023, and from 2003-2023.

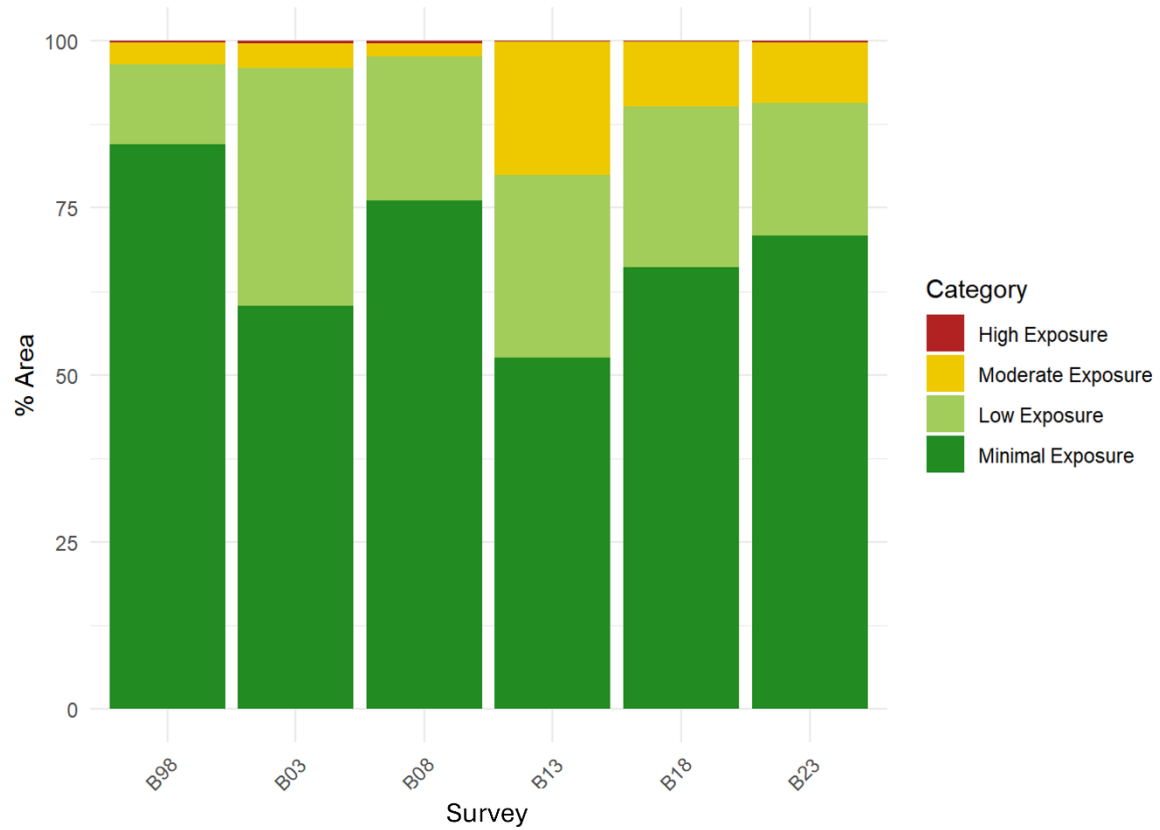


Figure 14: Bight-wide areal extent of sediment chemical contamination exposure by survey year, including only those strata used in Bight '98 (ports, marinas, bays, inner shelf, and mid-shelf).

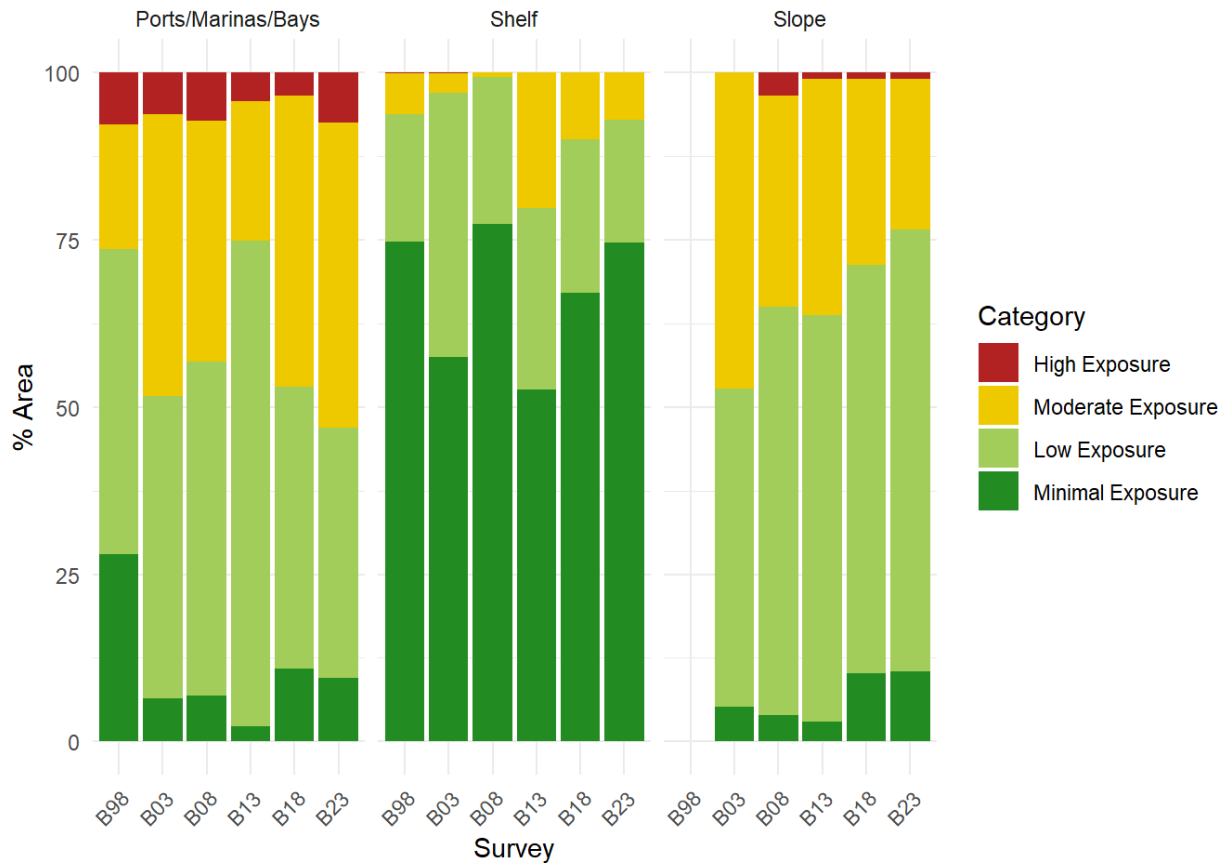


Figure 15: Areal extent of sediment chemical contamination exposure by composite stratum (ports/marinas/bay, inner/mid/upper shelf, and upper/lower slope) and survey year.

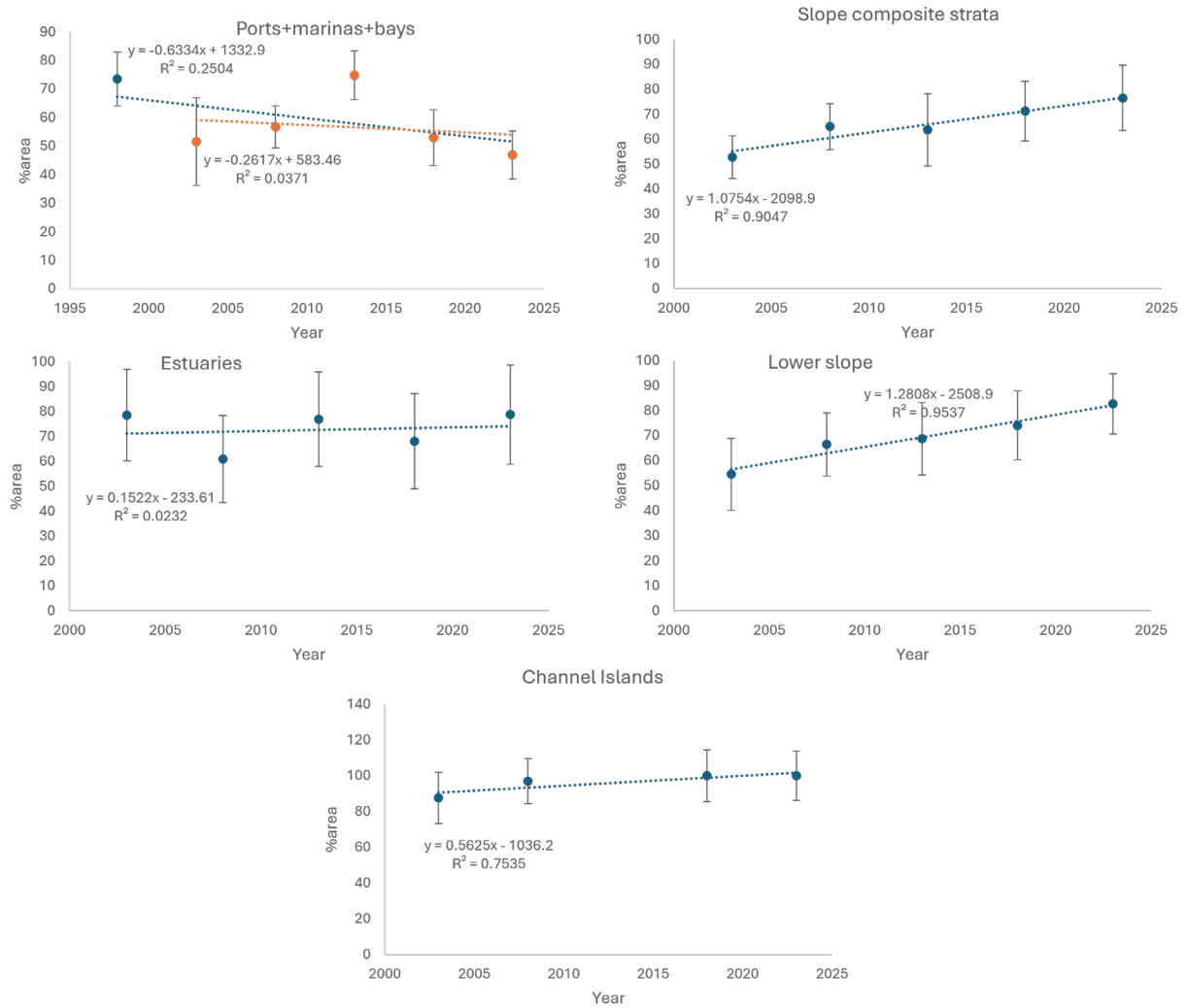


Figure 16: Temporal trends in selected strata meeting sediment chemistry objective (SQO) guidelines (minimal + low exposure) from 1998-2023 (ports+marinas+bays), and from 2003-2023 (ports+marinas+bays, estuaries, upper and lower slope, lower slope alone, and Channel Islands).

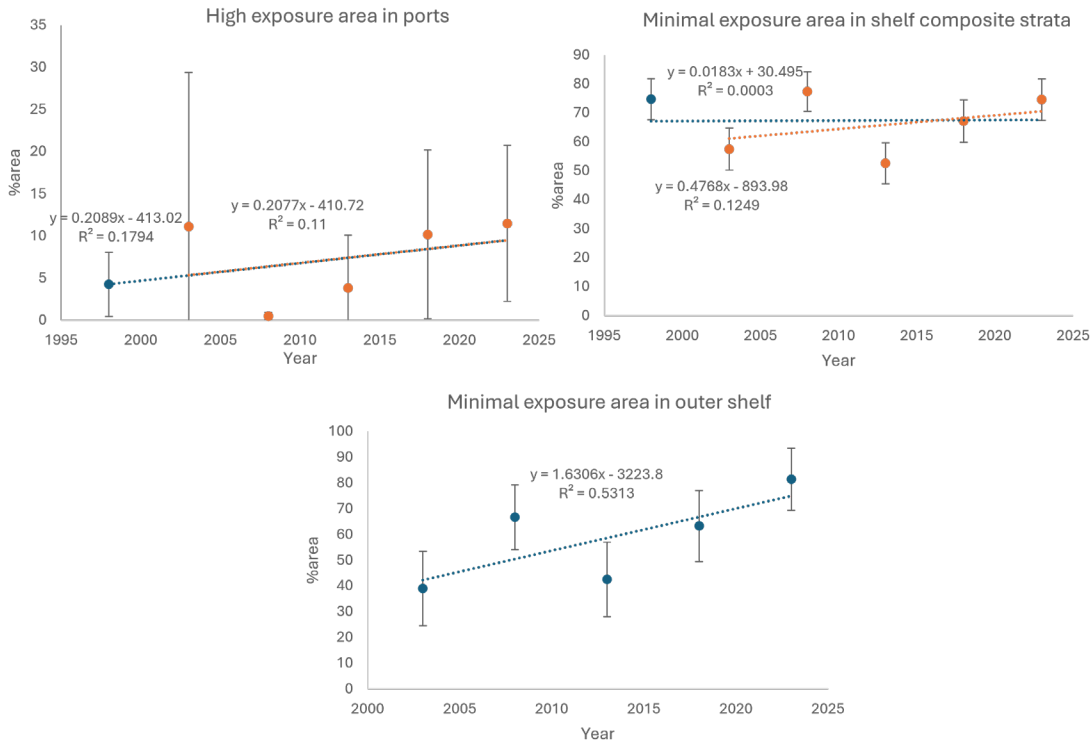


Figure 17: Temporal trends in selected strata for selected specific sediment chemistry objective (SQO) scores: high exposure in ports from 1998-2023 and 2003-2023, minimal exposure in shelf composite strata from 1998-2023 and 2003-2023, and minimal exposure area in outer shelf from 2003-2023.

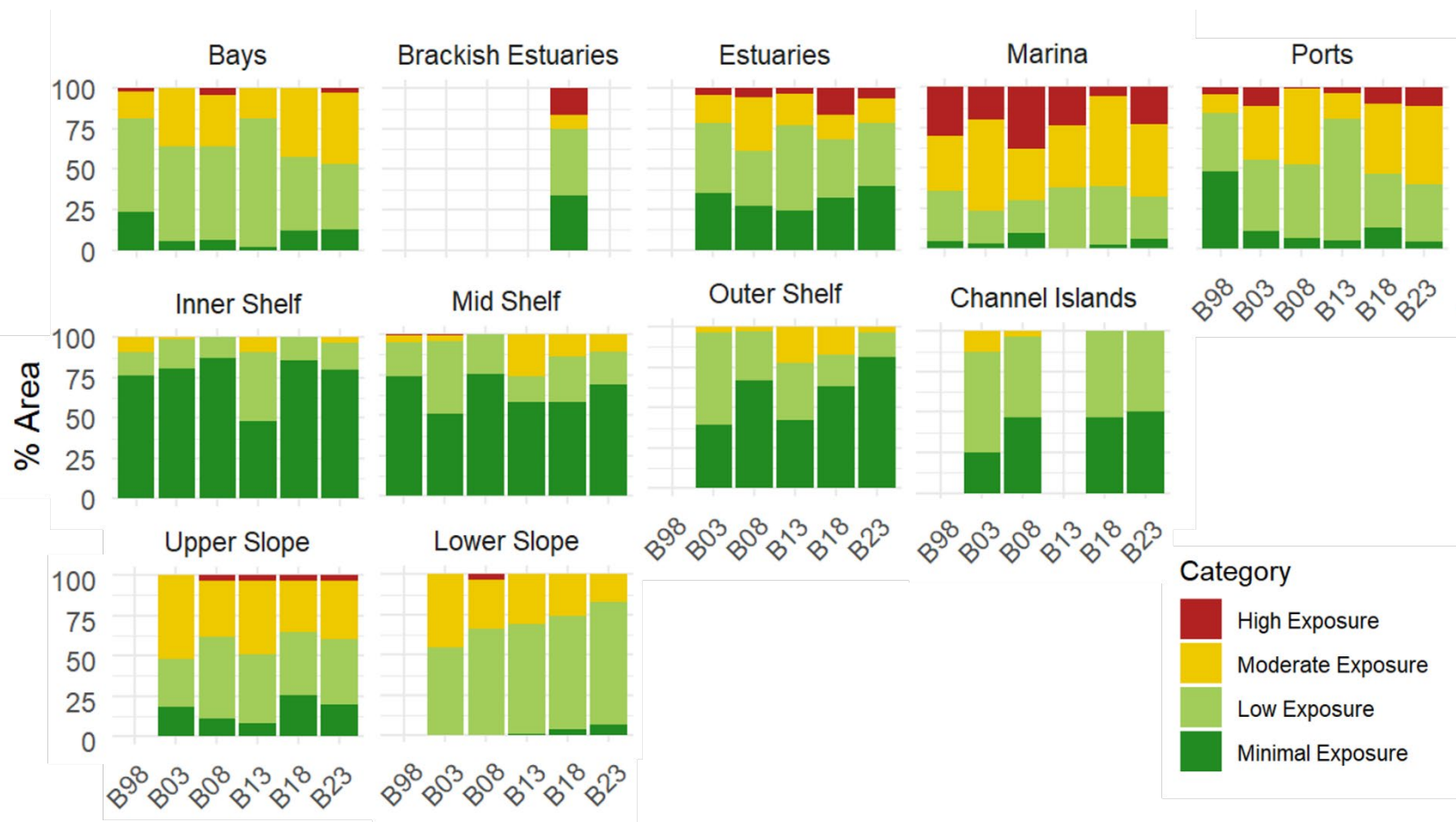


Figure 18: Areal extent of sediment chemical contamination exposure by individual stratum and survey year, arranged by embayment strata (top) closest to shore, shelf and Channel Islands strata (middle), and slope strata (bottom) farthest from shore.

VI. DISCUSSION

Bight '23 sampled similar habitats as was done in the previous five Bight cycles (1998-2018). Similar results were obtained with regards to the extent and magnitude of sediment contamination, spatial distribution, and temporal trends. In this section, we will discuss some noteworthy aspects of the current cycle, including a comparison of chemistry and toxicity SQO scores, new target analytes, and issues with reporting limits.

Comparison of Chemistry and Toxicity SQO Scores

The spatial distributions of chemical exposure and toxicity SQO scores (Mehinto et al. 2025) are similar (Figure 19), indicating that benthic organisms have greater stressors at embayments than they do further offshore. However, it does not necessarily follow that those stressors are related to chemical contamination in the sediments. The number of sites meeting and not meeting SQO guidelines for both chemistry and toxicity were tabulated (Figure 20), to explore what similarities may exist between assessments by the two lines of evidence, particularly for amphipods, which were measured in both embayments and offshore.

Many sites (41-78%) had both sediment toxicity and chemical exposure scores meeting SQO guidelines, irrespective of which metric was used (Figure 19). These sites would, for the toxicity SQO, be classified as either non-toxic or low toxicity, and for the chemistry SQO have either minimal or low exposure. A small minority (3.2-11%) of sites had non-compliant scores for both toxicity (moderate or high toxicity) and chemical exposure (moderate to high exposures). The reason for this observation is that only a small number of sites had non-compliant toxicity assessments (e.g., 14% moderate or high amphipod toxicity) or CSI scores (12% moderate or high exposure).

A sizeable proportion of sites (8-48%) had toxicity results meeting SQO guidelines, but chemical contaminant exposures that did not, particularly for CA LRM. This discrepancy may be due to the lack of bioavailability of the measured chemical exposure. Metal speciation can affect bioavailability. For example, metal sulfides typically have extremely low solubility. As total metals are evaluated in this study, speciation is unknown. Analysis of dissolved-phase metals, which are generally the species for metals that are bioavailable, would aid, as would analysis of methylmercury. That said, species-specific analysis would be more complex logistically, and in any case, SQO calculations require the measurement of total metals. As well, hydrophobic organic compounds can sequester further into sediment organic matter over time, such that desorption of such compounds to make them more bioavailable would be much slower (Schwarzenbach et al., 2017). This may be particularly the case for legacy organic contaminants, much of which were deposited decades ago.

A small proportion of sites (0-9.6%) had toxicity assessments that did not meet SQO guidelines, but chemical exposure levels that did. One reason for this may be the presence of chemicals that were measured and known to be toxic to aquatic organisms, but are not included in the chemical SQO evaluation (e.g., pyrethroids). Alternatively, there may be unmeasured chemicals that may potentially cause effects that can be identified and evaluated using a battery of techniques incorporating non-targeted chemical analysis and in vitro bioassays (Maruya et al. 2022). Finally, sources of toxicity other than organic contaminants may be in play. For example, ammonia is a reference toxicant for Bight measurements (Mehinto et al. 2025), and may have been responsible for some of the observed toxicity.

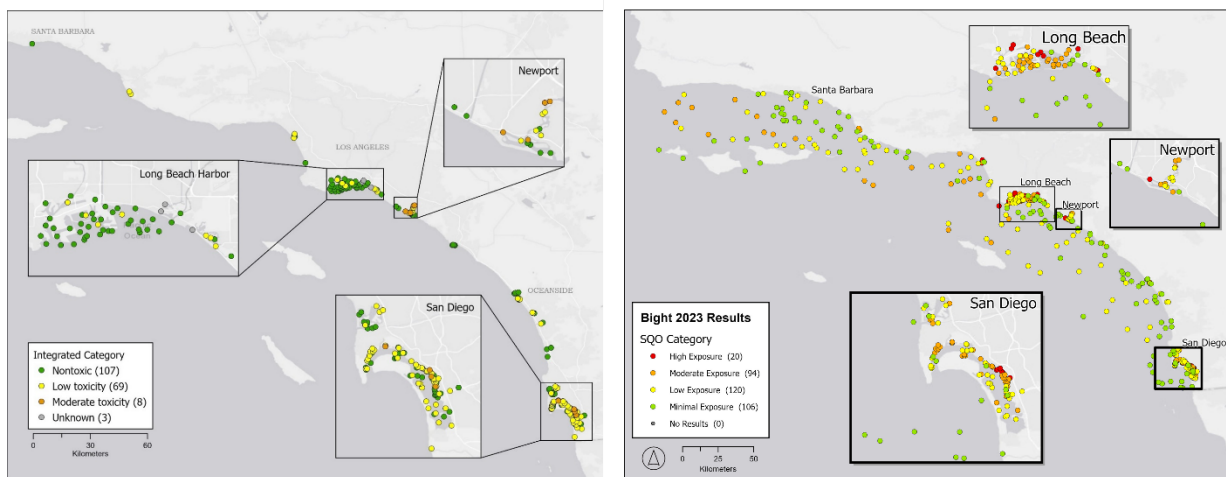


Figure 19: Spatial distribution of toxicity Sediment Quality Objective (SQO) scores (left), and sediment exposure SQO scores (right).

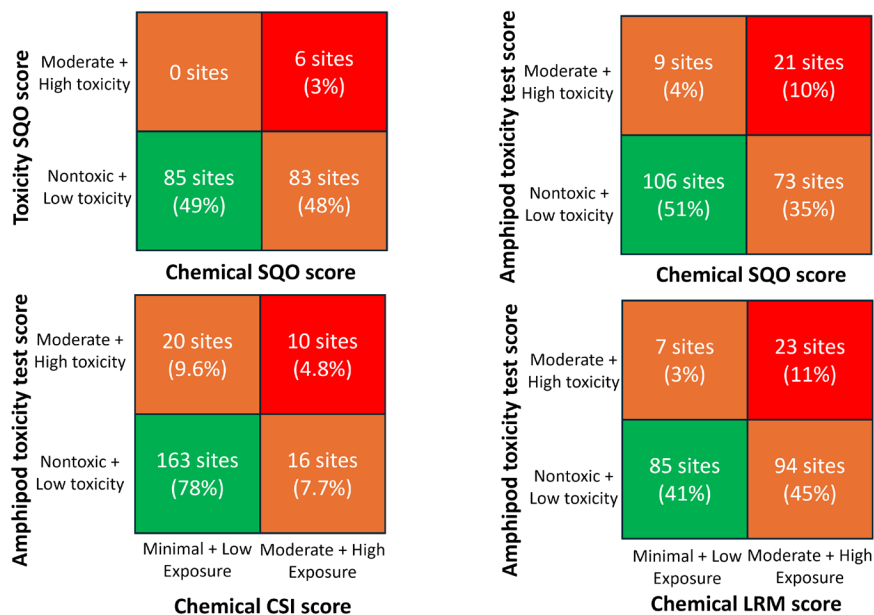


Figure 20: Number of sediment sites, for which both toxicity and chemistry Sediment Quality Objective (SQO) evaluations were both done, sediment and toxicity assessments that met SQO guidelines (minimal + low chemical exposure, nontoxic + low toxicity) and those that did not (moderate + high chemical exposure, moderate + high toxicity).

Extent and magnitude of new analytes in the SCB

The Bight program provides a venue for evaluating new compounds that may be of interest for further monitoring efforts in the region. Accordingly, Bight '23 enabled the analysis of PFAS, neonicotinoid insecticides, and tire-wear compounds, which are discussed here in further detail.

PFAS

Surficial sediment concentrations of PFOS and PFOA, the two most commonly measured PFAS analytes and the two required in Bight '23, were in keeping with observations of concentrations for these compounds in urbanized sediments, both freshwater and saltwater, around the world. Mean concentrations observed in the Southern California Bight were 1.2 and 0.19 ng/g for PFOS and PFOA, respectively (Table 10). These concentrations for PFOA are greater than concentrations found in sediments of Great Lakes tributaries (0.06 ng/g, Loken et al. 2025) but are lower than what was observed in urbanized sediments in Las Vegas and Reno, Nevada (~1 ng/g, Bai and Son 2021). Mean concentrations for PFOS are within the range of mean levels observed in San Francisco Bay from 2012 (1.5 ng/g) to 2023 (0.4 ng/g) (Méndez et al. 2025), and are in line with median levels for PFOA there as well (0.06-0.1 ng/g). Concentrations of both compounds in the Bight are lower than mean levels in sediments of the Great Lakes (Codling et al. 2018), the Seine River in Paris (Macorps et al. 2022), and some heavily impacted sediments

in Chinese cities (Li et al. 2017) and in the Pearl River Delta (Chen et al. 2019, Bian et al. 2025). These observations suggest that typical levels of contamination in the Southern California Bight are in line with the extent of such contamination in other anthropogenically impacted sediments, at least for PFOS and PFOA.

Table 10: Selected mean and maximum (in parenthesis) concentrations (ng/g) of PFOS and PFOA in urbanized sediments. - = not applicable, *=maximum median level reported, ND = not detected.

Location	Mean PFOS concentration (maximum)	Mean PFOA concentration (maximum)	Reference
Southern California Bight, CA	1.2 (68.3)	0.19 (7.1)	This study
San Francisco Bay, CA	1.5 (-)	0.1*	Méndez et al. (2025)
Truckee R. near Reno, NV	ND	1.3 (10)	Bai and Son (2021)
Las Vegas Wash, NV	ND	0.9 (6.3)	Bai and Son (2021)
Pearl River Delta offshore Hong Kong	-- (0.14)	-- (0.015)	Chen et al. (2019)
Pearl River Delta (dry season), China	0.35 (2.5)	1.3 (86)	Bian et al. (2025)
Pearl River Delta (wet season)	0.19 (3.1)	0.03 (0.56)	Bian et al. (2025)
Laurentian Great Lakes	5.1 (63.2)	3 (13.2)	Codling et al. (2018)
Anqing City, China	-- (13)	-- (0.42)	Li et al. (2017)
Laurentian Great Lakes tributaries	0.83 (20.8)	0.06 (0.92)	Loken et al. (2025)
Seine River in Paris, France	1 (-)	0.8 (-)	Macorps et al. (2022)
AFFF site, Lulea, Sweden	-- (64)	ND	Mussabek et al. (2019)
AFFF site, East Coast, USA	-- (120)	-- (2)	Krupa et al. 2025)

There are some hot spots for PFAS in Bight surficial sediments, which are greater in concentration than corresponding hot spots in other urbanized regions, but less than observed at sites known to be heavily impacted by PFAS use. Los Angeles/Long Beach Harbor had the highest concentrations of PFAS for Bight '23 sediments (Figure 45), with the highest concentration observed at 68 ng/g and three other sites at 39-48 ng/g. For PFOA, the highest concentration of 7.1 ng/g was observed in Los Cerritos estuary, with other high concentrations in two stations within San Diego Bay (4-4.8 ng/g) and three stations within Los Angeles/Long Beach Harbor (2.3-3.2 ng/g) (Figure 45). These levels are typically greater than maximum levels

observed at other urban areas (Table 10). However, they are lower than concentrations observed in sediments known to be impacted by PFAS-bearing aqueous film-forming foam (64-120 ng/g PFOS), commonly used for aircraft maintenance (Table 10).

It is clear that concentrations of PFAS were elevated in surficial sediments located inshore (Figure 45). This observation is consistent with anthropogenic activity and use of PFAS in urbanized regions, as shown by strong correlations between sediment concentrations and urban land cover (Loken et al. 2025). Contamination in urbanized areas by PFAS contamination is known to arise from direct manufacturing and use in production of PFAS-bearing products (Bian et al. 2025), industry (Byrne et al. 2025), and AFFF (Mussabek et al. 2019, Krupa et al. 2025). Contamination from sources such as these can end up in both wastewater effluent (Bian et al. 2025), as well as in stormwater in both the aqueous phase and in suspended particulate matter and sediments (Flanagan et al. 2021, Kali et al. 2025). However, specific sources of surficial sediment PFAS are unclear, and cannot be determined in this study.

It is also unclear to what extent PFAS surficial sediment contamination extends offshore the southern California coast. The reporting limits for PFAS across laboratories varied from 0.08-0.5 ng/g (Table 18), with most of the PFAS measurements in strata other than the embayments (i.e., shelf strata) were done by the laboratory with the highest reporting limit (i.e., 0.5 ng/g). However, 16 of the shelf stations were analyzed by another laboratory, which had MDL values for these analyses from 0.008 to 0.02 ng/g. The corresponding RLs were 0.018 to 0.042 ng/g. Thus, while the shelf overall clearly had concentrations of PFOS+PFOA less than 0.5 ng/g, it is plausible that levels of 0.018 to 0.04 ng/g are typical in shelf strata, given the general intercomparability across laboratories for Bight chemistry analytes based on our interlaboratory comparison exercise. Such concentrations are consistent with levels observed offshore in other urbanized sediments, such as in Pearl River Delta (Chen et al. 2019) and further downstream offshore Hong Kong (Bian et al. 2025).

It is unclear what effects may result from exposure to PFAS present in surficial sediments of the southern California Bight. Toxicity data for PFAS in sediment is extremely limited, and there is currently no consensus on sediment threshold values. Bakke et al. (2010) suggested a sediment no-effect level of 220 ng/g as established by the Norwegian Pollution Control Agency, while Simpson et al. (2021) estimated sediment thresholds of 60 ng/g for PFOS to be protective of 99% of species, based on equilibrium partitioning. Casado-Martinez et al. (2021) suggested a threshold for PFOS of 13.5 ng/g. Bai et al. (2025) suggested minimum predicted no-effect levels of 15 ng/g for PFOA and 0.04 ng/g for PFOS, derived from the NORMAN Ecotoxicology Database (<https://www.norman-network.com/nds/ecotox/lowestPnecsIndex.php>). The highest concentrations of PFOS observed in Bight are comparable to the threshold levels suggested by Bakke et al. (2010), Simpson et al. (2021), and Casado-Martinez et al. (2021). On the other

hand, the mean PFOS concentrations observed in Bight surficial sediments is greater than the no-effect levels used by Bai et al. (2025). To confound matters further, water mixed with sediments with PFAS at hundreds of ng/g concentrations had no adverse effects on acute survival of estuarine and freshwater test species (Krupa et al. 2025). It is recognized that there is currently insufficient data and a lack of scientific consensus to set definitive sediment thresholds for PFAS (Valsecchi et al. 2017).

Because Bight '23 is the first Bight cycle to analyze PFAS, this evaluation is limited. Two such limitations stand out. First, only two PFAS compounds were analyzed. Factors that went into analysis of only PFOS and PFOA included cost, availability of laboratories for PFAS analysis, concern for methodology, and the fact that these two commonly studied PFAS compounds were considered to be the best to measure given that Bight had never analyzed this class of compounds to date. While laboratories could and did report on other PFAS analytes (e.g., data meeting DQOs), such measurements did not necessarily cover the entire Bight or all of individual strata. As a result, no further conclusions can be drawn. However, it is clear that many other PFAS compounds are present in the environment. Levels of some such compounds may dwarf those of PFOS and PFOA. For example, perfluorodecane sulfonic acid (PFDS) and perfluorobutylsulfonic acid (PFBS) were found in urbanized Nevada sediments (Bai and Son 2021) at levels up to 88 ng/g and 29 ng/g, respectively, considerably greater than corresponding levels for PFOA (up to 10 ng/g). Short-chain PFAS compounds such as PFBS, as well as products such as hexafluoropropylene oxide dimer acid (HFPO-DX or Gen-X) are more prevalent in current-use PFAS formulations, given phase-out of older products bearing longer carbon chains (Workman et al. 2019). Analysis of PFAS analytes other than PFOS and PFOA is important, given recent regulations promulgated for Maximum Contaminant Levels by EPA on drinking water not just for PFOS and PFOA but also for perfluorohexanoic sulfonate (PFHxS), perfluorononanoic acid (PFNA), and Gen-X. These other analytes are readily measured using typical PFAS analytical methodology, such as EPA Method 1633.

The second limitation is that PFAS is commonly found in the aqueous phase, not just in sediments. Indeed, concentrations in urbanized surface waters are often nontrivial. For example, Bai and Son (2022) found average levels of 2-27 ng/L, and maximum levels of up to 66 ng/L, for PFOS and PFOA in waters overlying the contaminated sediments they measured. Similarly, average and maximum levels for PFOA were 4 and 21 ng/L, and for PFOS were 8 and 220 ng/L, in overlying waters of the Pearl River Delta, China (Bian et al. 2025). Concentrations of PFOS at AFFF sites have been reported at 410 ng/L (Mussabek et al. 2019). It is unknown what corresponding water column concentrations of PFAS are in the Southern California Bight, but it is likely that such concentrations may also be at levels comparable to those reported elsewhere. As well, porewater concentrations of PFAS in the SCB are currently unknown, but would underpin any sediment-water fluxes of PFAS that may be occurring. To be clear, Bight is

primarily a sediment monitoring program, and therefore focuses on sediment quality. However, it cannot be ruled out that aqueous-phase PFAS compounds, which are likely to be present as well at elevated concentrations, would influence levels of PFAS associated with sediment particulates and porewaters, and vice versa (e.g., concentration gradients transferring chemical from one environmental phase to another). Increasing levels of PFAS in waters may eventually result in greater sediment concentrations over time as well.

Neonicotinoid insecticides

Neonicotinoids were detected only at scattered locations within Bight sediments (Figure 41). These locales were generally in embayments, particularly Los Angeles/Long Beach Harbor and Newport Bay. Interestingly, neonicotinoids were detected at only one station each in sediments of Santa Monica Harbor and San Diego Bay, both of which are also heavily impacted by anthropogenic activity.

As with PFAS, neonicotinoid concentrations in Bight surficial sediments were generally in line with those observed in other urbanized sediments. This was particularly the case for thiamethoxam and imidacloprid, which had average (0.017 and 0.37 ng/g, respectively) concentrations similar to some of those observed, for example, in the Pearl River Delta, China (Table 11) impacted by both urban and agricultural use of neonicotinoids (Zhang et al. 2022). Bight sediment concentrations for these two pesticides were lower than those observed in western Canada's Prairie Pothole region (Table 11), which is susceptible to agricultural inputs, although the maximum imidacloprid concentration observed in Bight (59 ng/g) was greater than that at the Prairie Potholes. While average concentrations of imidacloprid, acetamiprid, and thiacloprid were also comparable, maximum concentrations in the Bight (14-59 ng/g) were considerably greater. These levels for imidacloprid were found in Newport Bay for imidacloprid (59 ng/g) and acetamiprid (14 ng/g), and within Los Angeles/Long Beach Harbor for thiacloprid (12-25 ng/g). The neonicotinoid residues in embayment sediments were likely due to urbanized inputs, such as non-point runoff from local rivers (Maruya et al. 2022), as these pesticides are commonly used for a variety of applications ranging from termite protection to pet flea control in products such as collars.

It is unclear what potential effects may arise from exposure to neonicotinoids in Bight surficial sediments. As with PFAS, sediment toxicity thresholds are poorly developed, as suitable data is scarce. An aqueous benchmark for total neonicotinoids of 35 ng/L for long-term chronic effects and 200 ng/L for short-term acute effects has been suggested (Morrissey et al. 2015). For imidacloprid, analogous freshwater invertebrate benchmarks of 10 ng/L and 385 ng/L, respectively, have been suggested (USEPA, 2017). Neonicotinoid insecticides have log organic carbon normalized sediment-water partition coefficients ranging from 1.8 for thiamethoxam to 3.7 for thiacloprid (Zhang et al. 2022). Based on these values, the maximum observed sediment

concentrations (Table 11) and using the maximum organic carbon content of 6.7% observed for all Bight '23 sediments (Figure 41), the maximum estimated equilibrium porewater concentrations for neonicotinoids in Bight surficial sediments can be approximately 180 ng/L for thiamethoxam, and 74 ng/L for thiacloprid. This level is within some of the benchmarks stated above. It should be noted, however, that actual porewater levels are unknown, and are likely lower than these estimations.

It should be noted that neonicotinoids are typically found in the dissolved phase and not associated with sediment particulates. Indeed, surface water concentrations averaged 2-30 ng/L in the Pearl River Delta, with maximum levels as high as 180 ng/L (Zhang et al. 2022). Emsinger et al. (2013) observed median and maximum concentrations for imidacloprid of 50 and 160 ng/L, respectively, in suburban Orange County, while Batikian et al. (2019) found average and maximum levels of this compound of 47 and 86 ng/L, respectively, in Forester Creek in the San Diego metropolitan area. The elevated levels of neonicotinoids observed in Bight sediments are plausibly from inputs from surface water runoff. Levels in such runoff are currently unknown, but previous measurements of neonicotinoids in southern California urban surface freshwaters did exceed existing toxicity thresholds (Emsinger et al. 2013, Batikian et al. 2019). As with PFAS, the effect of aqueous-phase neonicotinoids (both water column and porewater) on sediment concentrations in the Bight is currently unknown.

Table 11: Selected mean and maximum (in parenthesis) concentrations (ng/g) of the neonicotinoid insecticides thiamethoxam (THM), clothianidin (CLO), imidacloprid (IMI), acetamiprid (ACE), and thiacloprid (THA) in urbanized sediments. - = not applicable, *=maximum median level reported, ND = not detected.

Location	Mean [THM] (max)	Mean [CLO] (max)	Mean [IMI] (max)	Mean [ACE] (max)	Mean [THA] (max)	Reference
Bight, CA	0.017 (1.2)	ND	0.37 (59)	14 (14)	1.1 (25)	This study
Pearl River Delta (spring), China	0.43 (4.3)	0.59 (1.9)	1.7 (7.2)	0.71 (5.3)	0.085 (0.24)	Zhang et al. (2022)
Pearl River Delta (summer), China	0.068 (0.29)	0.14 (0.38)	0.61 (5.8)	0.18 (2.0)	0.05 (0.19)	Zhang et al. (2022)
Prairie Pothole region, western Canada	20 (20)	2.8 (3.3)	17.5 (17.5)	--	--	Main et al. (2014)
East China Sea, China	0.12 (0.25)	0.087 (0.23)	--	0.85 (1.1)	0.15 (0.38)	Chen et al. (2022)

Tire-wear compounds (6-PPD-quinone)

As noted, 6-PPD-quinone was detected in surficial sediments only at scattered embayment sites within Los Angeles/Long Beach Harbor, Santa Monica Bay, and Newport Bay. This compound was not detected in San Diego Bay sediments. One laboratory reported concentrations at their RL levels. However, these RL levels were in the range of 0.08-0.1 ng/g, and therefore met Bight-wide DQOs (Table 7). In addition, few measurements exceeded the laboratory's RL. Accordingly, non-detected values from this laboratory were treated as zeroes.

When detected, concentrations were low, with a median of 3.4 ng/g and a maximum concentration of 18.6 ng/g in Newport Bay (Figure 47). These concentrations are comparable to those observed in urbanized sediments elsewhere. For example, in the heavily populated and industrialized Pearl River Delta of China, sediment concentrations of 6-PPD-quinone ranged from 2-18 ng/g in urban rivers (median 9 ng/g), dropping to 2-5 ng/g (medians from 1-3 ng/g) in the river estuary and further off-coast into the South China Sea (Zeng et al. 2023). Greater concentrations of up to 46 ng/g were observed in surficial sediments of the Jiaojiang River near Taizhou in central coastal China (Zhu et al. 2024).

The low sediment concentrations of 6-PPD-quinone are consistent with both its physical-chemical properties and its likely source. Many *p*-phenylenediamine compounds have low propensity to sorb to natural organic matter, such as that in sediments. For example, 6-PPD-quinone has a log octanol-water partition coefficient of 4 (Zeng et al. 2023), which is two to four orders of magnitude lower than for hydrophobic chemicals measured in Bight (e.g., PAHs, PCBs, and organochlorine and pyrethroid insecticides). In addition, *p*-phenylenediamines are antioxidant additives in tire rubber. As such, they likely enter the aquatic environment within rubber particles that have worn off tires. The chemicals then may leach into the aqueous phase, either in the water column, or within sediment porewaters for rubber particulates that settle to the bottom. In both cases, compounds with low partitioning capacity would be more amenable to desorption to the dissolved phase. The quinone is formed from ozonation and photooxidation reactions of 6-PPD. For all these reasons, concentrations in the water column (which was not measured in Bight '23) are likely to be at least comparable to those in sediments, as was observed in the Jiaojiang River, which had up to 21 ng/L of 6-PPD-quinone (Zhu et al. 2024). Such levels are typical of surface waters (Jiang et al. 2024), but could be dwarfed by concentrations in stormwater runoff, which have been observed to be as high as 86-1400 ng/L (Challis et al. 2021).

Ecological thresholds for tire-wear compounds are currently not established in sediment (Jiang et al. 2024). However, based on equilibrium partitioning, the maximum sediment porewater

concentration would be approximately 30 ng/L, based on the maximum sediment levels observed. Such a concentration is lower than the LC50 toxicity thresholds of 95 ng/L (Tian et al. 2022) observed for sensitive aquatic species such as coho salmon (*Oncorhynchus kisutch*). However, it is greater than both the EPA Aquatic Life Screening Value of 11 ng/L (USEPA 2024a) for freshwater, and a more recent dose-response regression analysis suggesting a lower value of at most 1 ng/L is more protective of coho salmon (Stark 2025). Data is currently insufficient to develop analogous values for estuarine or marine water (USEPA 2024a). In any event, toxicity to *p*-phenylenediamine compounds is highly species-specific (Jiang et al. 2024), as other organisms are much less sensitive. For example, the LC50 for rainbow trout (*Oncorhynchus mykiss*), a closely-related salmonid natively present in southern California as steelhead trout, is an order of magnitude greater at 1.0 µg/L (Brinkmann et al. 2022). Similarly, Hiki et al. (2021) found that 6-PPD-quinone exhibited no acute toxicity to its aqueous solubility limit (67 µg/L) for other freshwater fish (*Danio rerio* and *Oryzias latipes*) and invertebrates (*Daphnia magna* and *Hyalella azteca*). The parent compound, 6-PPD, was not measured in Bight '23, but is also likely to be present in sediments at levels comparable to or somewhat greater than 6-PPD-quinone (e.g., up to 172 ng/g, Zhu et al. 2024). In summary, the predicted aqueous (porewater) concentrations of tire-wear compounds in SCB sediments may exceed some suggested threshold values but not others; such values are currently in their infancy.

Reporting limit issues

A collaborative, performance-based monitoring program, such as Bight, needs to have capability and comparability. One element of comparability is sensitivity, defined by the RL in the sediment chemistry portion of Bight '23. Having similar sensitivity is important, because widely differing RL values can result in bias when one laboratory cannot detect contaminants that other laboratories can. Alternatively, bias can also result when laboratories report data using widely differing RL values, for the same reason. Logistical constraints can also be a factor. For example, laboratory availability may be constrained by location and by contracting issues. Also, having only a few laboratories, or a single laboratory, analyze samples may introduce bias as well if there are systematic issues with the results.

In general, laboratories participating in Bight '23 chemistry had comparable RLs, with most exceptions arising from slightly elevated RLs for scattered individual batches (Table 18). While one laboratory did have higher RLs for PFAS (0.5 ng/g) than others, this did not affect overall results (Figure 45); measurements of shelf levels in the 0.02 to 0.04 ng/g range by another laboratory suggest that such concentrations are likely to be more representative of PFAS levels in the shelf. A similar issue arose for 6-PPD-quinone, but all DQOs were met in that case. It would be useful for multiple laboratories to analyze samples from a specific locale or stratum, to provide robustness in data quality for future work.

To account for the bias potentially introduced by MDLs, a sensitivity analysis was performed to determine if there were changes in chemical exposure SQO scores. In this analysis, non-detected values, reported as -88 in the Bight database, were assigned either half the MDL reported by the analyzing laboratory, or the full MDL value, instead of zero. Six of the 340 sites changed chemical exposure category in the former case, and an additional six sites changed in the latter (Table 12). All sites were on the shelf or slope strata. When half the MDL was used, one site changed from minimal to low exposure, and the remaining five changed from low to moderate exposure. Using the full MDL reported, three sites changed from minimal to low exposure, and three others from low to moderate. A maximum of ten sites, or 2.9% of the 340 sites, were affected, a proportion similar to the 9 sites of 376 total with changed SQO scores from an analogous MDL sensitivity analysis for Bight '18 (Du et al. 2020).

The overall assessment of sediment exposure that met SQO guidelines changed from $83\pm 9.3\%$ to $75\pm 8.6\%$ if half the MDL was used for non-detected values; if the full MDL value was used, the percentage changed to $73.5\pm 8.2\%$ (Figure 21). For individual affected strata, sediment exposure that met SQO guidelines changed from $96.7\pm 14.0\%$ to $93.3\pm 13.8\%$ for the inner shelf, $89.7\pm 15.3\%$ to $89.7\pm 14.7\%$ for the mid-shelf, $96.3\pm 16.4\%$ to $96.2\pm 12.1\%$ for the outer shelf, $60.0\pm 17.0\%$ to $53.3\pm 16.6\%$ for the upper slope, and $82.7\pm 15.8\%$ to $65.5\pm 14.1\%$ for the lower slope using the most extreme substitution with the full MDL (Figure 22). The most dramatic changes were observed for the lower and upper slopes, for which substitution reclassified affected sites from low to moderate exposure; however, only the lower slope had areal changes that were greater than the associated 95% confidence intervals.

The changes in areal assessment are considerably greater than those observed for Bight '18. In that cycle, the overall assessment changed by 0.02% and that for individual strata changed by only a few percentage points, despite a similar number and proportion of sites being affected by substitution sensitivity analysis (Du et al. 2020). However, it is important to note that the sites in Bight '18 for which SQO scores changed in the sensitivity analysis were all embayment sites. In Bight '23, all such sites were offshore. Stations of the slope and shelf have large area weights, particularly compared to stations in embayments. As a result, changes in concentrations used for SQO calculations will be accordingly larger, as they are here, particularly at the slopes where the SQO classification changed from compliant to non-compliant.

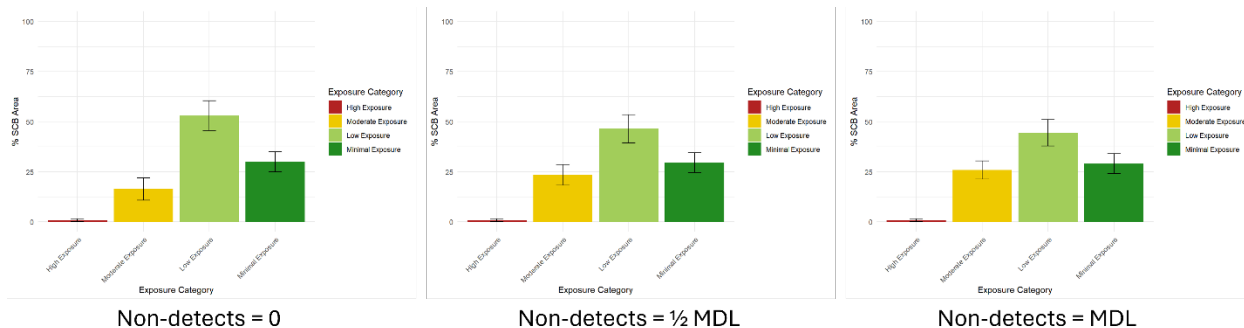


Figure 21: Sediment Quality Objective (SQO) sensitivity analysis, using zero for non-detected values (left), half the method detection limit (MDL) (center), and the full MDL (right).

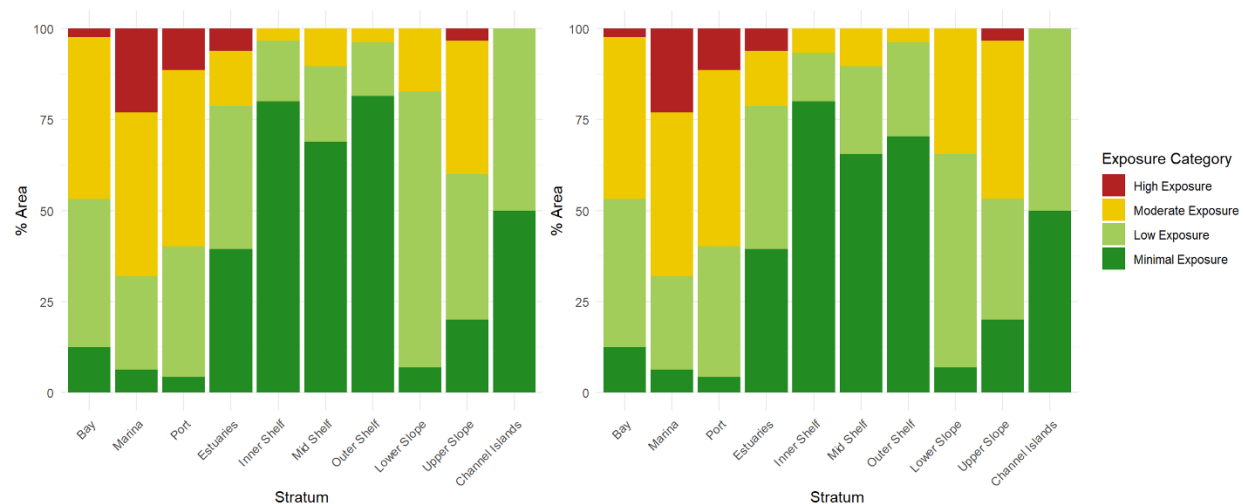


Figure 22: Sediment Quality Objective (SQO) sensitivity analysis by strata, using zero for non-detected values (left) and the full MDL (right).

Table 12: Sites for which substitution of half or full Method Detection Limit (MDL) values changed in Sediment Quality Objective (SQO) categorizations.

Stratum	Bight '23 region (site)	Original exposure category	Exposure category (1/2 MDL)	Exposure category (full MDL)
Inner shelf	B23-12199	low	Low	moderate
Mid-shelf	B23-12235	minimal	Low	low
Outer shelf	B23-12260	minimal	Minimal	low
Outer shelf	B23-12262	minimal	Minimal	low
Outer shelf	B23-12265	minimal	Minimal	low

Stratum	Bight '23 region (site)	Original exposure category	Exposure category (1/2 MDL)	Exposure category (full MDL)
Upper Slope	B23-12301	low	low	moderate
Upper Slope	B23-12684	low	moderate	moderate
Lower Slope	B23-12333	low	moderate	moderate
Lower Slope	B23-12339	low	moderate	moderate
Lower Slope	B23-12340	low	moderate	moderate
Lower Slope	B23-12342	low	moderate	moderate
Lower Slope	B23-12343	low	low	moderate

VII. CONCLUSIONS

The Bight '23 Program provided a regional assessment of the Southern California Bight's sediment chemistry through analysis of multiple chemical constituents. Based on the results of this survey, the Chemistry Technical Committee had several conclusions, as follows:

The majority of the SCB has minimal to low sediment contamination.

The SQO sediment chemistry assessment tool integrates sediment concentrations from several key constituents into one of four sediment exposure categories. Using the SQO assessment tool to assess extent, an estimated 83% of SCB sediments have exposure levels that met SQO guidelines (i.e., minimal to low exposure) to the chemical contaminants included in the SQO. Only 1% of SCB sediments have high exposure, the worst category of contamination. While the SQO assessment tool is specifically designed, calibrated, and validated for embayments, we also used it for offshore sediments because no assessment tool exists for offshore sediments. Varying levels of detection limits may have some influence on SQO scoring, as evidenced by the sensitivity analysis performed in this study.

Embayment strata had the greatest extent and magnitude of sediment contamination.

While 83% of the area of the SCB had sediment contamination that met SQO guidelines, not all strata were equally impacted by sediment contamination. Embayments had the lowest relative extent of chemical contaminant exposure that met SQO guidelines (47%), followed by the slopes and basin (77%), and then the shelf (93%), with the Channel Islands stratum having 100% of its area at minimal to low exposure levels. In addition, the highest concentrations were most commonly found in the embayment stratum. Of the four embayment strata, the greatest

extent and magnitude of high sediment exposure occurred in marinas and estuaries. Marinas and estuaries are both habitats with multiple pollutant inputs (except for DDT, for which additional historical contamination offshore is documented) and relative quiescent waters that do not promote mixing.

Over time, there has been limited and modest improvement in sediments based on chemistry SQO scores.

There were no statistically significant changes in chemistry SQO scores from 2003 to 2023 Bight-wide. Some apparent improvements in sediment quality, based on these SQO scores, were observed over this time period in many strata. However, these changes were not statistically significant, except for the increase in chemical exposure conditions meeting SQO guidelines (i.e., minimal to low exposure) in the lower slope, and the increase in the area categorized as minimal exposure in the outer shelf. The lack of appreciable changes in SQOs over time is consistent with the general similarities in spatial distributions and concentration levels of chemical analytes measured across multiple Bight cycles. There have been considerable increases in population and anthropogenic activity (e.g., shipping traffic in ports) since the inception of the Bight program in the 1990s. This fact suggests that maintaining similar SQO scores overall during this period, and indeed making modest improvements in certain cases, reflects effectiveness in management efforts in keeping up with growing chemical stressors.

Contaminants of Emerging Concern (CECs) measured in Bight '23 for the first time are at levels consistent with those in other urbanized sediments.

2023 was the first time the Bight Program measured PFAS, tire-wear compounds, and neonicotinoid insecticides in surficial sediments. These compounds were generally found only in embayments, and were detected intermittently for tire-wear compounds and neonicotinoids. Concentrations of all these new analytes were in the ranges of surficial sediments measured in other urbanized regions of the world.

VIII. RECOMMENDATIONS

Based on the efforts from Bight '23, both the Chemistry Technical Committee and the Sediment Quality Planning Committee agree on the following recommendations to follow up on current survey results, or to improve the next implementation of the regional survey (e.g., Bight '28).

Recommendation 1: Incorporate risk-based CEC prioritization process for future Bight cycles.

While monitoring is required for priority chemical pollutants, many agencies also include select classes of unregulated constituents of emerging concern (CECs). Indeed, the Bight Program

itself is an opportunity to see if CEC occurrence in the region may be a potential issue. However, it is not necessarily clear *a priori* which CECs are of greatest concern to human or ecosystem health, and therefore should be monitored. The other side of that coin is also true, in that it is not clear what CECs can be de-prioritized. As well, neither is it clear how often either monitoring or de-prioritization should be done.

Risk-based categorization and prioritization can be used to address such issues. A risk-based approach for CECs was recommended and adopted by the State of California. These were initiated by the State Water Board's first CEC Expert Panel for aquatic ecosystems in 2012 (Anderson et al. 2012), for which strategies can be incorporated to calculate risk and guide prioritization (Drewes et al. 2023). The goal of such frameworks is to rank CECs based on occurrence and risks to human and ecosystem health, using a consistent prioritization framework to develop monitoring and management recommendations. Such frameworks are intended to be periodically updated, as new knowledge, techniques, and CECs come into play (Drewes et al. 2023). Similar approaches have been taken regionally, such as in the San Francisco Bay (Sutton et al. 2024). A current effort is underway to develop a prioritization framework for prioritizing and managing CECs for Southern California, to address specific regional specific issues and needs. This joint project between SCCWRP and its member agency Commission's Technical Advisory Group (CTAG) began in fall 2025 and is slated to report in spring 2027. Recommendations from risk-based categorization and prioritization should be considered in future Bight cycles.

Recommendation 2: Consider changes to analytes measured in Bight '23.

Sub-recommendation 2a: Reduce effort in sediment for CECs with low extent and magnitude, and potential for effects.

Several new analytes and analyte classes were introduced in Bight '23, including neonicotinoid insecticides and tire-wear compounds. Only scattered detections of both classes were detected in surficial sediments, even in heavily urbanized and impacted areas such as embayments. Detected values, while in line with levels observed in sediments from other urbanized areas worldwide, were low. Indeed, some porewater levels for these compounds, based on equilibrium partitioning, may be below known effects thresholds.

Accordingly, further sediment monitoring for these compounds in future Bight cycles may not be as effective as monitoring for other analytes, or monitoring for these compounds in other environmental media. For example, fipronil compounds, monitored for the first time in Bight '18 (Du et al. 2020), were not analyzed in the current Bight cycle for similar reasons (Du et al. 2021).

Sub-recommendation 2b: Increase effort for exploring CECs with potential for large extent and magnitude, or potential for effects.

In Bight '23, PFAS was also measured for the first time, but was limited to two compounds: PFOS and PFOA (Table 4). Sediment contamination was generally limited to embayments at concentrations within the ranges observed in other heavily urbanized sediments. However, the vast majority of other such studies in the literature analyze for many more PFAS compounds than just PFOS and PFOA. There is increasing interest and concern about environmental contamination of PFAS, including those compounds beyond PFOS and PFOA, as evidenced by recently promulgated regulations and Maximum Contaminant Levels by EPA on drinking water for PFOS, PFOA, PFHxS, PFNA, and HPDO-DA (GenX). While analogous regulations on PFAS in the ambient aquatic environment do not currently exist, these are likely to be developed further in the future.

Accordingly, an understanding of PFAS levels, beyond just PFOS and PFOA, in the coastal environment of Southern California would provide an understanding of where contamination exists and to what extent, as well as where it does not. It should be noted that typical PFAS analysis methods (e.g., EPA Method 1633) already include such analytes, which are measured in the same chromatographic run as PFOS and PFOA. Thus, little additional effort, specifically reporting, would be necessary to include further PFAS analytes, as was already done by one laboratory in Bight '23.

Another example of further exploration of new CECs would entail expansion of the current DDT analyte list (Table 4). This includes not only 2,4'- and 4,4'-DDT and their major degradates both aerobic (DDE) and anaerobic (DDD), but also 4,4'-DDMU, which is formed from reductive dechlorination of 4,4'-DDE. Metabolites such as DDMU, first included in Bight '08, its further reductive dehalogenation product DDNU, and other transformation products (e.g., DDOH, DDNS, DDA, DDM, DBP) are being increasingly detected in sediments, surface waters, and biota off southern California (USEPA 2024, Mackintosh et al. 2018, Kivenson et al. 2019, USEPA 2024), likely from degradation of sediment-borne contaminants. The occurrence and extent of these potentially toxic degradation products is unknown.

Recommendation 3: Investigate other abiotic environmental media in the Southern California Bight.

The Bight program has historically measured surficial sediments for many reasons. For example, many chemical analytes sequester in sediments, which serve as an integrator of such contamination. However, sediment is not the only pathway of exposure to aquatic ecosystem and human health. The goal of Bight chemistry is to evaluate occurrence and exposure to chemical stressors. Many hydrophobic and legacy chemical contaminants such as PCBs and

DDT, are primarily found in sediments in the Bight, leading to sediment exposure being an important potential pathway for toxicity for these stressors. However, we have noted in Bight '23 that some chemical analytes are more likely to be present in the aqueous phase, not just in the sediments. Indeed, they may be more prevalent in the former than the latter, given their physical-chemical properties, as well as observations in urbanized sediments and overlying waters elsewhere in the world. These analytes include PFAS, tire-wear compounds, and neonicotinoid insecticides. Concentrations of analytes in the water column may affect concentrations in other environmental media, such as sediments, from processes such as sediment-water exchange from concentration gradients. Accordingly, increases in aqueous concentrations over time, which may occur for many current use compounds (e.g., CECs), may result in greater sediment concentrations over time. In addition, water column concentrations of some hydrophobic chemicals, such as DDT, are of considerable interest in the region, given high concentrations in bulk sediments, efflux of these levels to the overlying waters, and potential human and pelagic ecosystem exposure from chemicals in the dissolved phase and in suspended particulate matter (USEPA 2009, USEPA 2024). Porewater would also be of interest for similar reasons.

Accordingly, investigation of chemical stressors in other abiotic Bight environmental media is recommended. In particular, measurement is recommended in the aqueous phase (i.e., water column and porewater), given the likelihood of occurrence of many CECs. Such efforts can take the form, for example, of special studies to investigate concentrations in the water column and/or porewater of at least some of these chemicals, particularly at selected locales most likely to be impacted (e.g., some embayments) to determine aqueous levels and distributions. This knowledge would provide more holistic understanding of exposure levels of these compounds in the aquatic environment, besides just bulk sediment, and may also evaluate the potential for changes in sediment concentrations over time as a result of changing aqueous levels. For some analytes such as DDT and PCBs, a timeline of chemical exposure in the water column can be established (Zeng et al. 1999, Zeng et al. 2002, Zeng et al. 2005, USEPA 2009, Fernandez et al. 2012, Fernandez et al. 2014, Lohmann et al. 2023, USEPA 2024) that can provide insights into temporal changes, if any, of such chemicals in southern California coastal waters.

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APPENDIX A. SEDIMENT CHEMISTRY INTERCALIBRATION EXERCISE

Scope and Requirements

A key point of Bight surveys, past and current, is a substantial effort in the development of analytical comparability. Because such regional programs were conducted in a collaborative manner with multiple analytical laboratories participating, intercalibration studies were critical for analyses of trace metal and trace organic constituents. Although all participating laboratories were certified by the State of California (i.e., by the Environmental Laboratories Accreditation Program) in the analytical methodologies used, there can still be significant discrepancies at times for specific constituents. Therefore, iterative inter-comparison and intercalibration exercises were performed until all laboratories could meet prescribed data quality objectives for inter-laboratory precision. These intercalibrations remain one of the foundational elements of the regional monitoring quality assurance/quality control program.

In the Bight '23 intercalibration study, both certified reference materials (CRMs) and field reference materials (FRMs) were analyzed by Bight '23 participating laboratories in triplicate. Assessment of laboratory performance for comparability was as follows. First, for each analyte, the average for each laboratory was computed as the mean of the triplicate samples. Second, a grand mean was calculated as the average across all laboratories. Potential outlier results were identified using the Grubbs Test, and excluded when calculating the grand mean. Finally, depending on the analyte class, 70 to 80% of the target analytes must achieve passing results to qualify for Bight '23 participation for that compound class. All intercalibration results for PBDEs, pyrethroids, neonicotinoids, tire-wear compounds, and PFAS are for informational purposes only.

Previous Bight cycles included a criterion that for a given analyte, at least two of three replicates must be within the defined variability of the CRM certified value or FRM grand mean, as appropriate, to pass. This criterion was not applied for Bight '23. In addition, for metals, only those metals for which reporting limits were required were used in the evaluation of pass/fail criteria (i.e., aluminum, barium, and iron did not have Bight-wide reporting limits).

Laboratories that participated in the intercalibration exercise were required to meet specified performance criteria, which can be found in the Bight '23 Survey Quality Assurance Plan. Prior to any analysis of the survey samples, laboratories analyzing Bight samples were required to pass the assessment for both CRMs and FRMs.

Overall, all of Bight '23 participating laboratories passed in both CRM and FRM for the designated analyses (Table 13).

Intercalibration Certified and Field Reference Materials

Two field reference materials were used during the intercalibration exercise. The first was provided by the Los Angeles County Sanitation District, sourced from the Palos Verdes Shelf, to test both inter- and intra-laboratory precision when analyzing a sample with high levels of DDT and potential interferences not present in SRM 1944 and ERA 540. The second was a less contaminated sediment, provided by WSP (courtesy C. Stransky), fortified with pyrethroids, neonicotinoids, tire-wear compounds, and PFOS and PFOA at SCCWRP, then sent to intercalibration participants.

Trace Metals

ERA 540 Metals in Soil (Lot #D099-540)

ERA 540 (<http://www.eraqc.com/>) tested method accuracy. Laboratories were required to obtain concentrations within the Performance Testing acceptable limits of this material's certificate for all analytes.

Field Reference Material from Palos Verdes Shelf

Field reference material tested precision, both inter-laboratory and intra-laboratory. This specific material has high levels of DDT natively present, as well as potential interferences not present in ERA 540. Laboratories were required to obtain a concentration within 30% of the grand mean value for 12 of 15 analytes.

Organics

NIST SRM 1944 New York/New Jersey Waterway Sediment

SRM 1944 (<https://catalog.data.gov/dataset/srm-1944-new-york-new-jersey-waterway-sediment-2a3d0>) tested method accuracy. Laboratories are required to obtain concentrations within 40% of the certified or reference value for 80% of the compounds for PAHs, and for 70% of the compounds each for PCBs, OC pesticides, and PBDEs. This material was also used to assess the accuracy and precision for grain size and TOC, as reference values for these analytes are available.

NIST SRM 1941b Organics in Marine Sediment

SRM 1941b (<https://www.nist.gov/image/srm-1941b>), is a marine sediment from Baltimore Harbor. This reference material, which also tested method accuracy, was used as an alternate CRM for organics, because SRM 1944 became unavailable during the intercalibration exercise due to depletion of stock. The same DQO criteria for SRM 1944 applied to SRM 1941. This material was also used to assess accuracy and precision for TOC, as reference values for this

analyte are available. Note: SRM 1941b, unlike SRM 1944, does not have reference values for grain size.

Field Reference Material from Palos Verdes Shelf

Field reference material tested precision, both inter-laboratory and intra-laboratory. This is the same material as that used as a field reference material for metals, and it had potential interferences not present in SRM 1944. Laboratories were required to obtain a total class concentration within 40% of the grand mean value for PCBs, OC pesticides, and PAHs.

Spiked Marine Field Reference Material

This sediment, from a regional locale less contaminated than Palos Verdes Shelf, was provided by C. Stransky, and was fortified at SCCWRP with 20 ng/g wet weight of 6PPD-quinone, PFOS, PFOA, and each pyrethroid and neonicotinoid insecticide. Participating laboratories were not informed of the spiked concentrations a priori, to enable blind sample analysis. This material was used to evaluate precision for tire-wear compounds, PFAS, pyrethroids, and neonicotinoids, as reference materials with certified values for these analytes are not currently available commercially.

Intercalibration Exercise Results

There were a total of three rounds of chemistry intercalibration for Bight '23. Successive rounds beyond the first were necessary to address individual deficiencies in intercalibration.

In the first round (Table 14), all laboratories passed intercalibration for both CRMs and FRMs (i.e., system pass) for PAHs, OC pesticides, PBDEs, and TOC. All laboratories passed intercalibration for CRMs for metals, PAHs, PCBs, and OCs; and also passed intercalibration for FRMs for PAHs, and OCs. CLAEMD did not pass FRM intercalibration for metals, nor did LACSD for PCBs.

The participating laboratories agreed to conduct a second round of intercalibration (Table 15) to address failed aspects of the first round. This FRM used a different material for metals, as none of the first-round FRM for metals was available. The FRM for organics in the second round was actually SRM 1944, which was analyzed without knowledge by the laboratory and evaluated against CRM intercalibration data.

In the second round, all laboratories passed intercalibration for grain size. In addition, LACSD passed FRM intercalibration for PCBs, for which it failed in the first round. However, CLAEMD did not pass FRM intercalibration for metals in the second round.

A third round of FRM intercalibration was done by CLAEMD (Table 16), which analyzed ERA 540 as an unknown. Results were evaluated against CRM intercalibration data for this material, and a pass result was obtained.

Table 13: Summary of overall sediment chemistry intercalibration results (percent success). Laboratory name abbreviations: CLAEMD = City of Los Angeles, Environmental Monitoring Division; CSD = City of San Diego; LACSD = Los Angeles County Sanitation Districts, OC San = Orange County Sanitation District. TOC = total organic carbon, PAH = polycyclic aromatic hydrocarbon, PCB = polychlorinated biphenyls, OC = organochlorine pesticides, PBDE = polybrominated diphenyl ethers, PFAS = per- and polyfluorinated alkyl substances. *For informational purposes only. N/A=not applicable.

Reference material	Analyte class	Criteria	CLAEMD	CSD	LACSD	OC San	Physis	Weck	Pass rate
SRM 1944	Grain size	±20% of target value for %fines (£ 174 mm)	100	100	N/A	100	100	N/A	4/4
SRM 1944	TOC	±30% target value	100	100	100	100	100	100	6/6
SRM 1944	Individual PAHs	±40% target value for 80+% analytes	80	85	95	90	90	N/A	5/5
SRM 1944	Individual PCBs	±40% target value for 70+% analytes	89	92	94	92	100	N/A	5/5
SRM 1944	Individual OCs	±40% target value for 70+% analytes	78	89	75	100	100	N/A	5/5
Organics FRM	Total PAHs	Lab mean ±40% grand mean	100	100	100	100	100	N/A	5/5
Organics FRM	Total PCBs	Lab mean ±40% grand mean	100	100	100	100	100	N/A	5/5
Organics FRM	Total OCs	Lab mean ±40% grand mean	100	100	100	100	100	N/A	5/5
Organics FRM	Total PBDEs	Lab mean ±40% grand mean	N/A	N/A	N/A	N/A	100	N/A	1/1

Reference material	Analyte class	Criteria	CLAEMD	CSD	LACSD	OC San	Physis	Weck	Pass rate
Organics FRM	Pyrethroids*	Lab mean \pm 40% of grand mean for 70+% analytes	N/A	N/A	71	N/A	75	N/A	*
Organics FRM	Neonicotinoids*	Lab mean \pm 40% of grand mean for 70+% analytes	N/A	N/A	58	N/A	67	0	*
Organics FRM	Tire-wear compounds*	Lab mean \pm 40% of grand mean	N/A	N/A	100	N/A	67	0	*
Organics FRM	PFAS*	Lab mean \pm 40% of grand mean for 70+% analytes	N/A	N/A	100	100	100	83	*
ERA 540	Individual Metals	Within PT performance acceptable limits for all analytes	100	100	100	100	100	N/A	5/5
Metals FRM	Individual Metals	Lab mean \pm 30% of grand mean for 80+% analytes	100	88	100	91	94	N/A	5/5

Table 14: Summary of first-round sediment chemistry intercalibration results. Laboratory name abbreviations: CLAEMD = City of Los Angeles, Environmental Monitoring Division; CSD = City of San Diego; LACSD = Los Angeles County Sanitation Districts, OC San = Orange County Sanitation District. TOC = total organic carbon, PAH = polycyclic aromatic hydrocarbon, PCB = polychlorinated biphenyls, OC = organochlorine pesticides, PBDE = polybrominated diphenyl ethers. N/A = not applicable.

Reference material	Analyte class	Criteria	CLAEMD	CS D	LACS D	OC San	Physis	Weck	Pass rate
SRM 1944	TOC	±30% target value	100	100	100	100	100	100	6/6
SRM 1944	Individual PAHs	±40% target value for 80+% analytes	80	85	95	90	90	N/A	5/5
SRM 1944	Individual PCBs	±40% target value for 70+% analytes	89	92	94	92	100	N/A	5/5
SRM 1944	Individual OCs	±40% target value for 70+% analytes	78	89	75	100	100	N/A	5/5
Organics FRM	Total PAHs	Lab mean ±40% grand mean	100	100	100	100	100	N/A	5/5
Organics FRM	Total PCBs	Lab mean ±40% grand mean	100	0	100	100	100	N/A	4/5
Organics FRM	Total OCs	Lab mean ±40% grand mean	100	100	100	100	100	N/A	5/5
Organics FRM	Total PBDEs	Lab mean ±40% grand mean	N/A	N/A	N/A	N/A	100	N/A	1/1
ERA 540	Individual Metals	Within PT performance acceptable limits for all analytes	100	100	100	100	100	N/A	/5

Reference material	Analyte class	Criteria	CLAEMD	CS D	LACS D	OC San	Physis	Weck	Pass rate
Metals FRM	Individual Metals	Lab mean $\pm 30\%$ of grand mean for 80+% analytes	69	88	100	91	94	N/A	5/5

Table 15: Summary of second-round sediment chemistry intercalibration results. Laboratory name abbreviations: CLAEMD = City of Los Angeles, Environmental Monitoring Division; CSD = City of San Diego; LACSD = Los Angeles County Sanitation Districts, OC San = Orange County Sanitation District. PCB = polychlorinated biphenyls. N/A = not applicable.

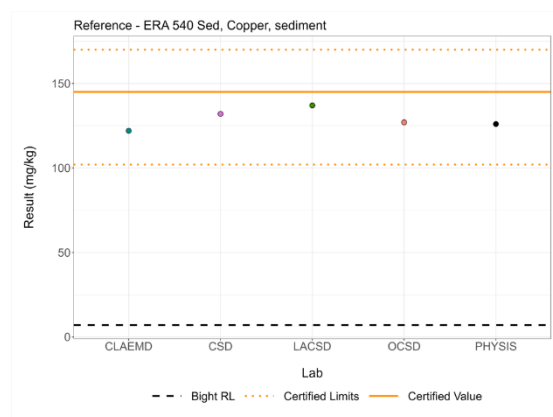
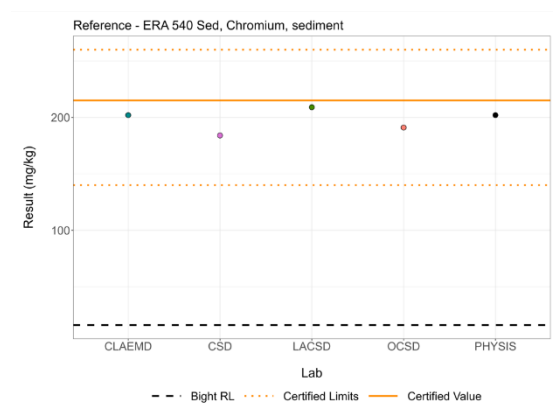
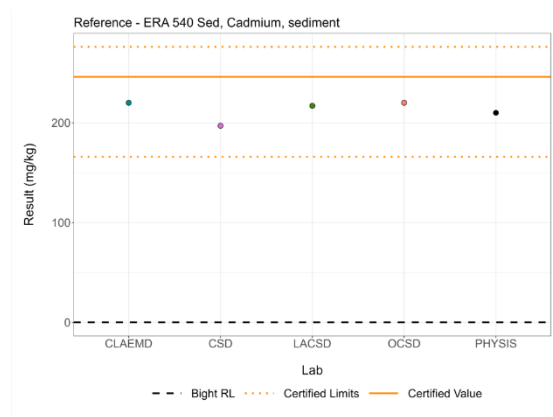
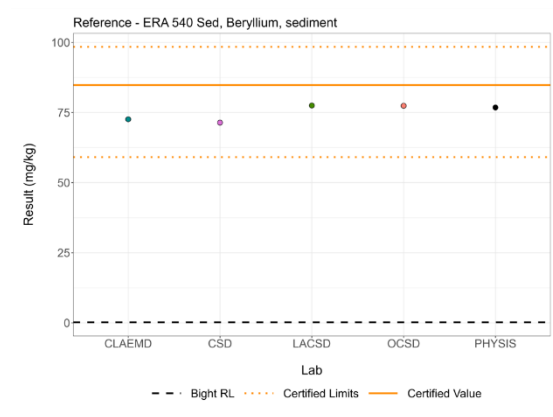
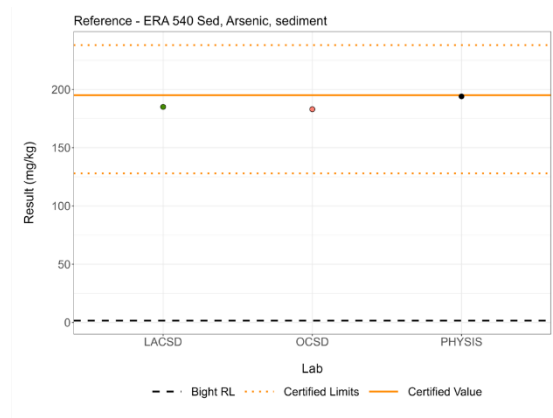
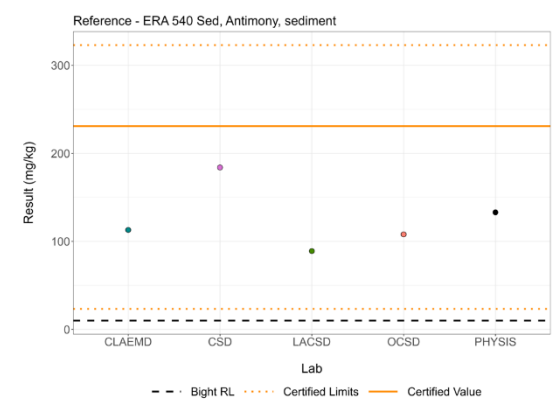
Reference material	Analyte class	Criteria	CLAEMD	CS D	LACS D	OC San	Physis	Weck	Overall pass rate
SRM 1944	Individual PCBs	$\pm 40\%$ target value for 70+% analytes (analyzed as FRM unknown by laboratory)	--	--	100	--	--	--	5/5
SRM 1944	Grain size	$\pm 20\%$ of target value for %fines (≤ 174 mm)	100	100	N/A	100	100	N/A	4/4
Metals FRM	Individual Metals	Lab mean $\pm 30\%$ of grand mean for 80+% analytes	67	80	82	82	80	N/A	4/5

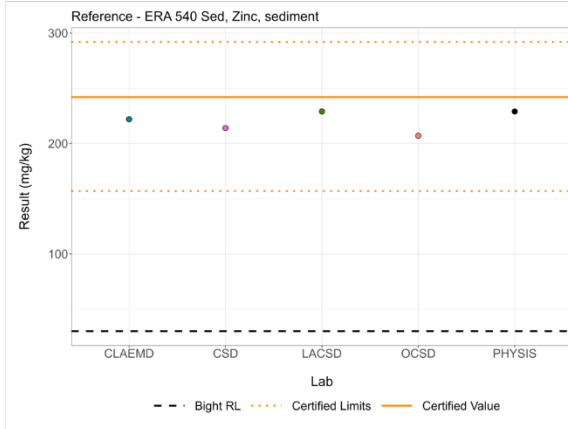
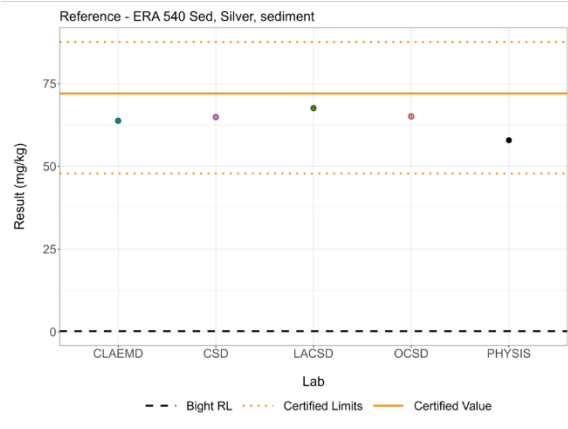
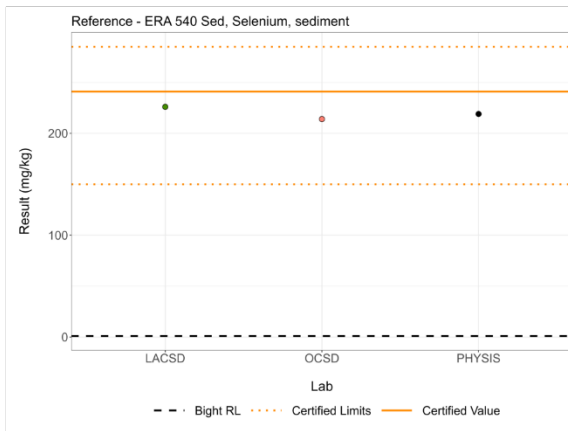
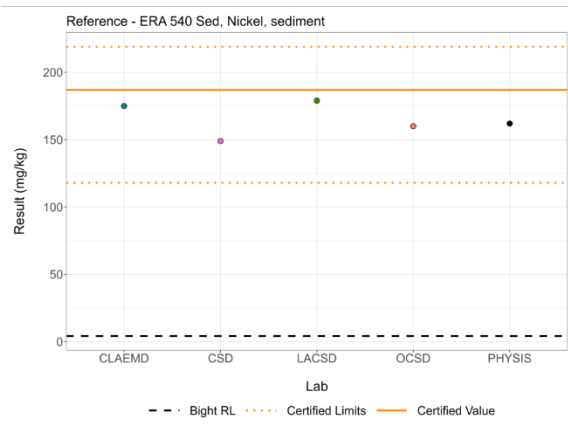
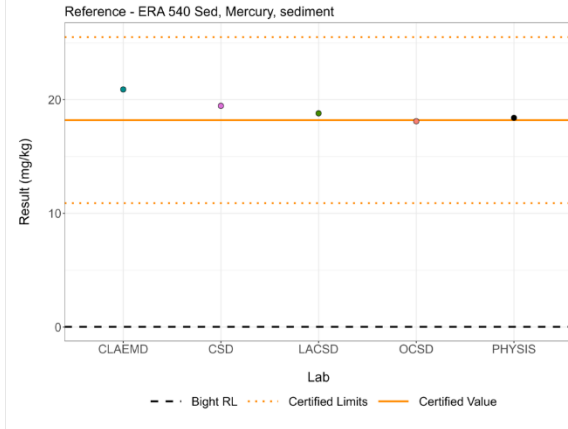
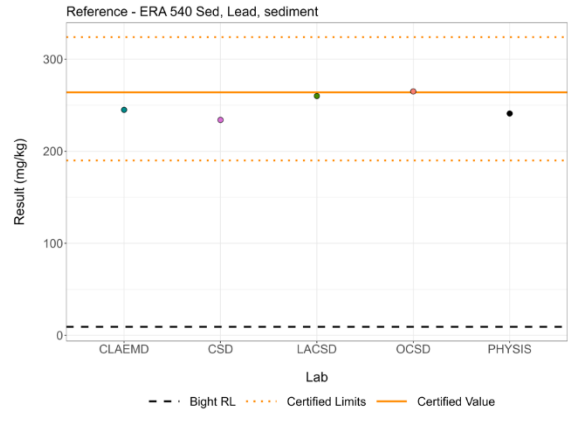
Table 16: Summary of third-round sediment chemistry intercalibration results. Laboratory name abbreviations: CLAEMD = City of Los Angeles, Environmental Monitoring Division; CSD = City of San Diego; LACSD = Los Angeles County Sanitation Districts, OC San = Orange County Sanitation District.

Reference material	Analyte class	Criteria	CLAEMD	CSD	LACSD	OC San	Physis	Weck	Overall pass rate
ERA 540	Individual Metals	Within PT performance acceptable limits for all analytes (analyzed as FRM unknown by laboratory)	100	--	--	--	--	--	5/5

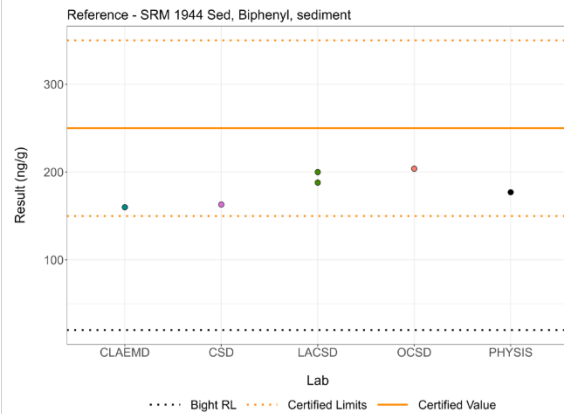
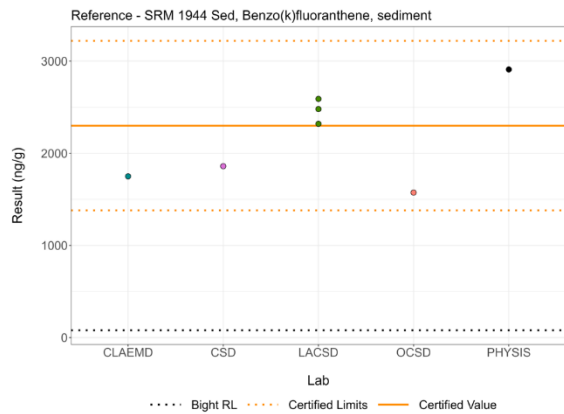
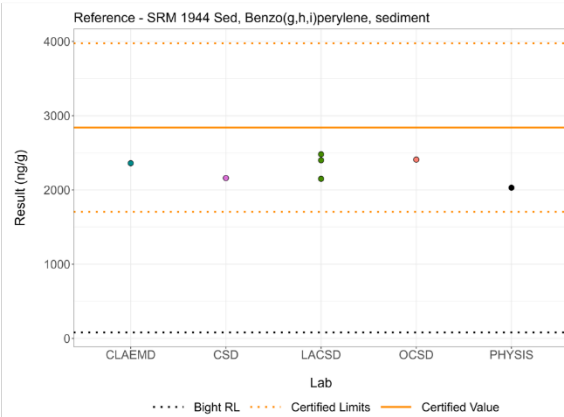
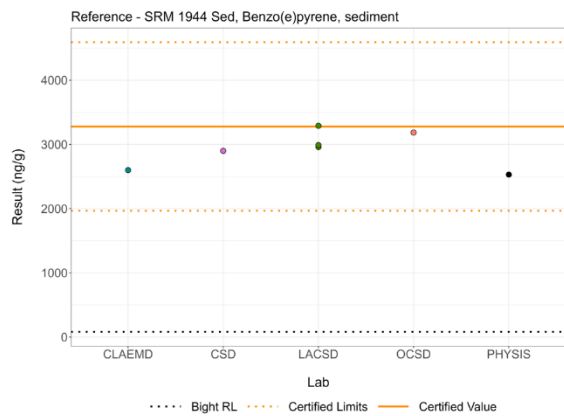
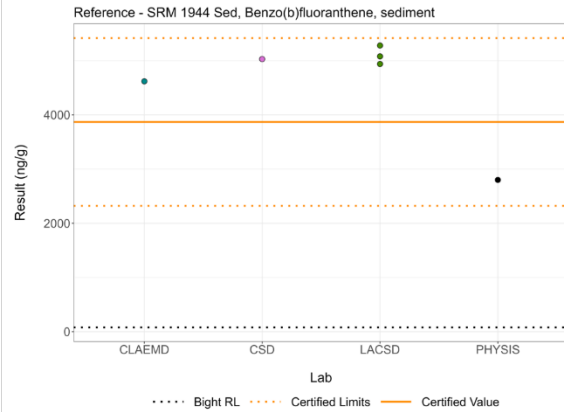
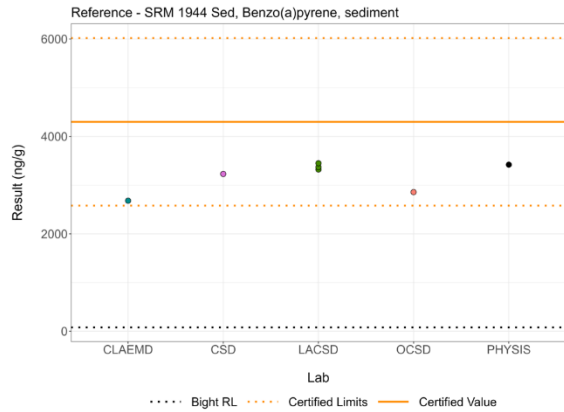
First-Round Intercalibration Results for Core Individual Analytes

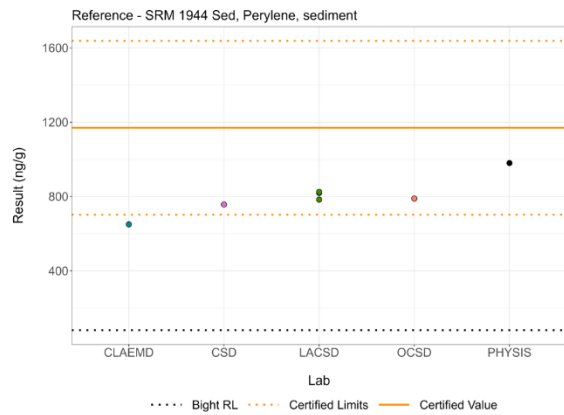
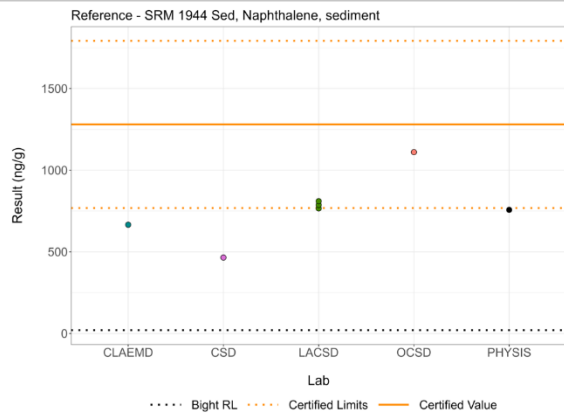
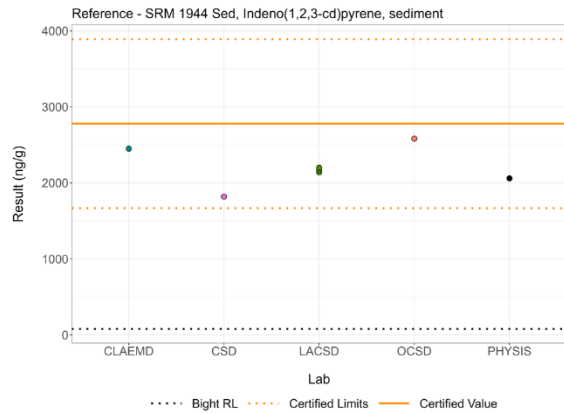
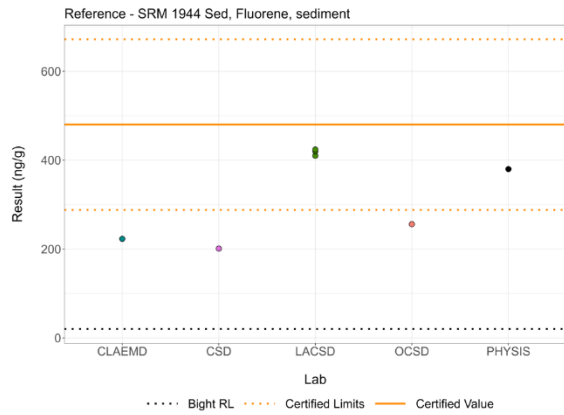
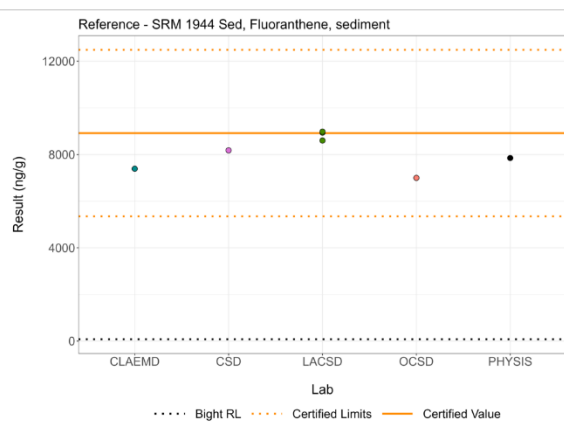
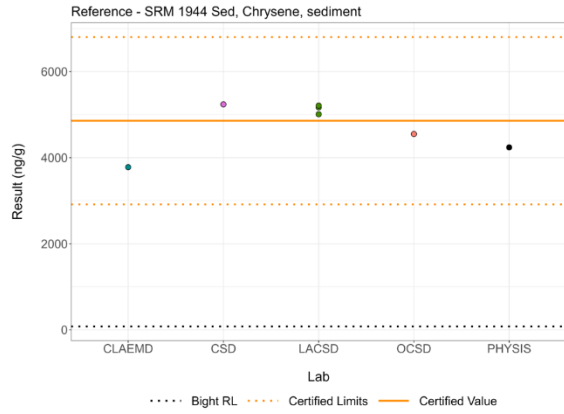
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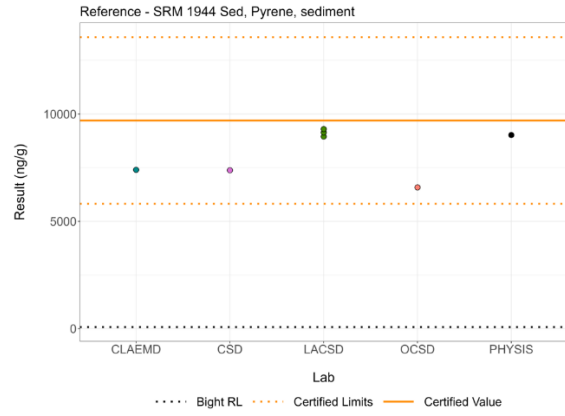
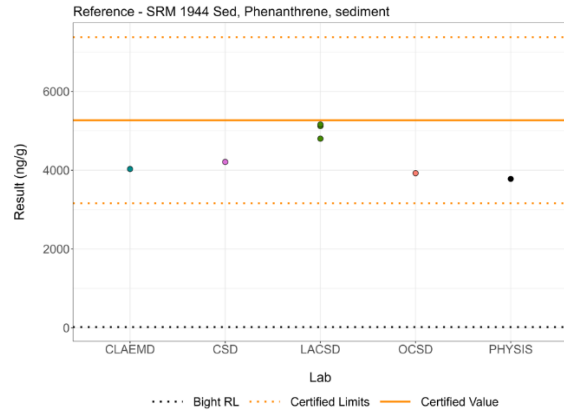




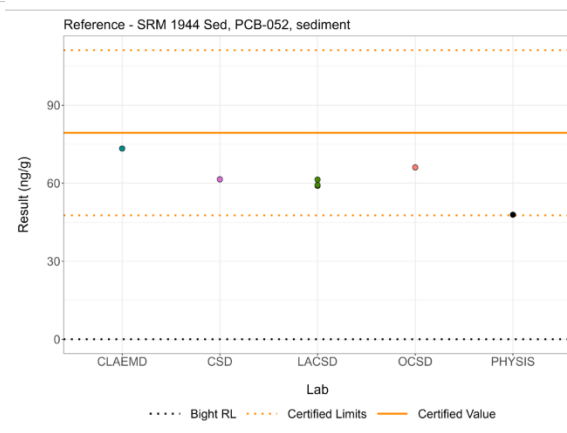
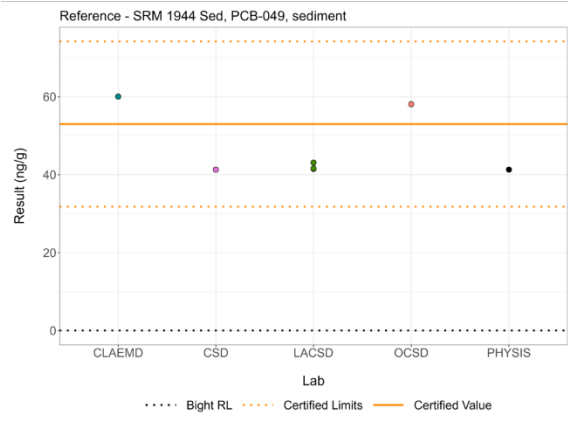
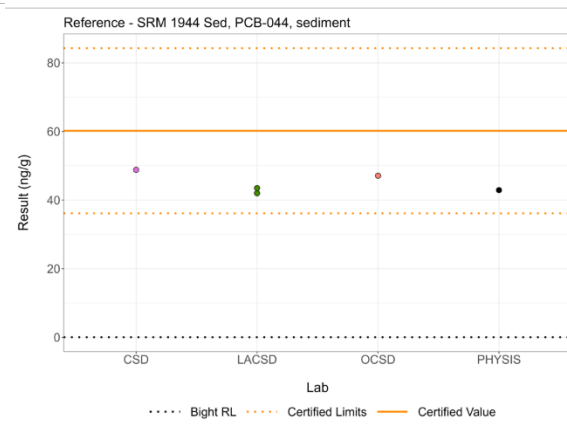
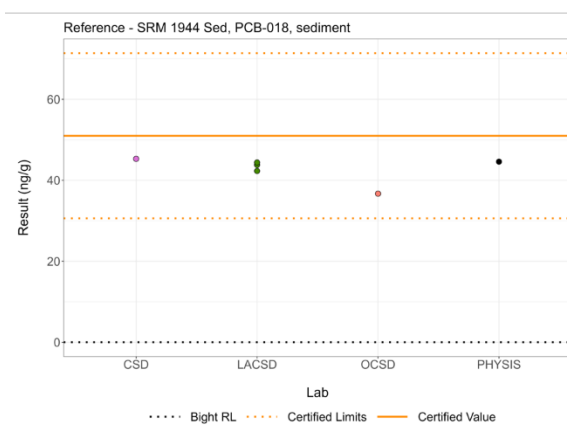
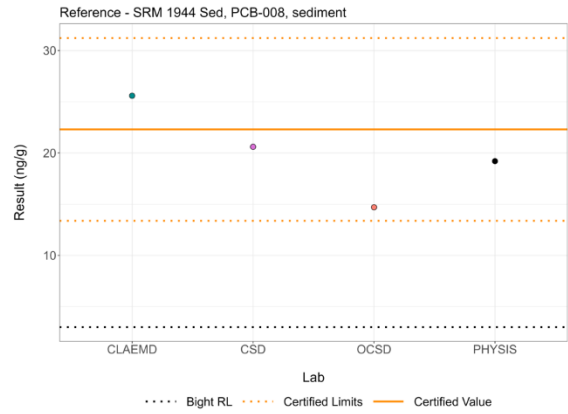
CRM PAHs: SRM 1944

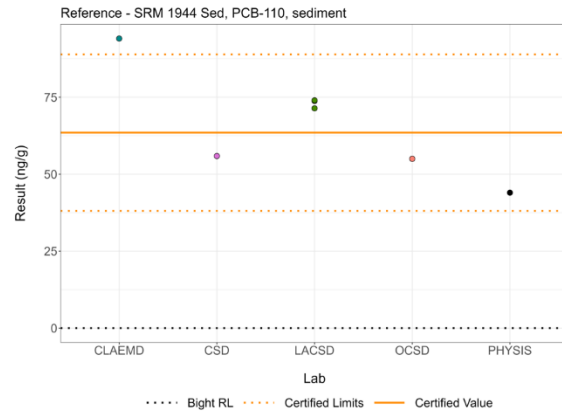
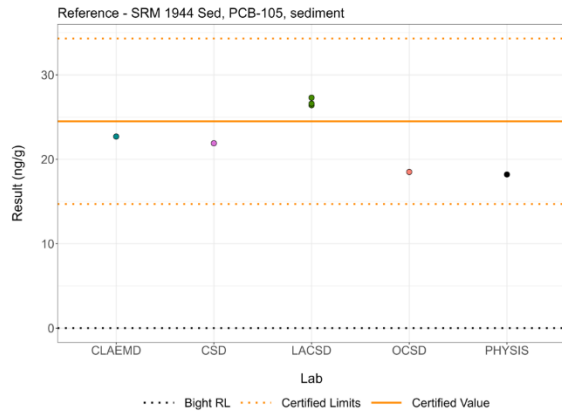
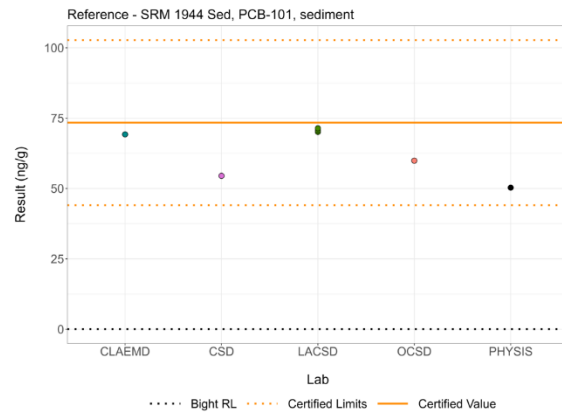
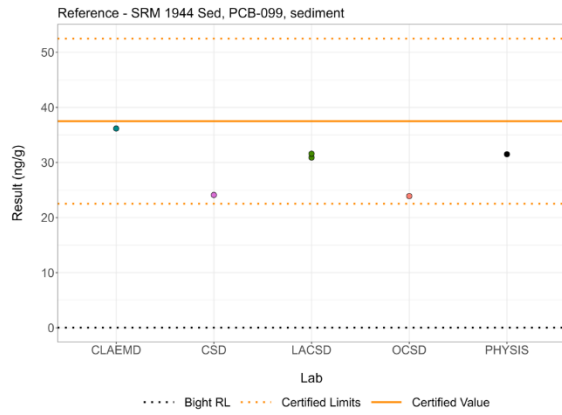
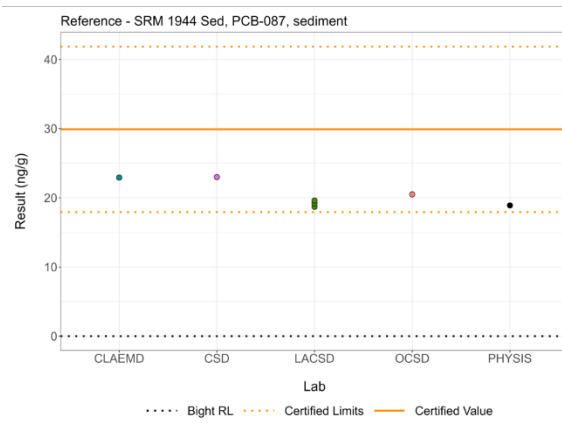
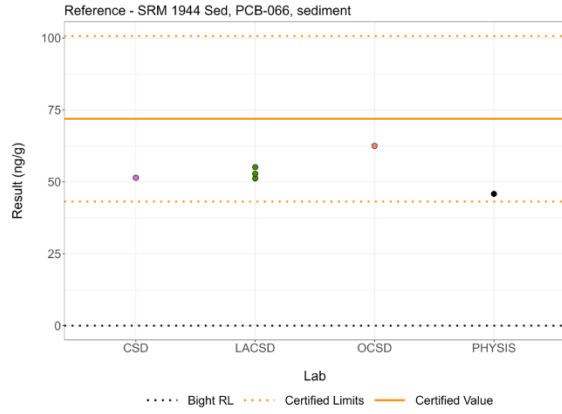


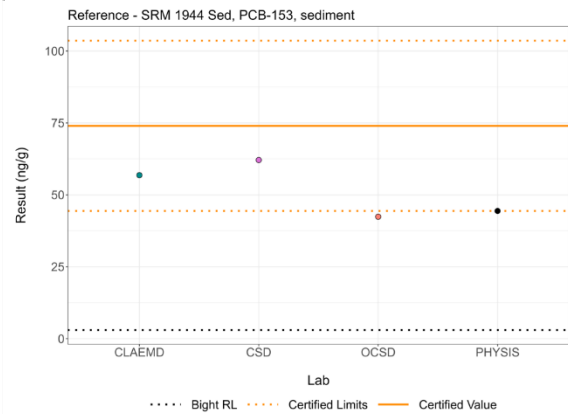
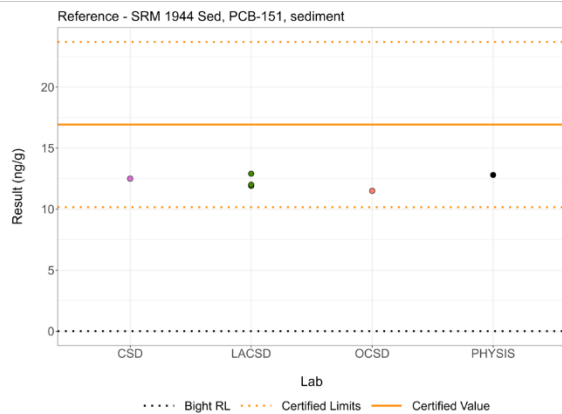
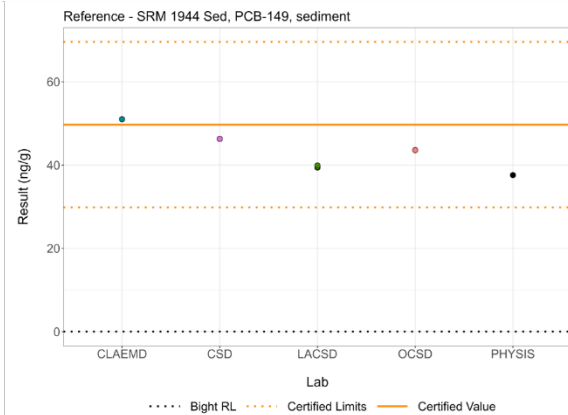
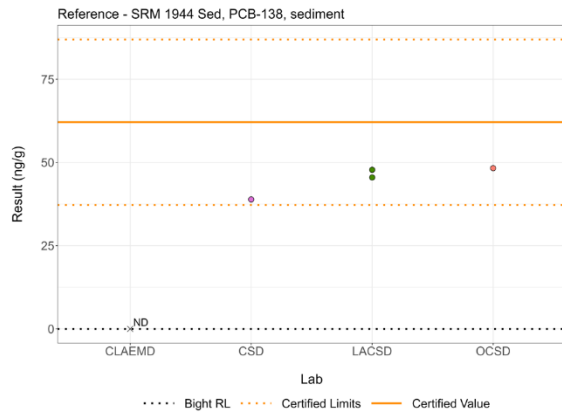
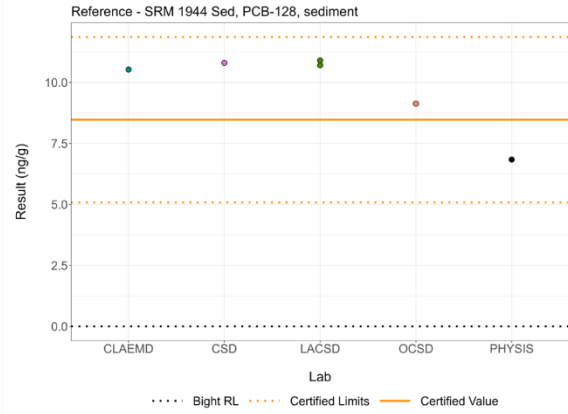
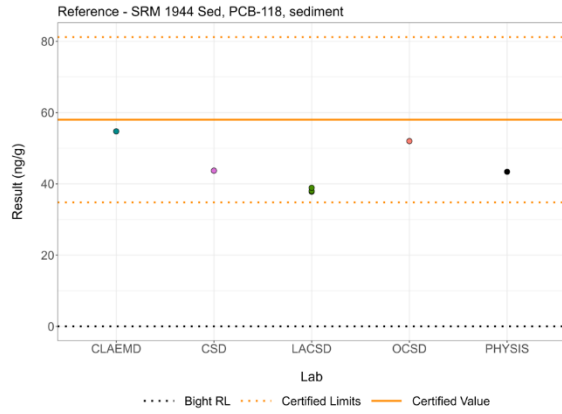


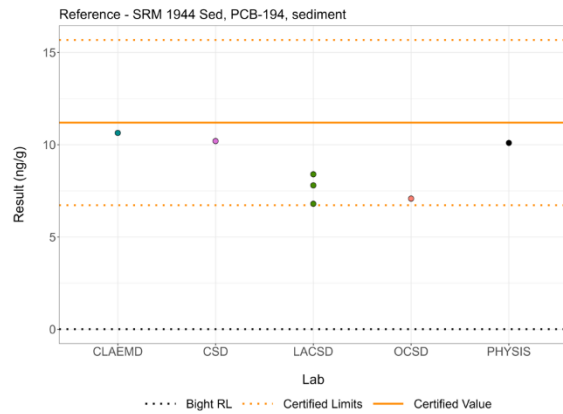
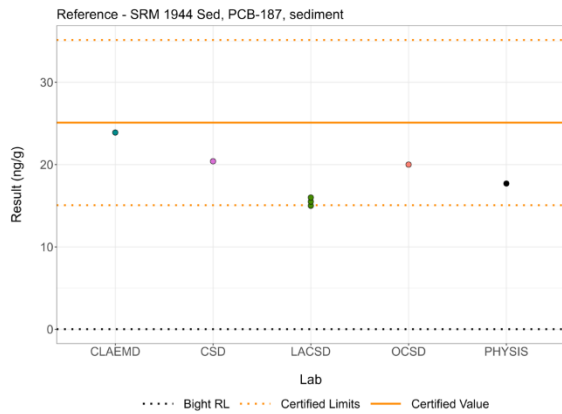
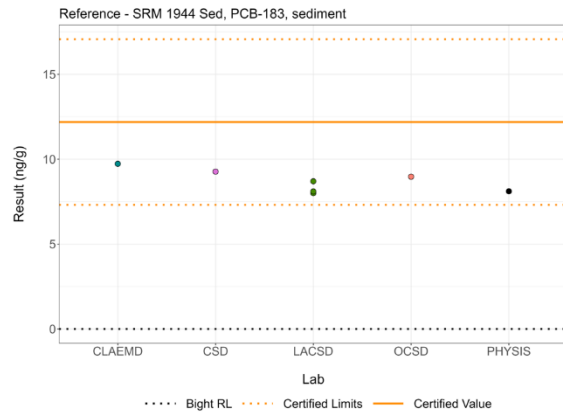
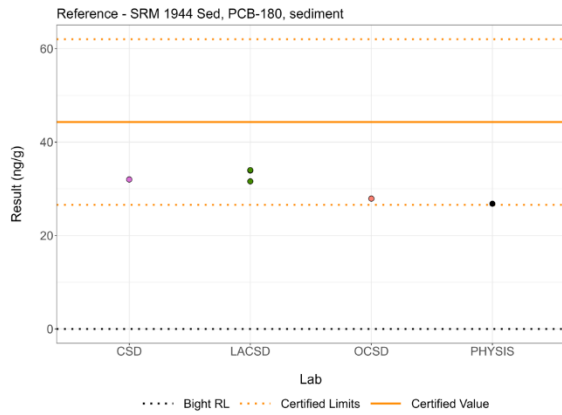
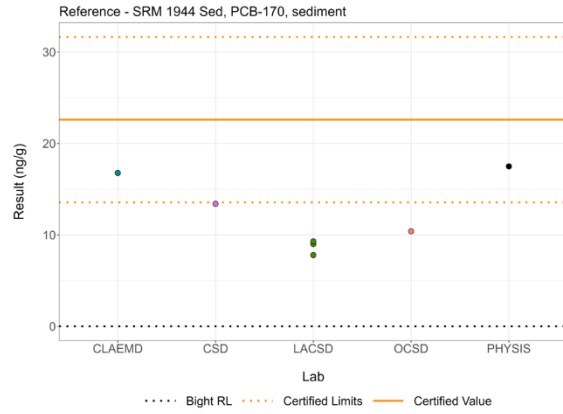
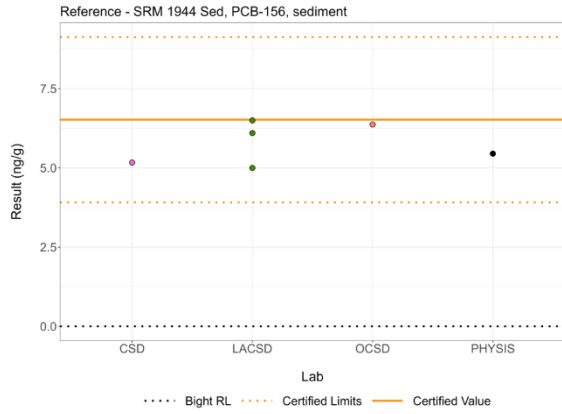


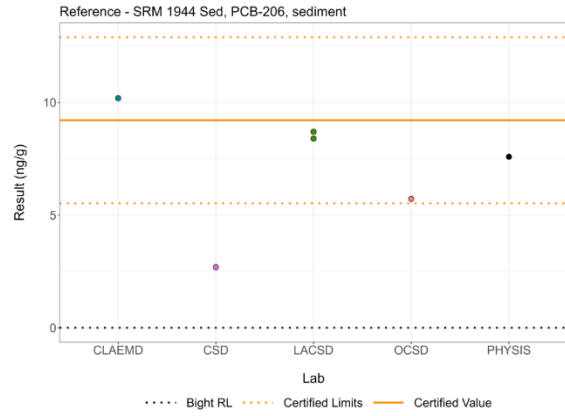
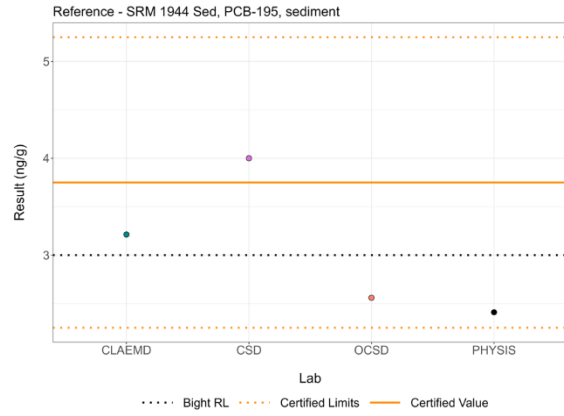
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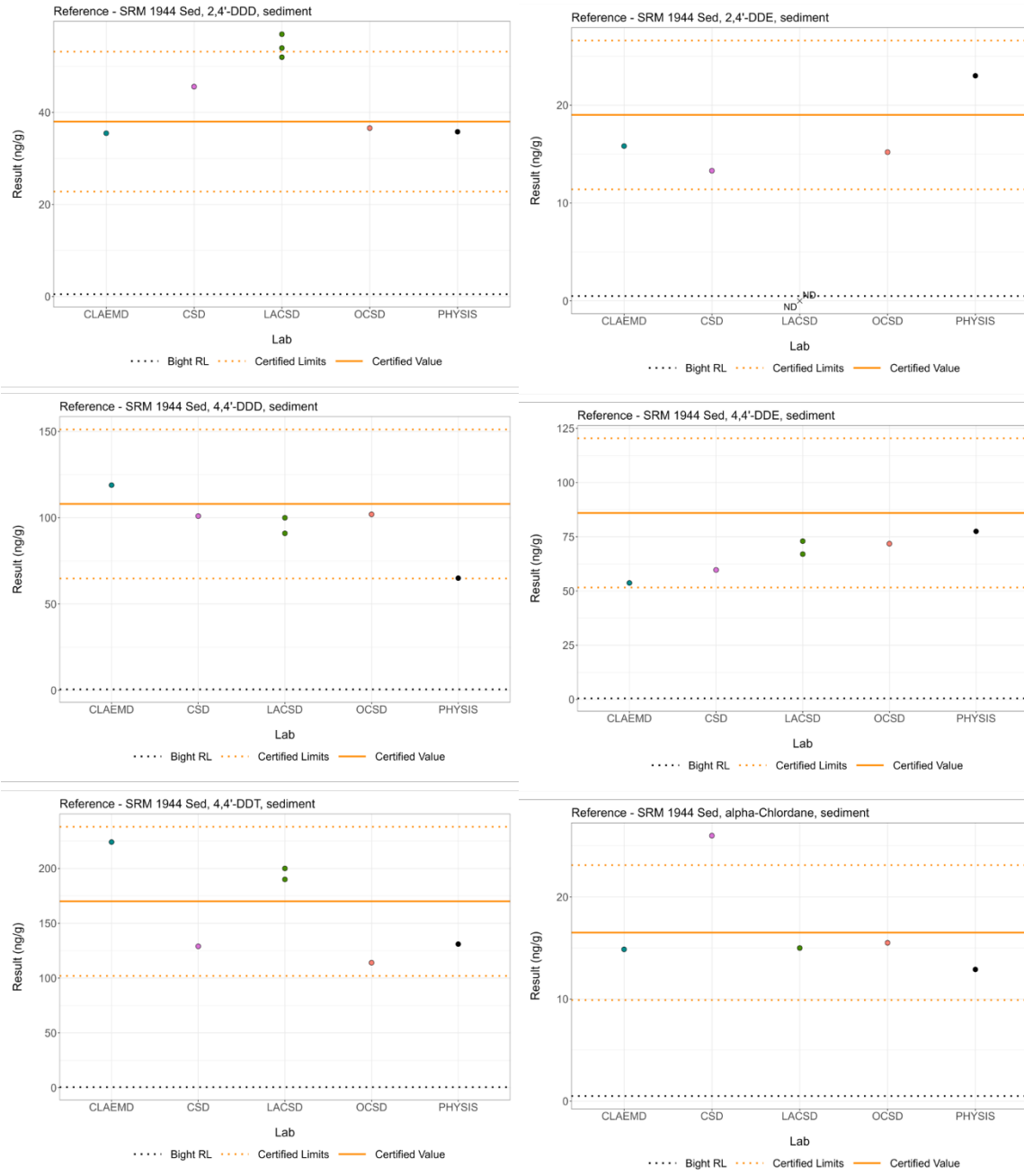


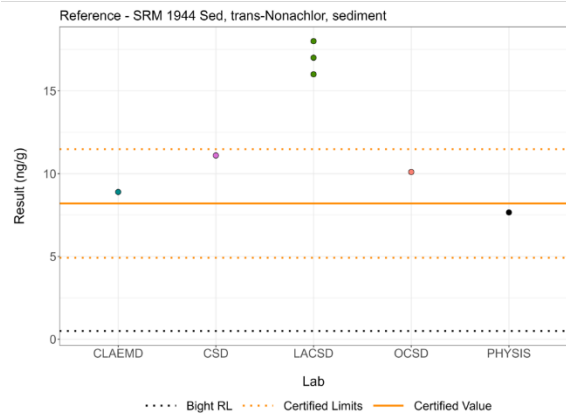
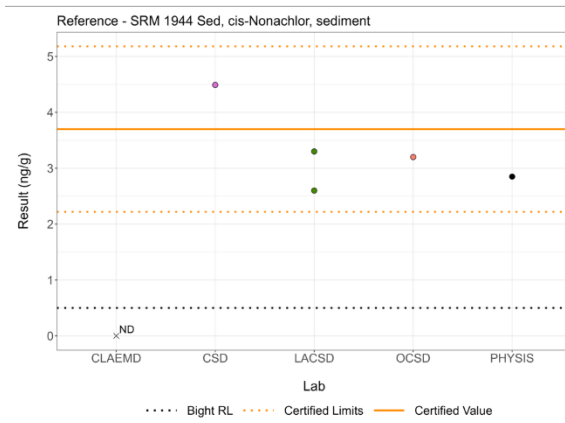
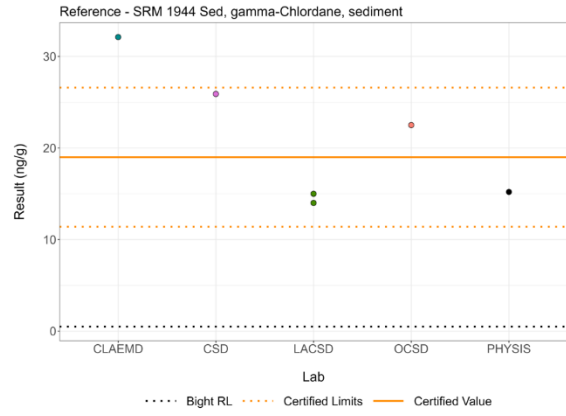




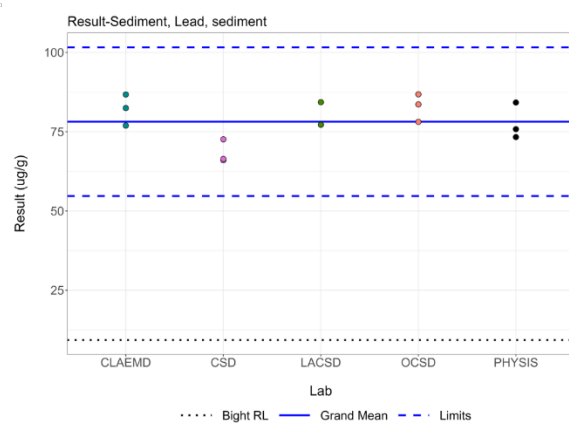
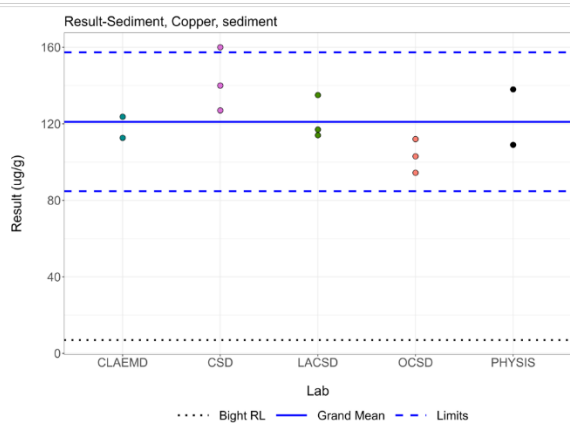
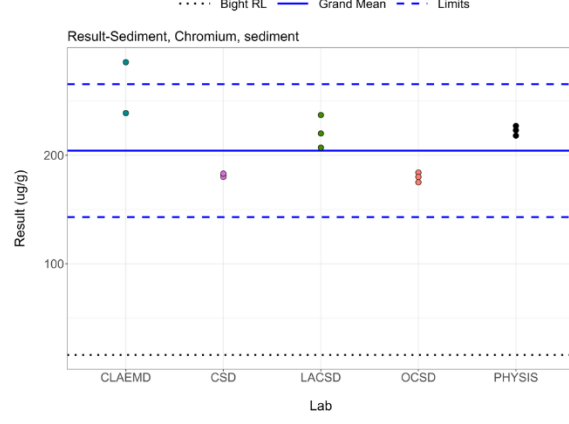
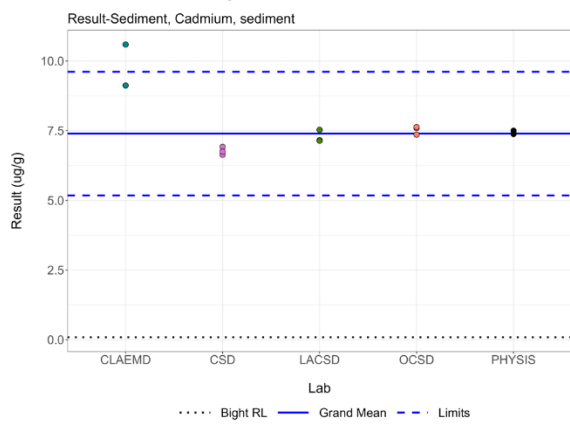
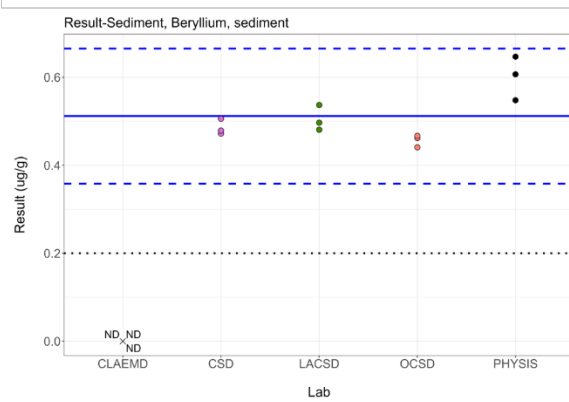
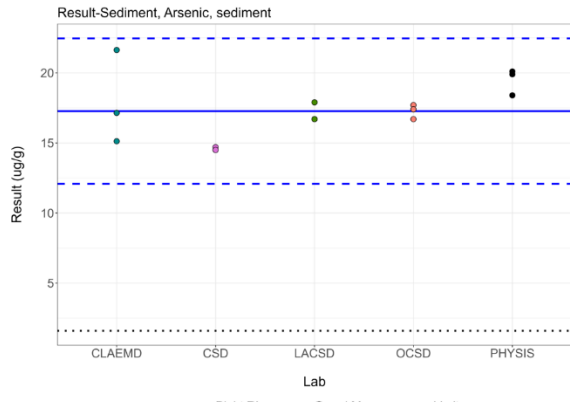


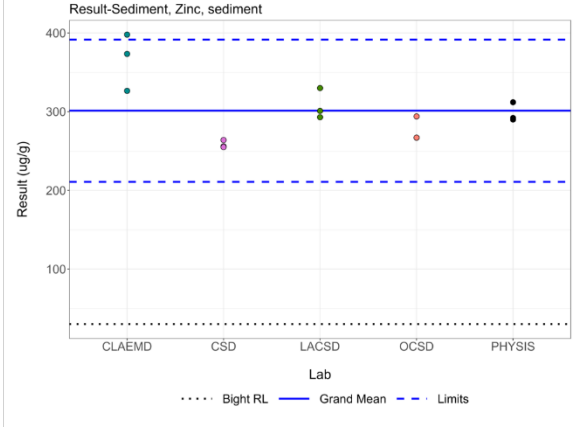
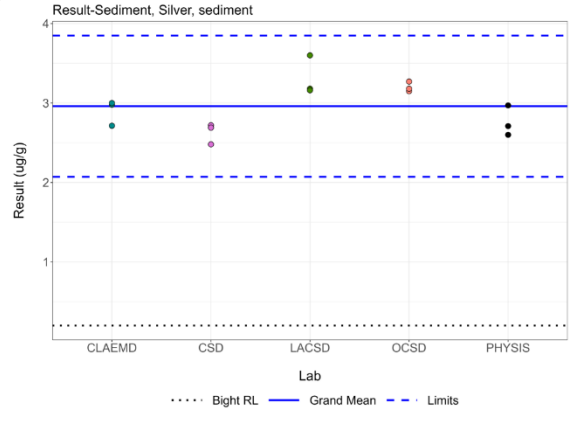
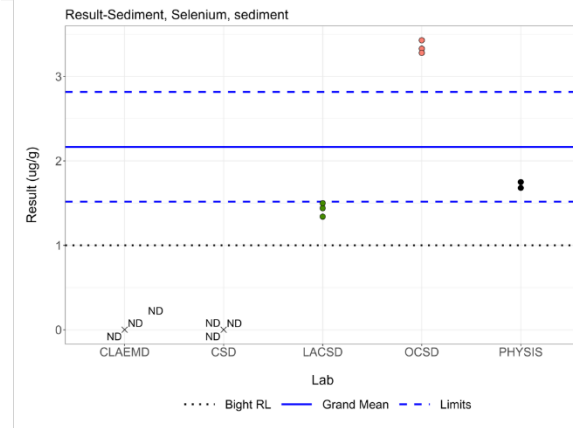
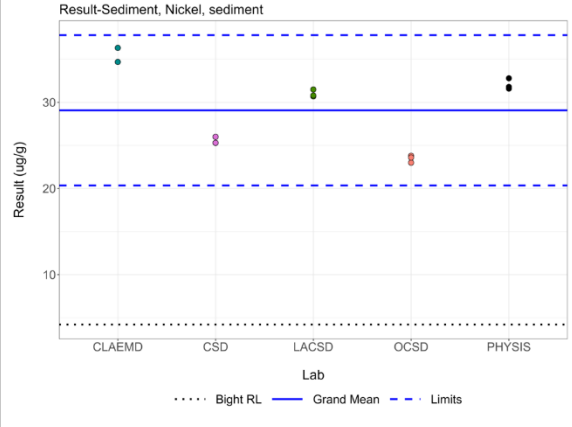
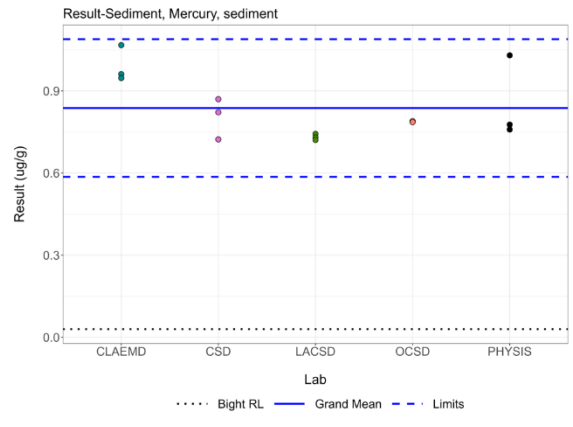
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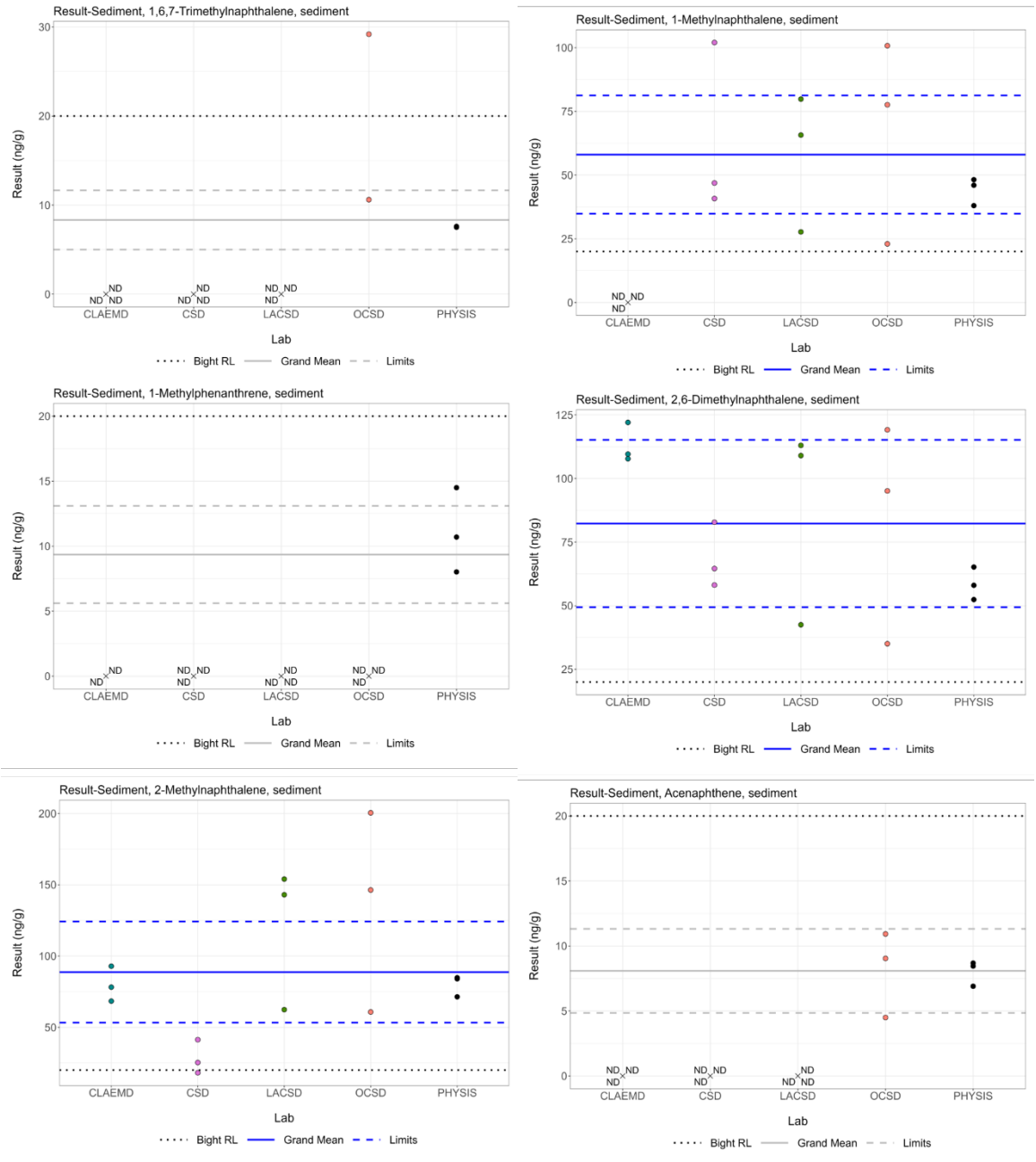


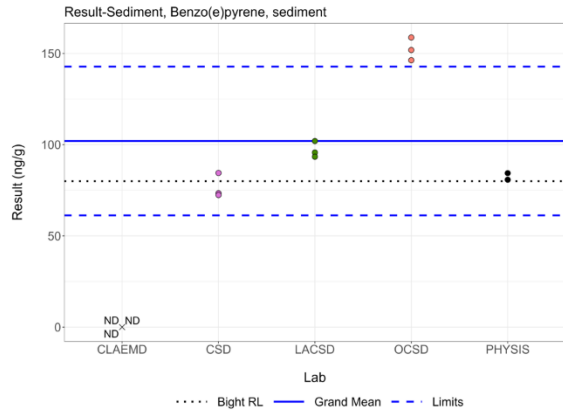
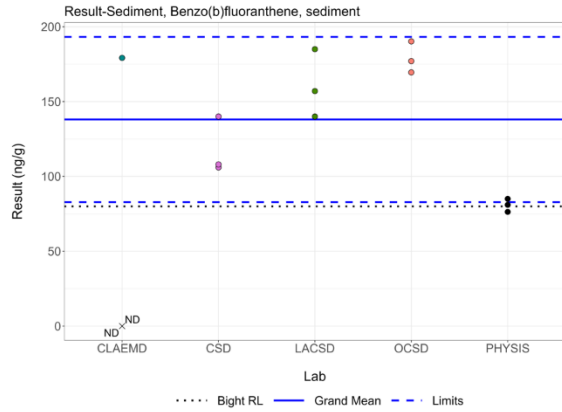
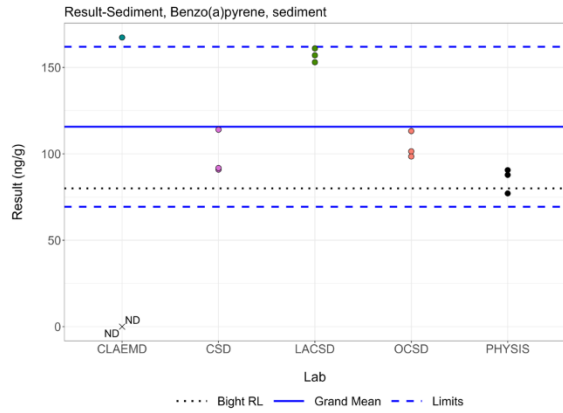
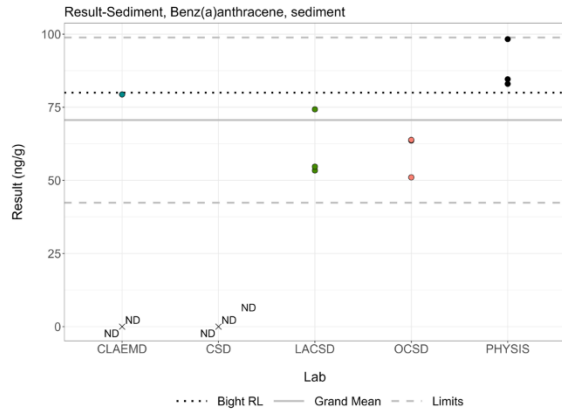
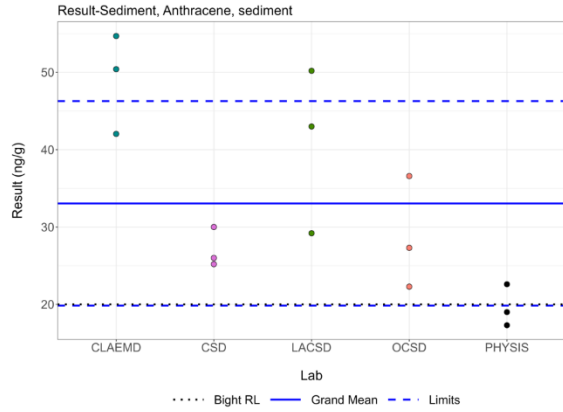
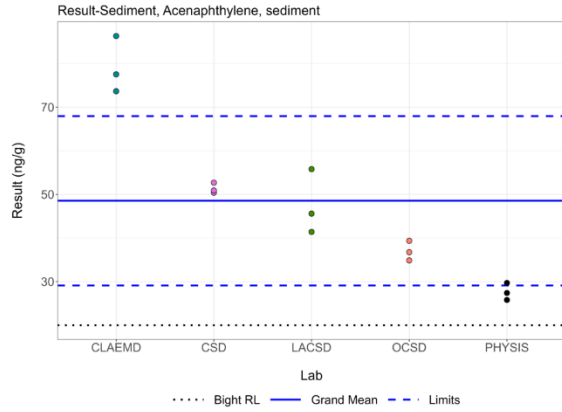
FRM metals

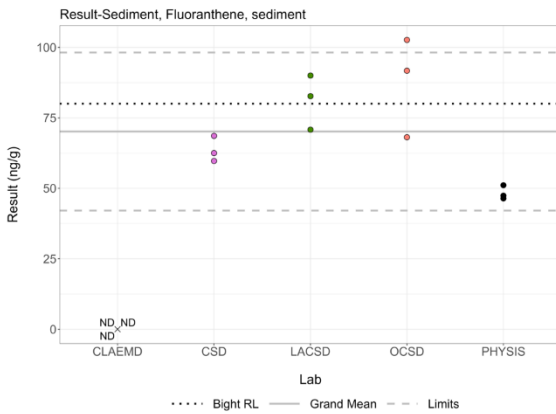
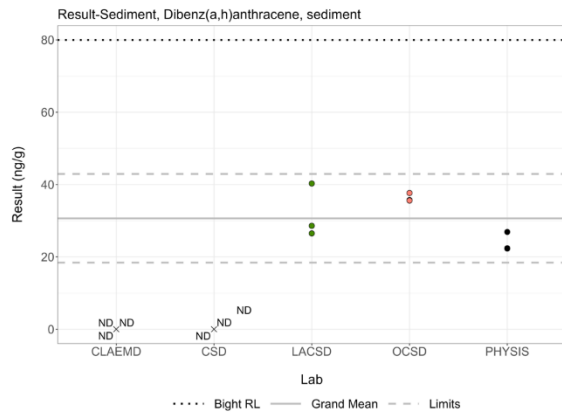
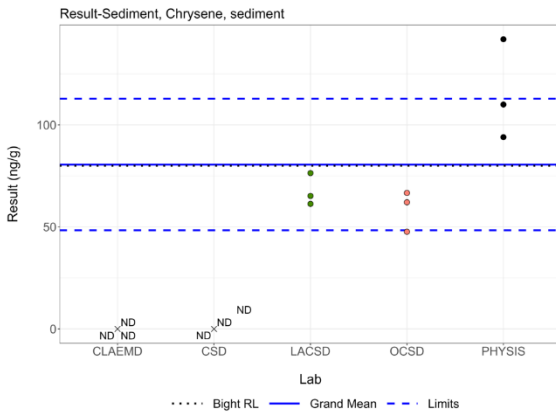
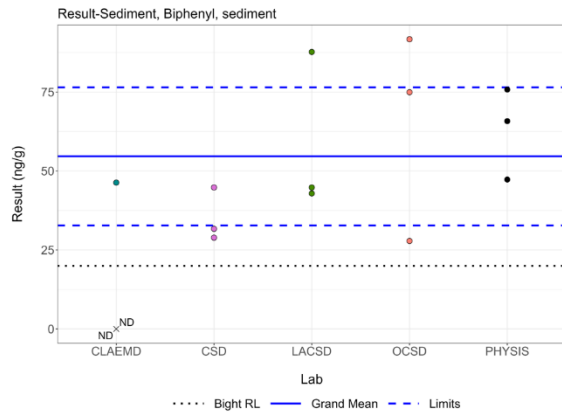
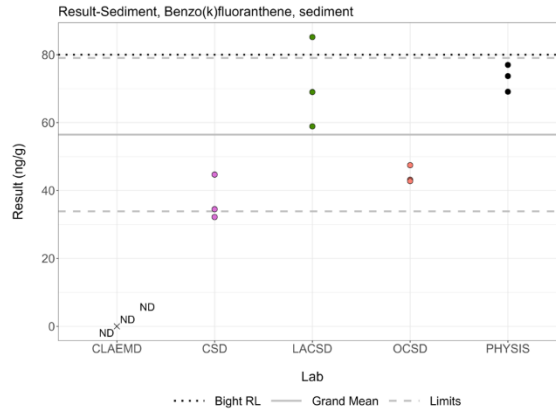
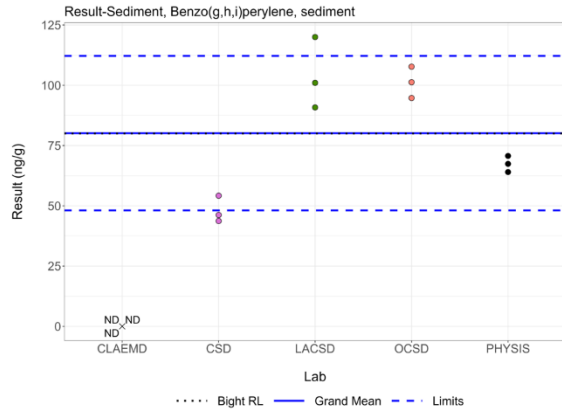


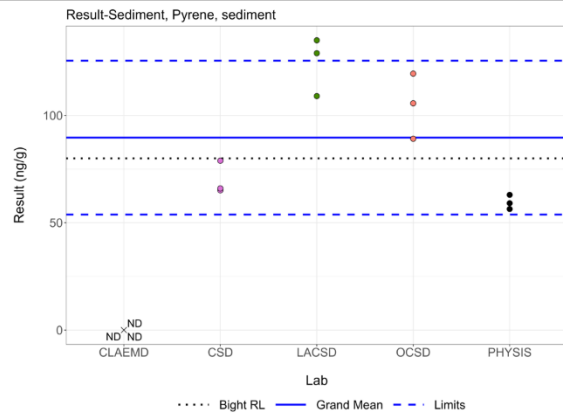
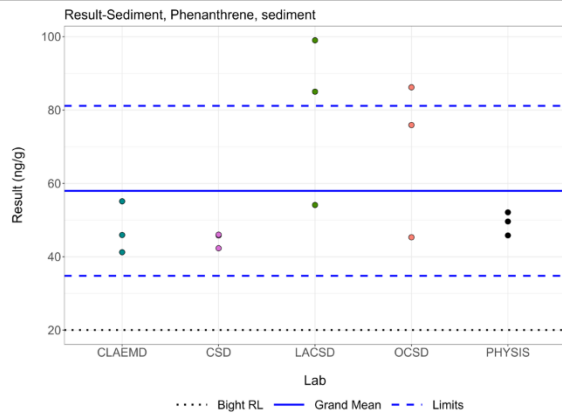
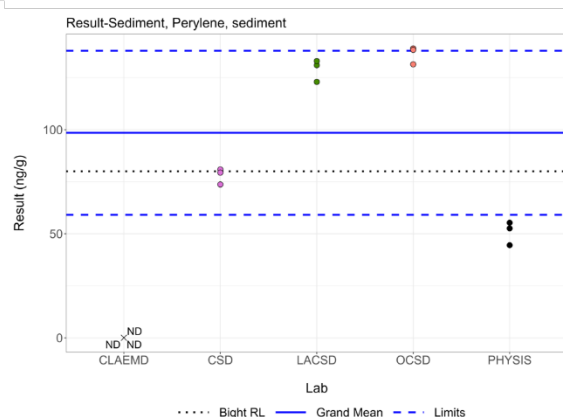
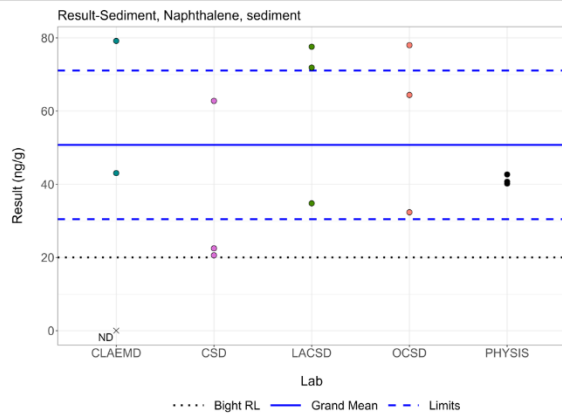
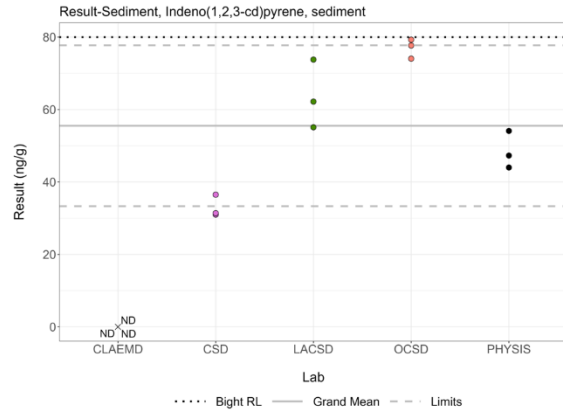
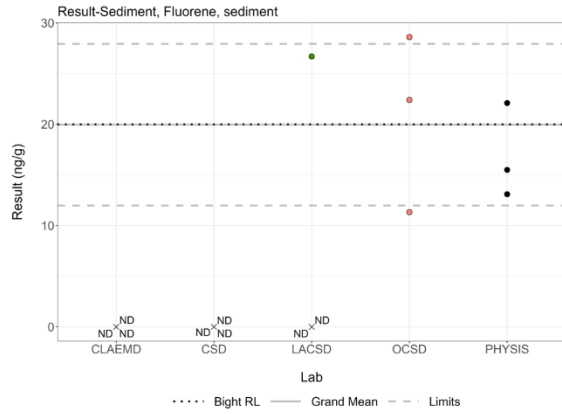


FRM PAHs

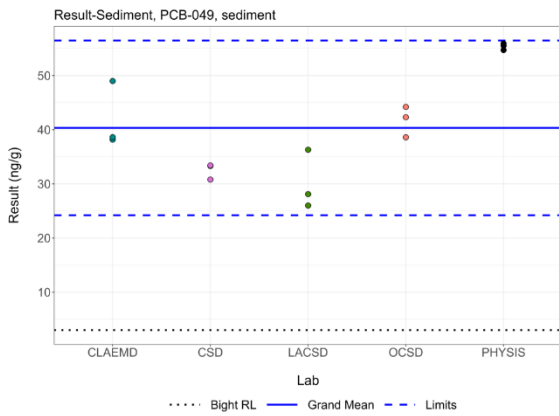
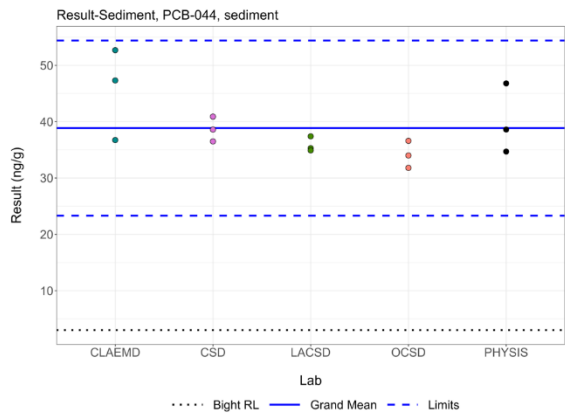
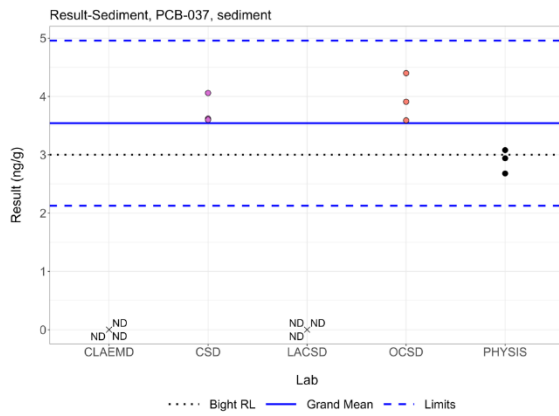
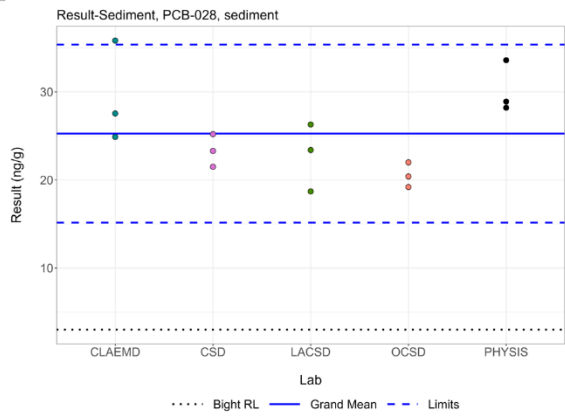
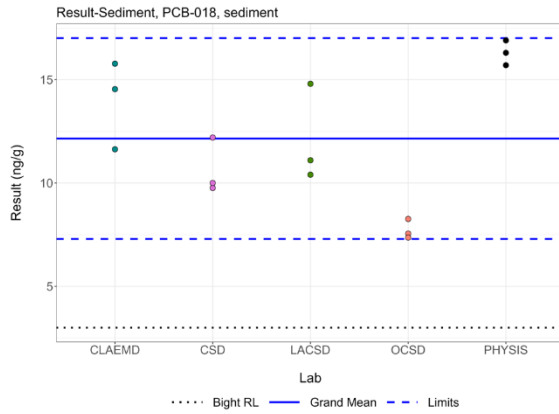
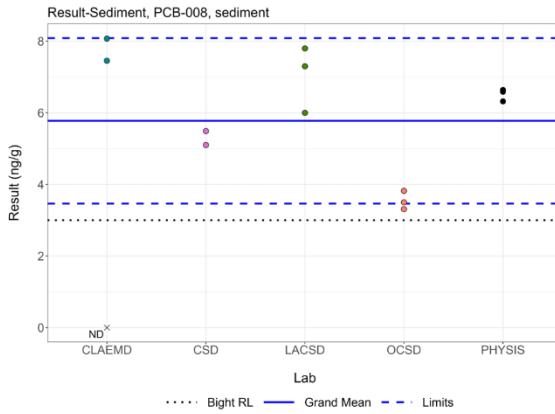


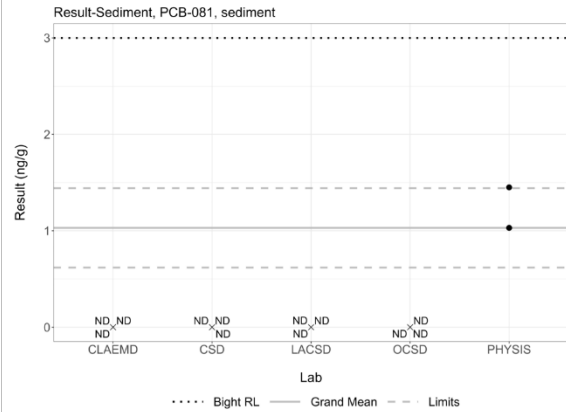
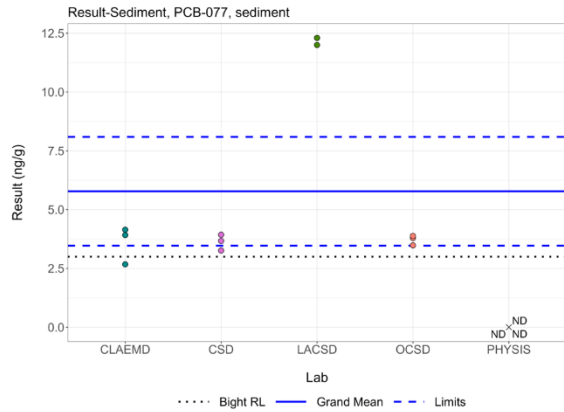
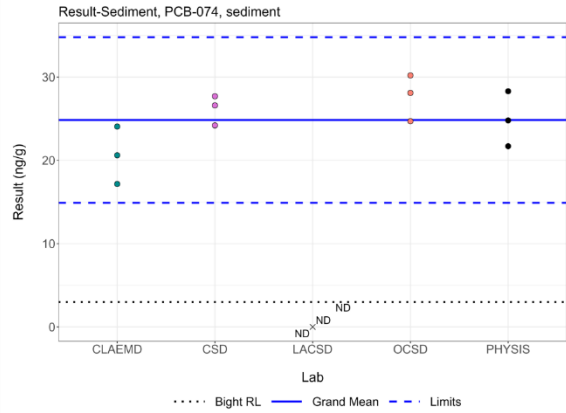
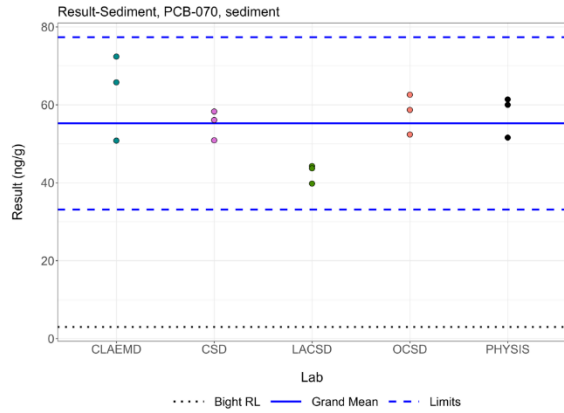
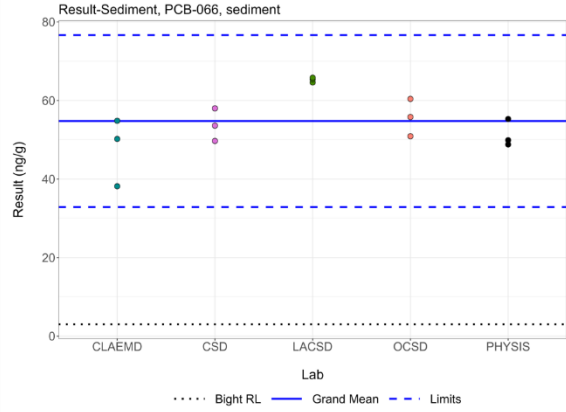
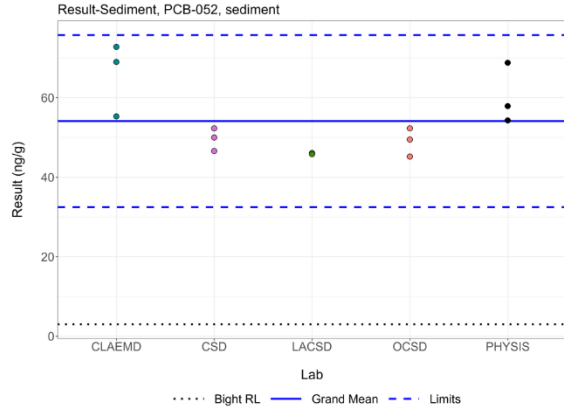


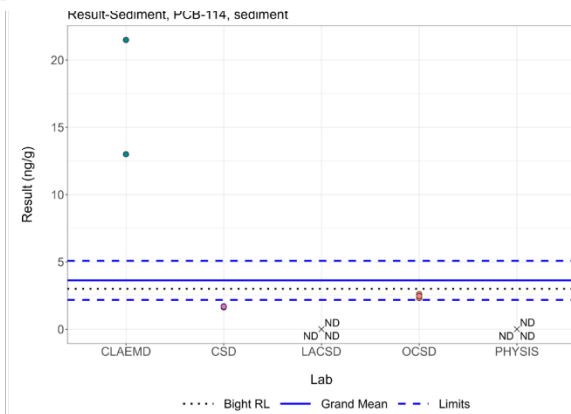
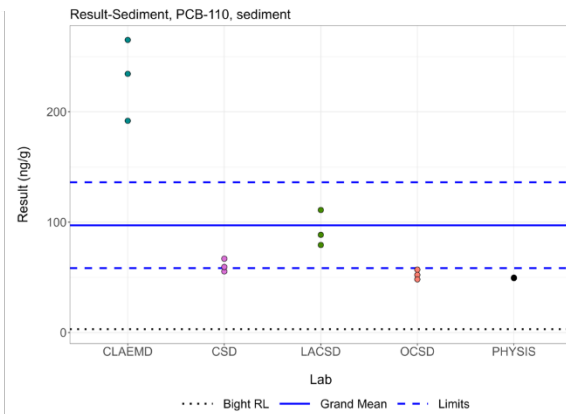
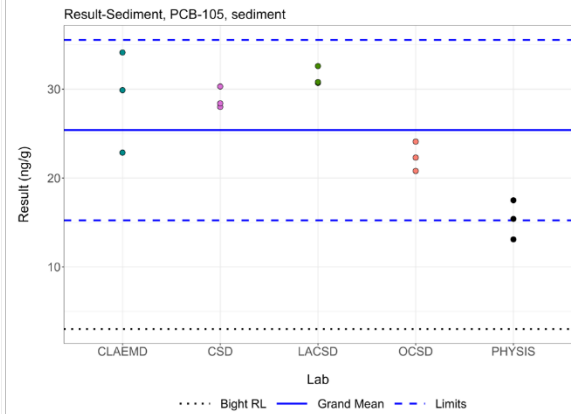
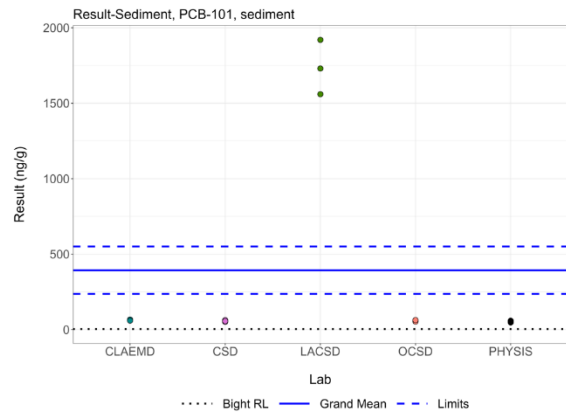
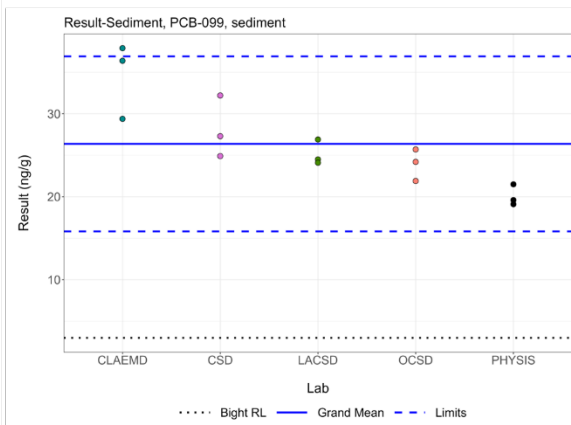
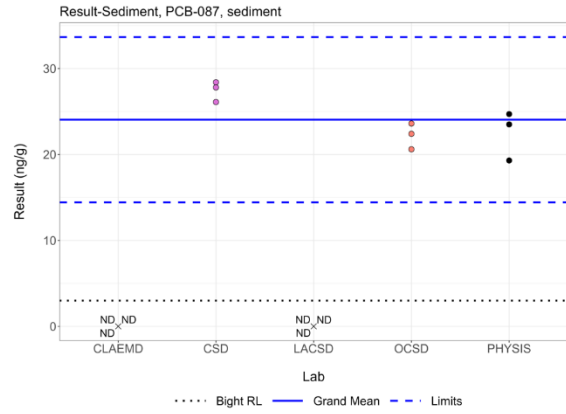


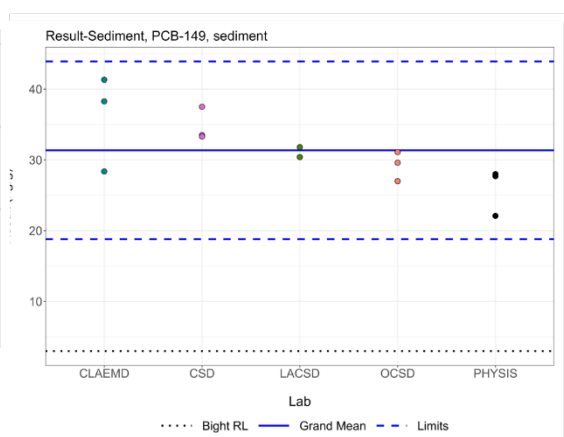
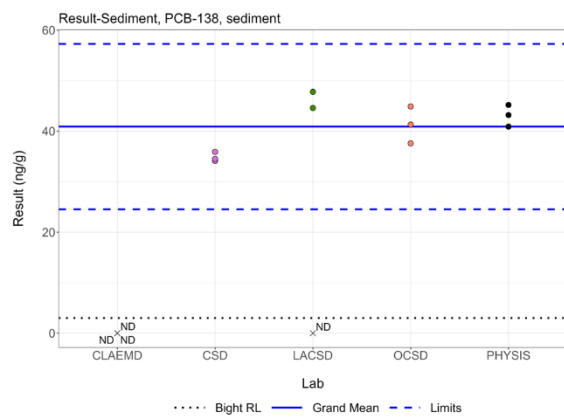
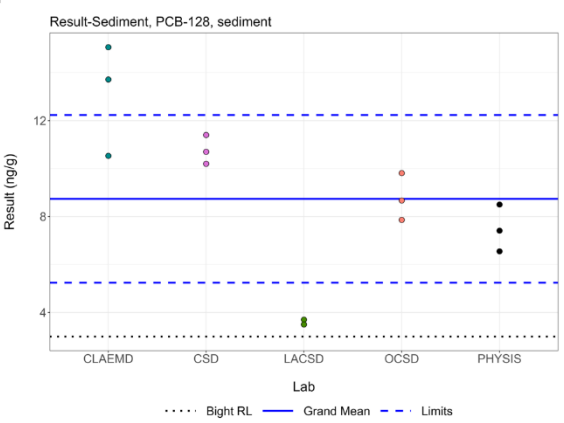
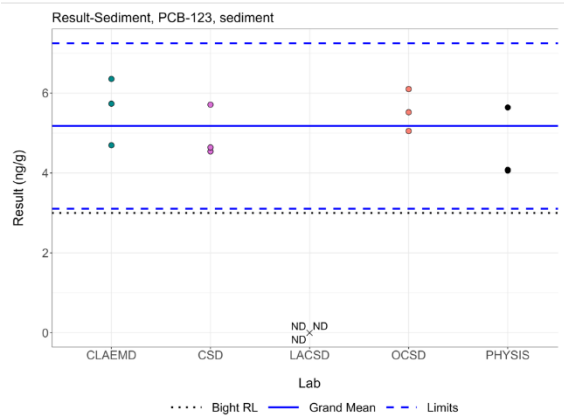
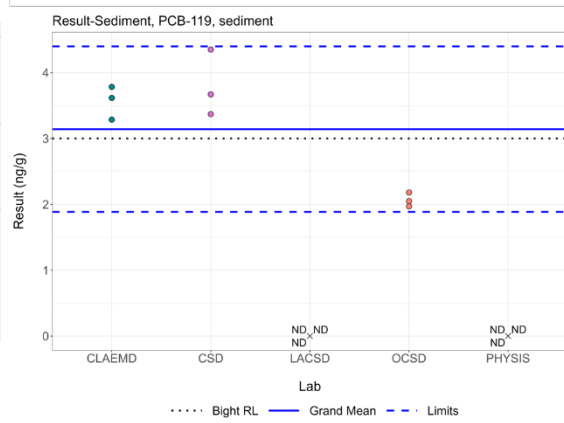
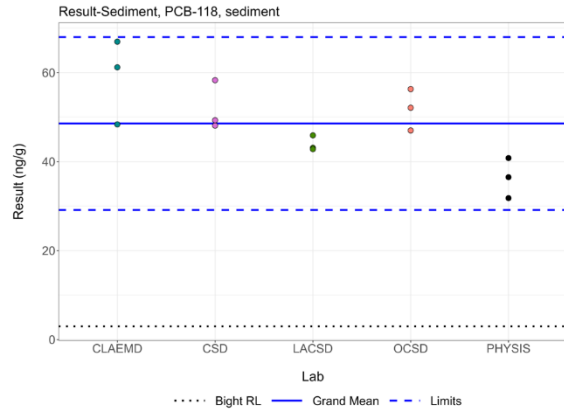


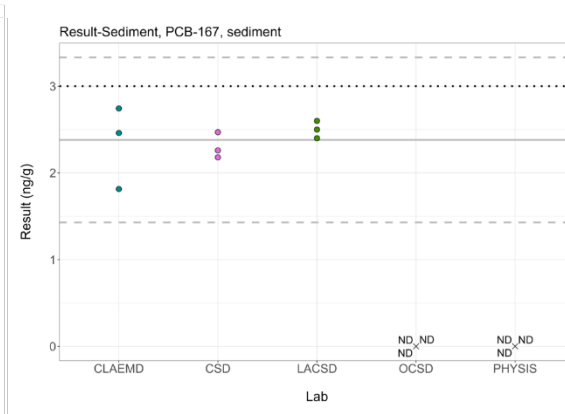
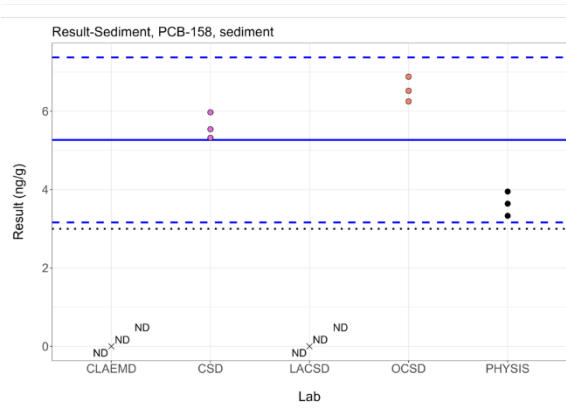
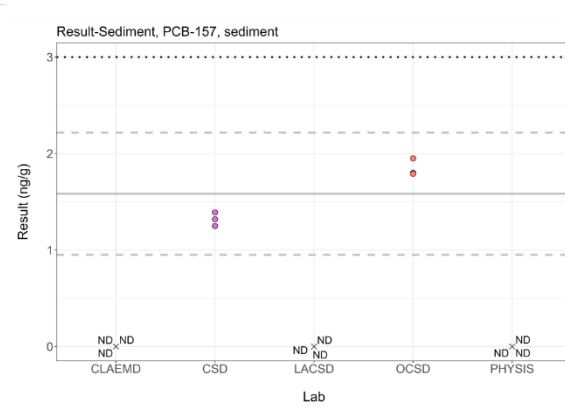
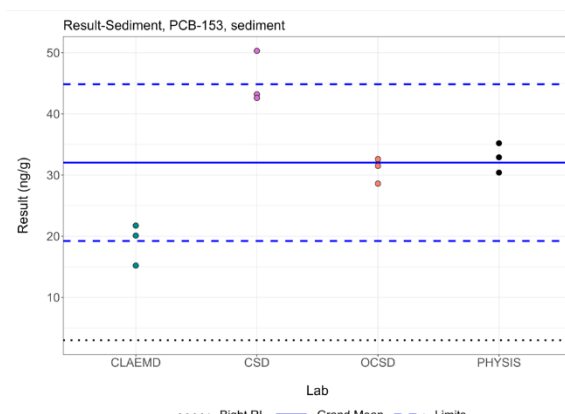
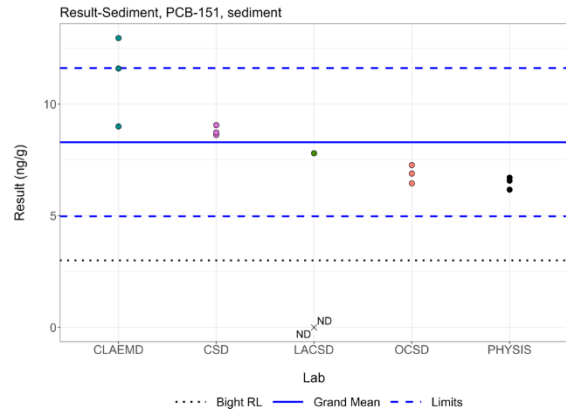
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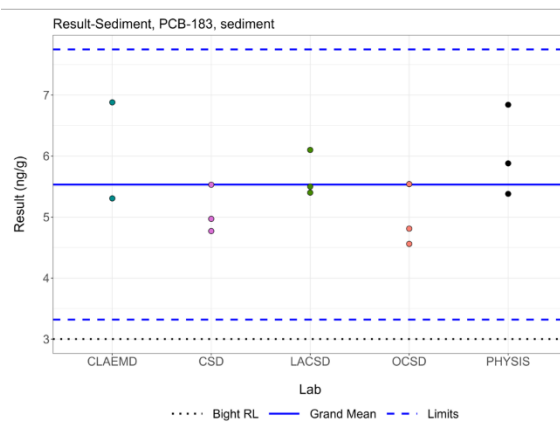
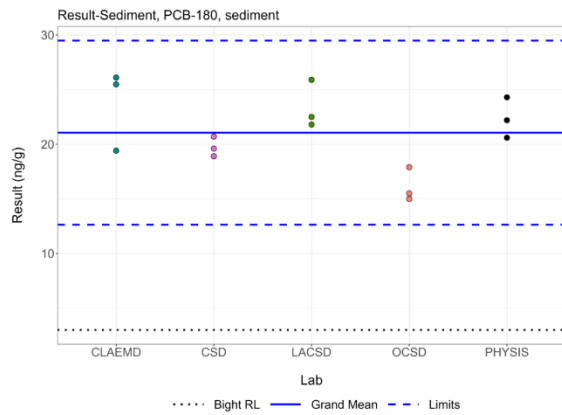
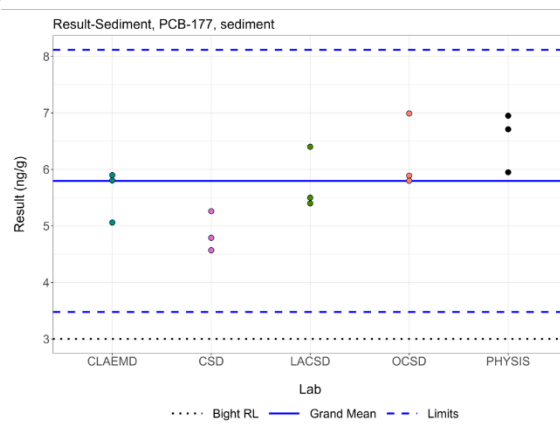
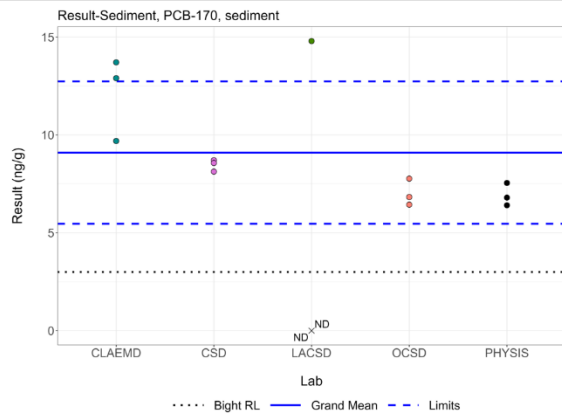
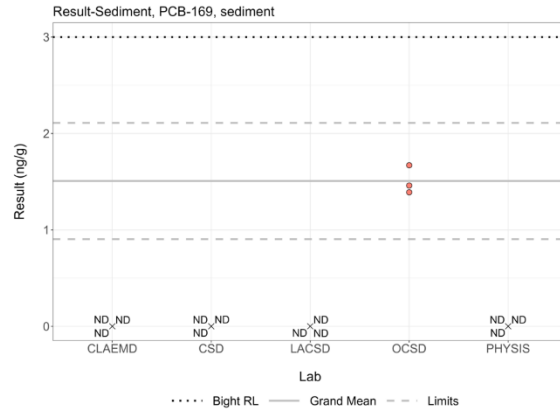
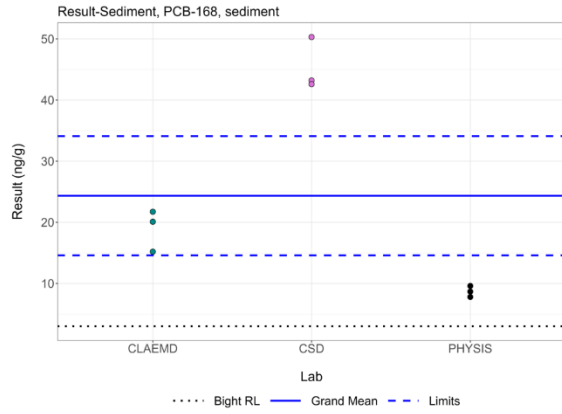


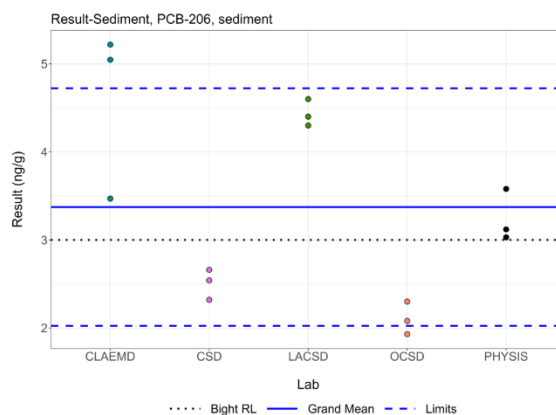
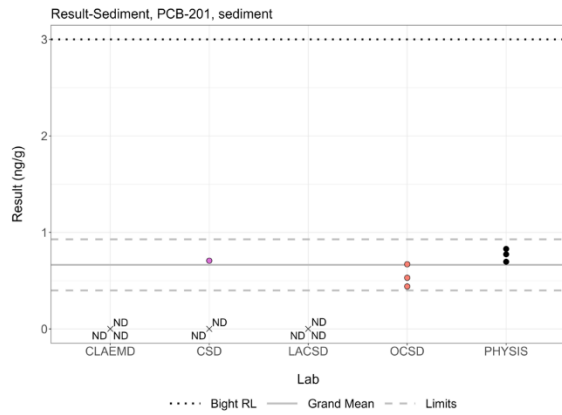
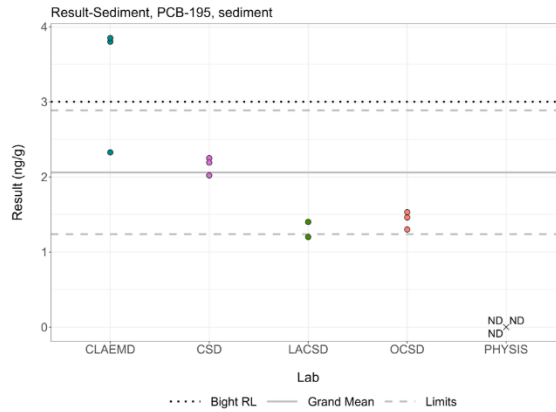
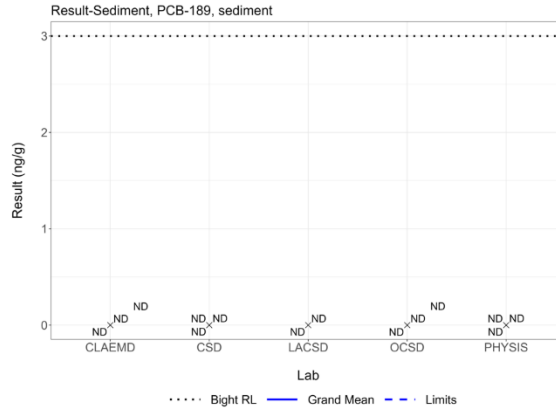
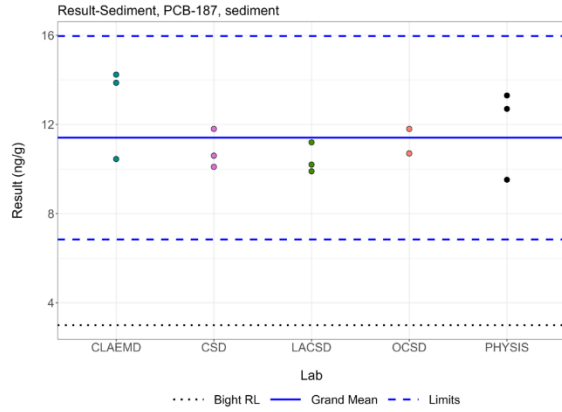




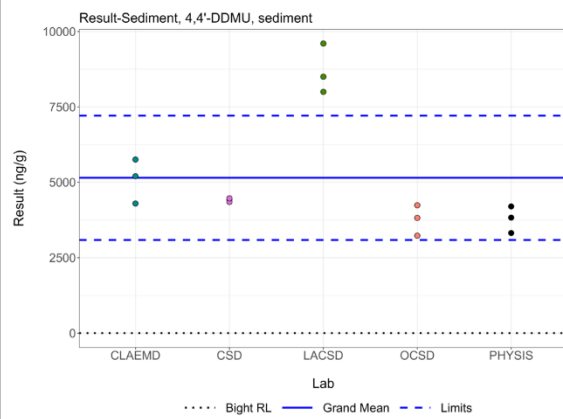
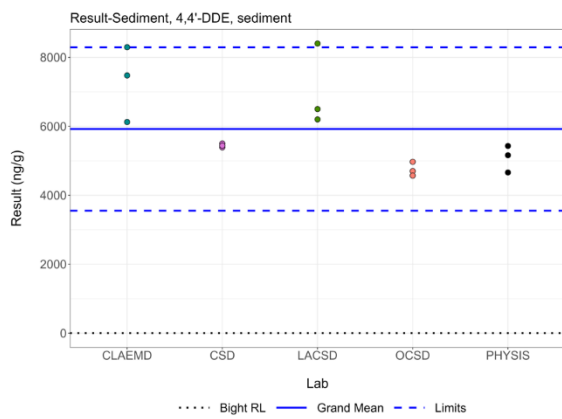
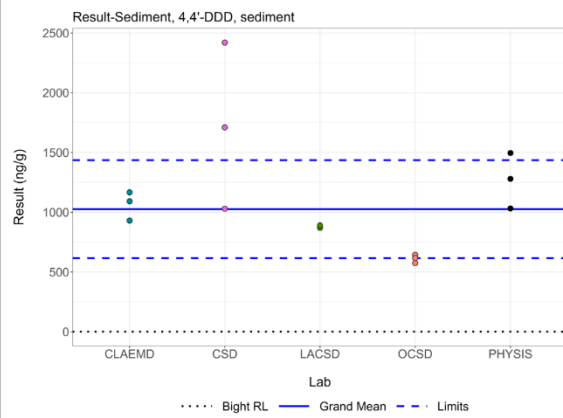
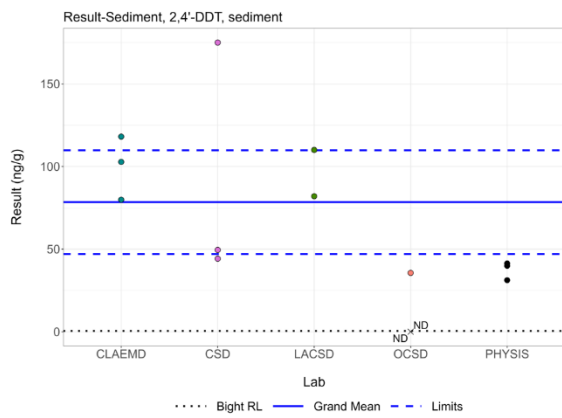
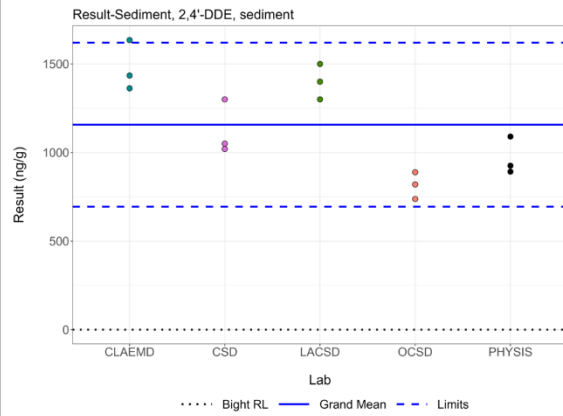
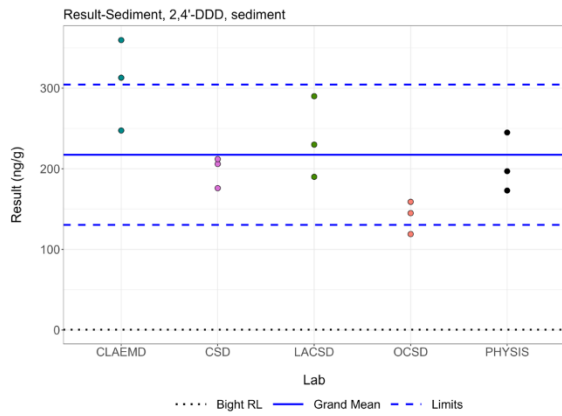


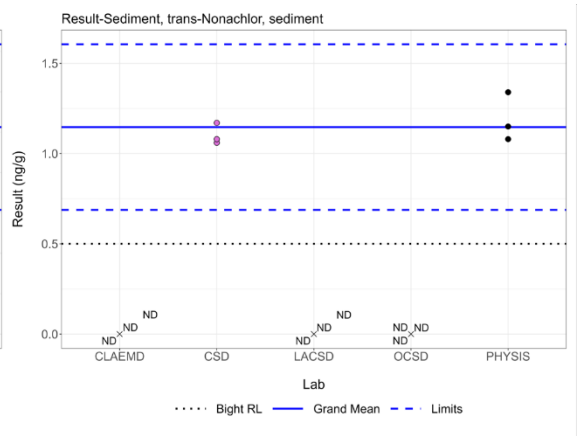
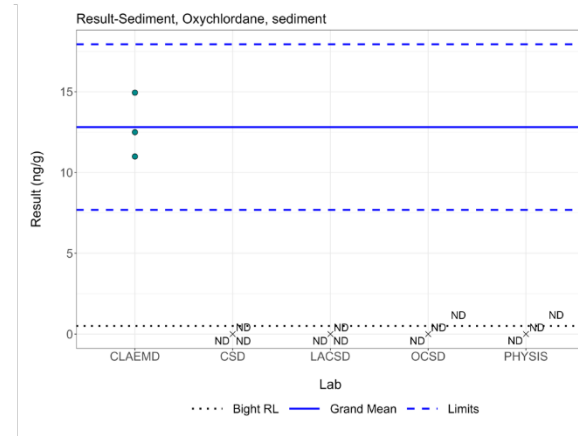
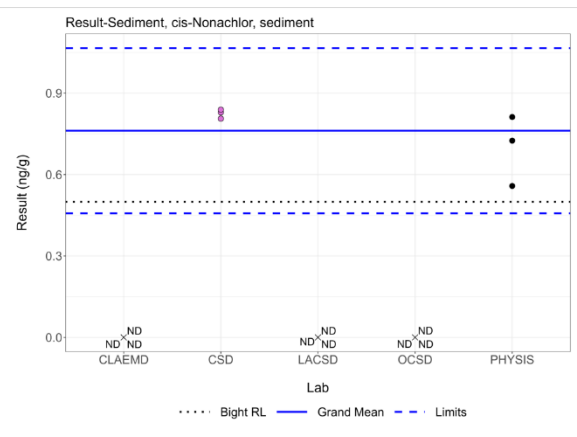
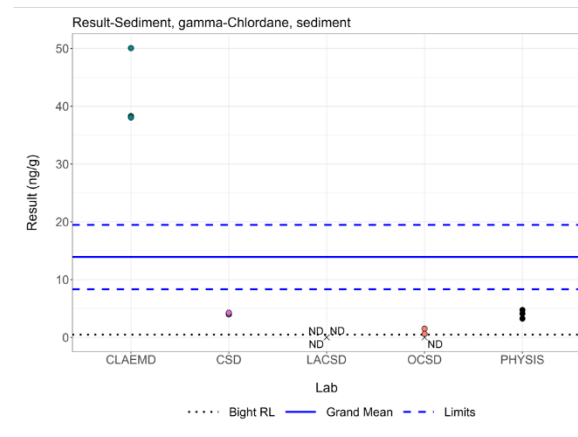
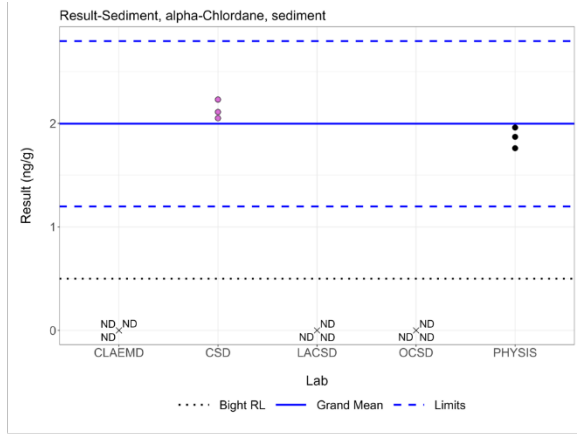
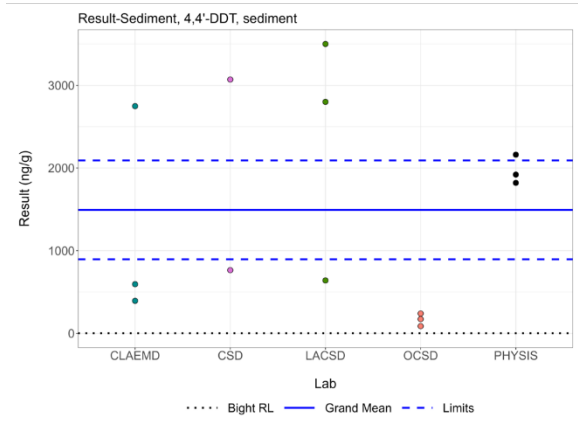






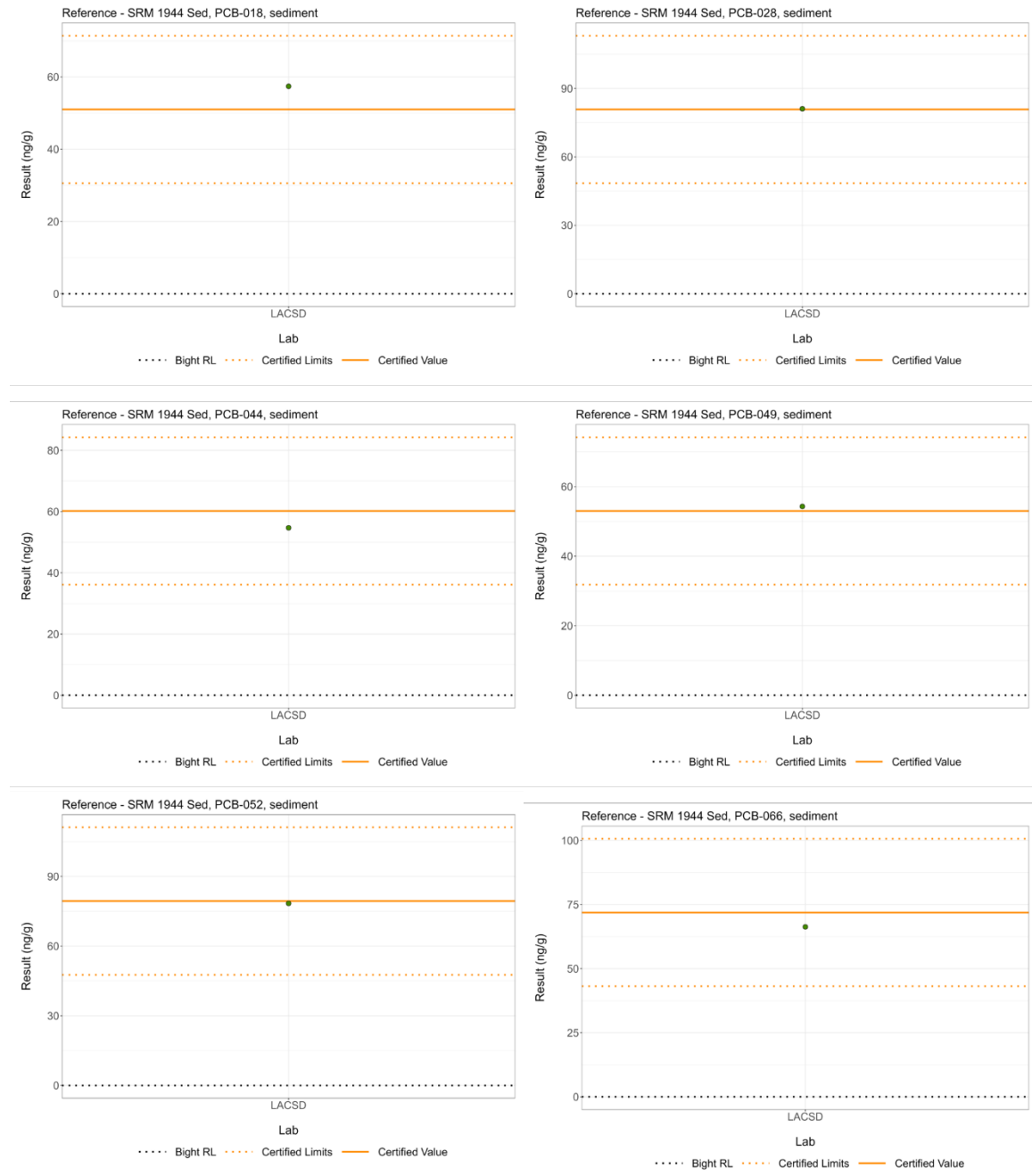
FRM OCs

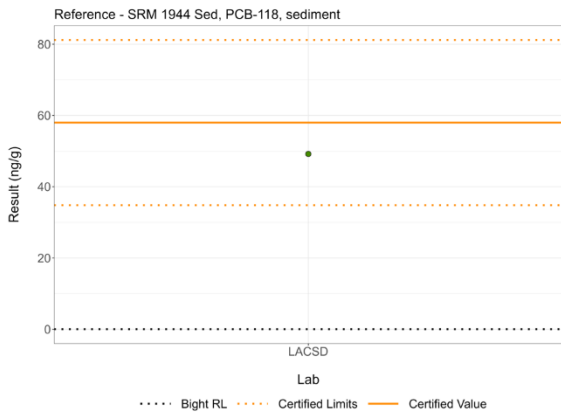
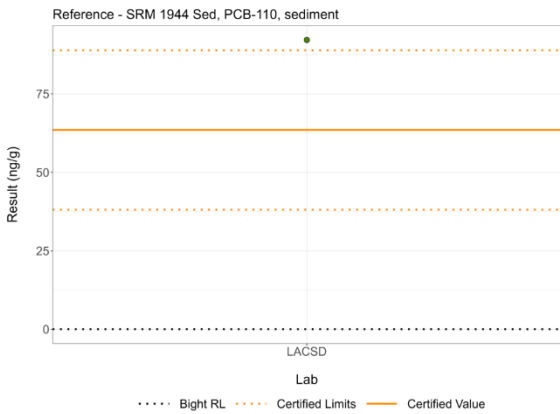
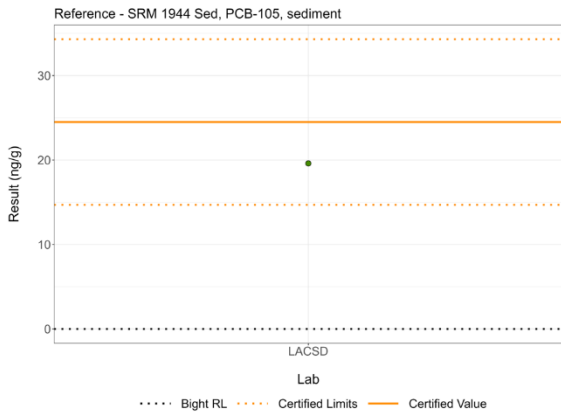
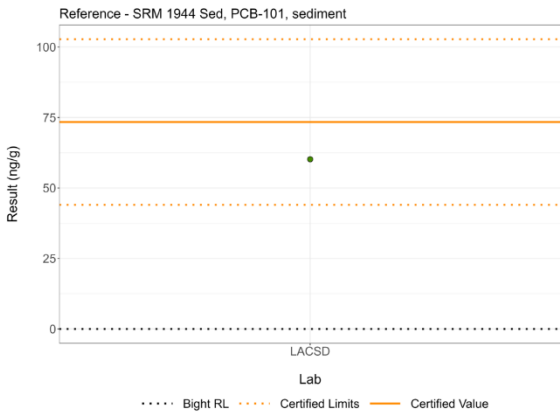
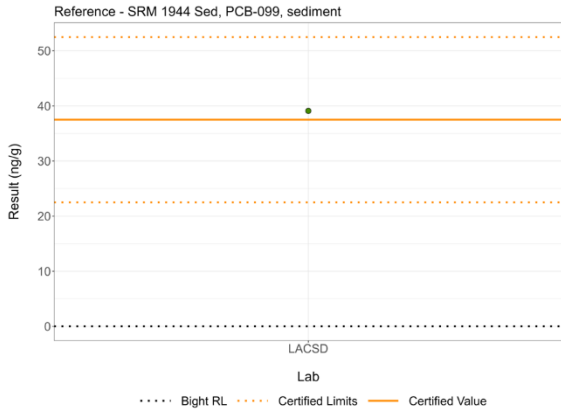
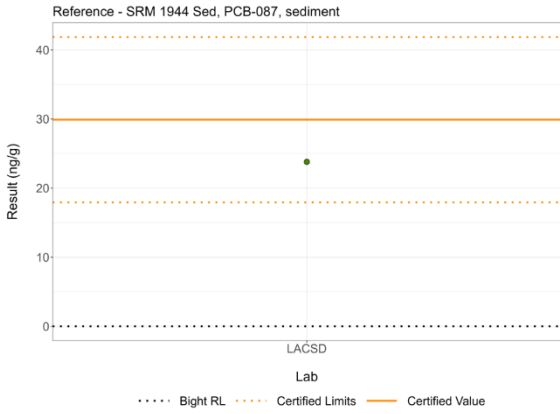


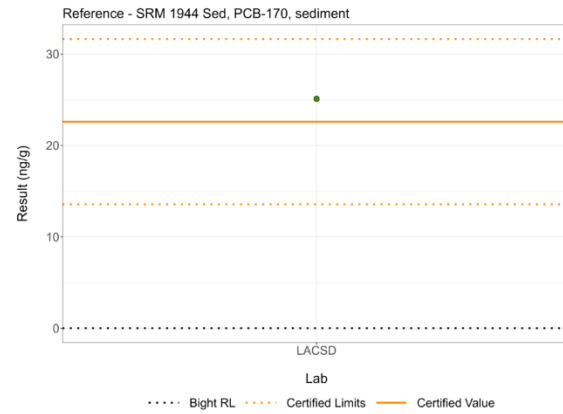
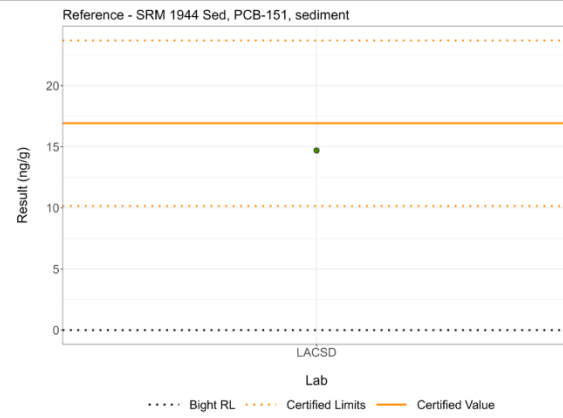
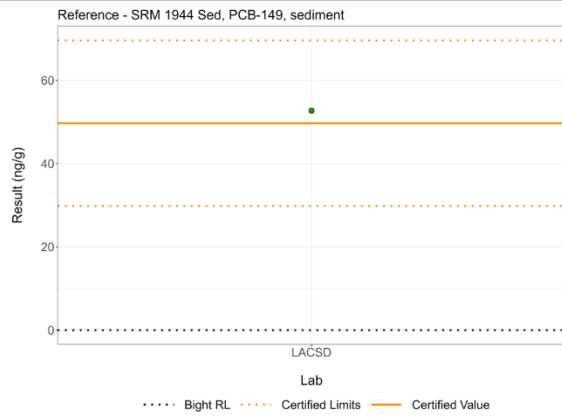
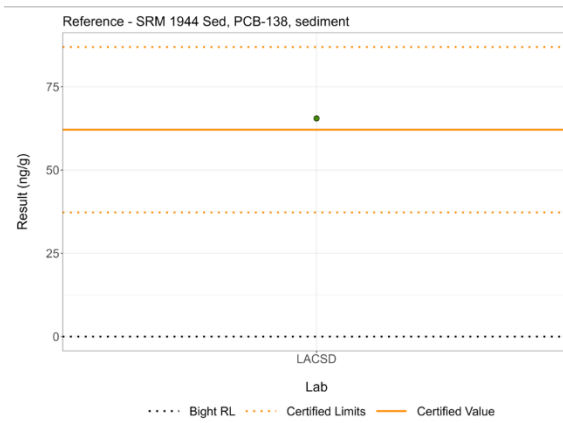
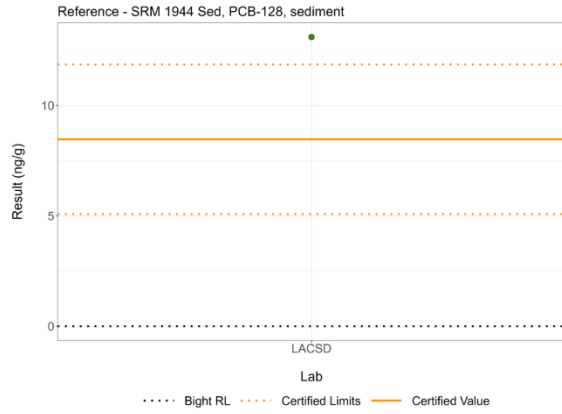


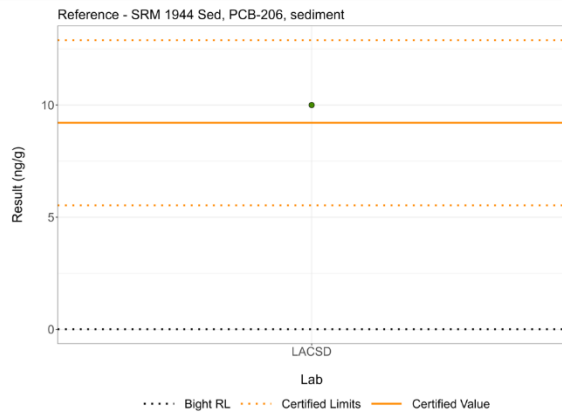
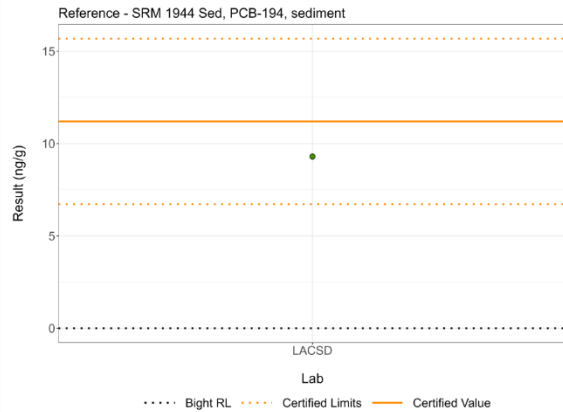
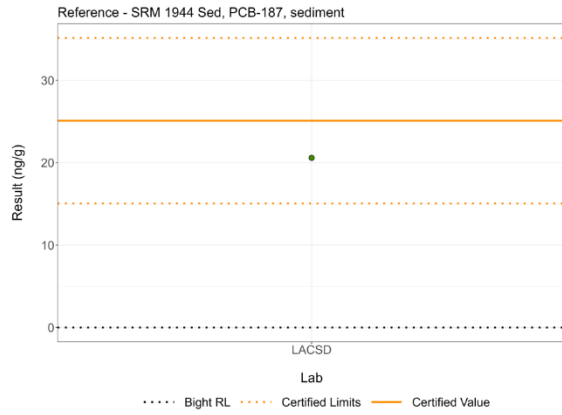
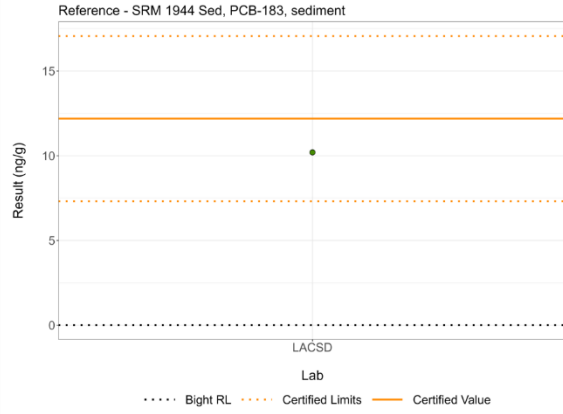
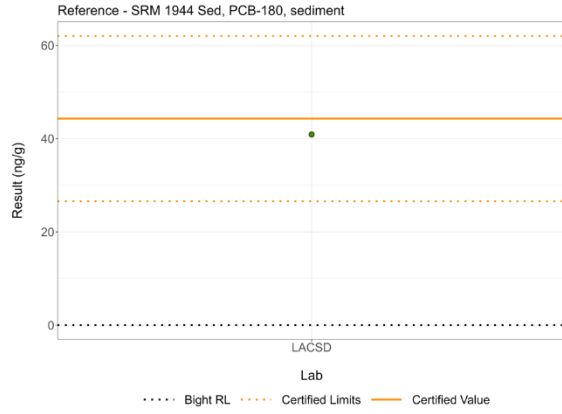
Second-Round Intercalibration Results for Core Individual Analytes

CRM PCBs: SRM 1944 (analyzed as unknown by laboratory)

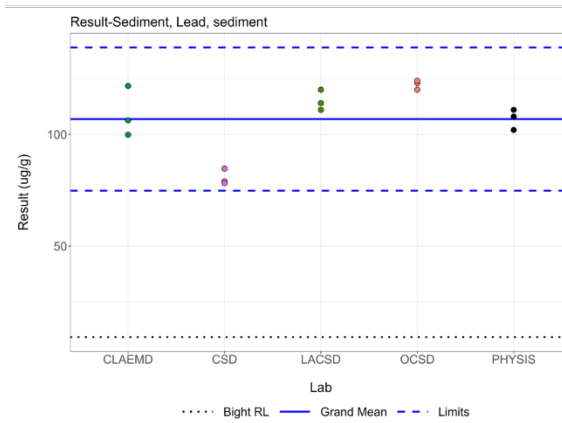
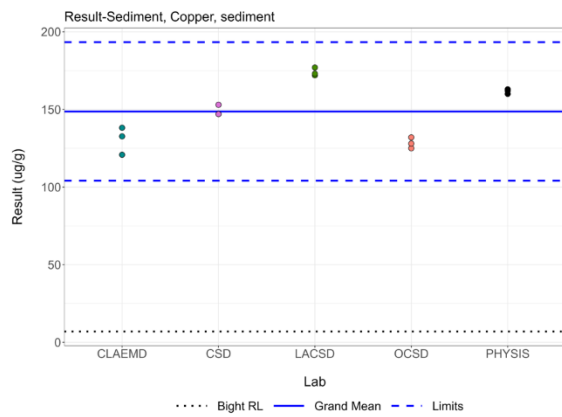
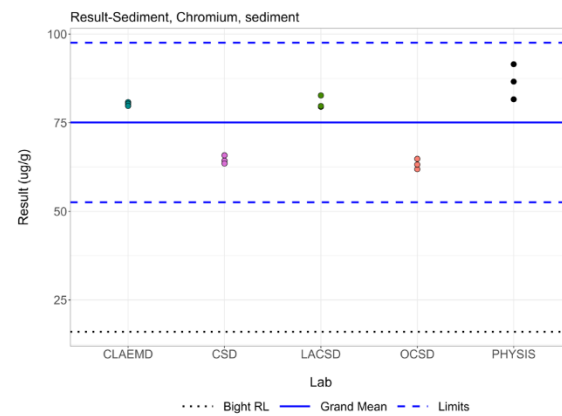
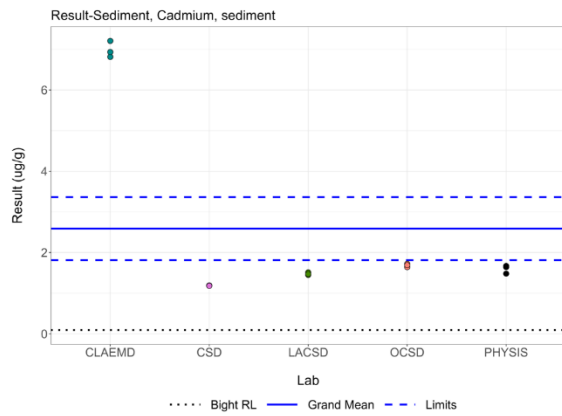
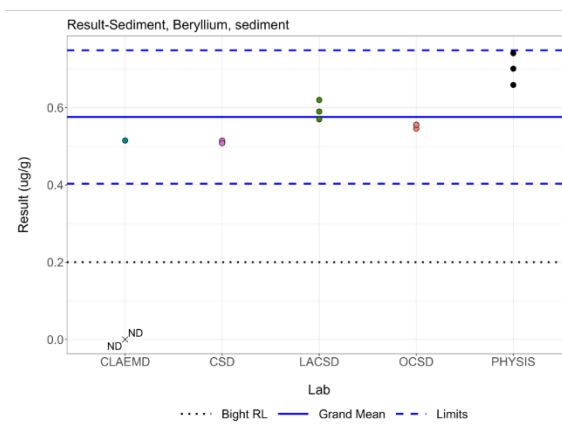
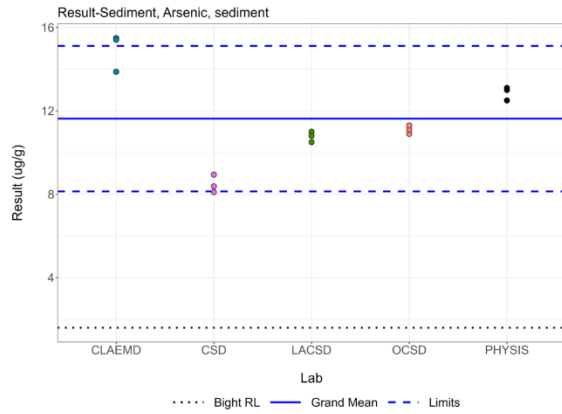


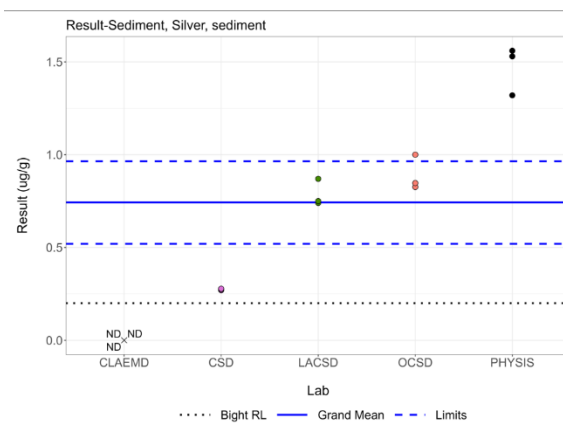
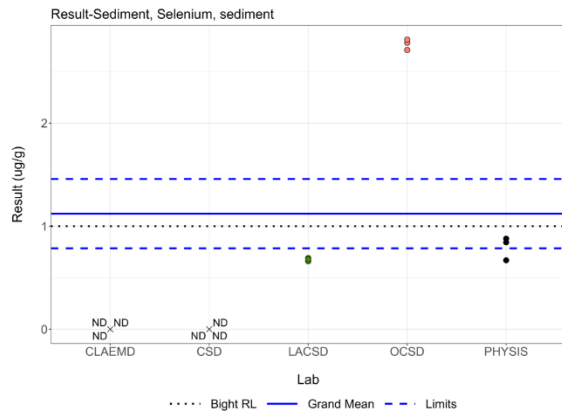
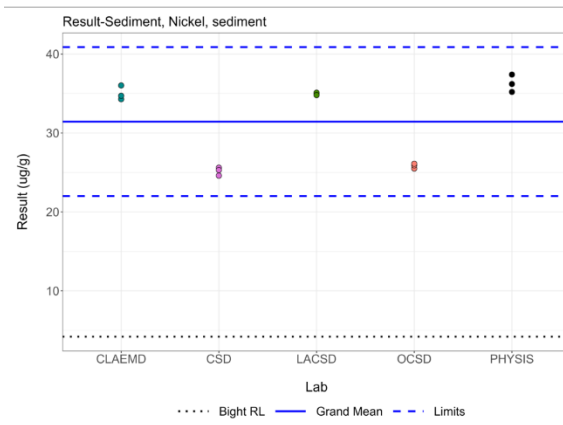
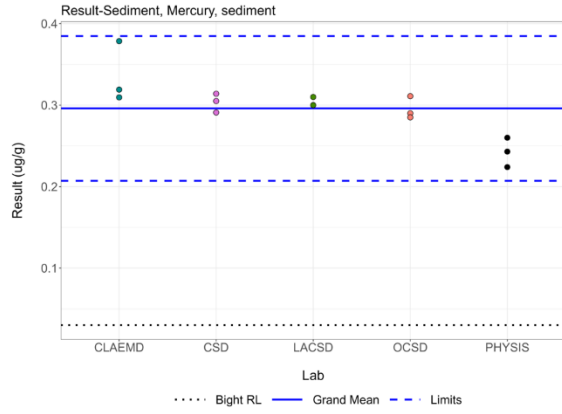






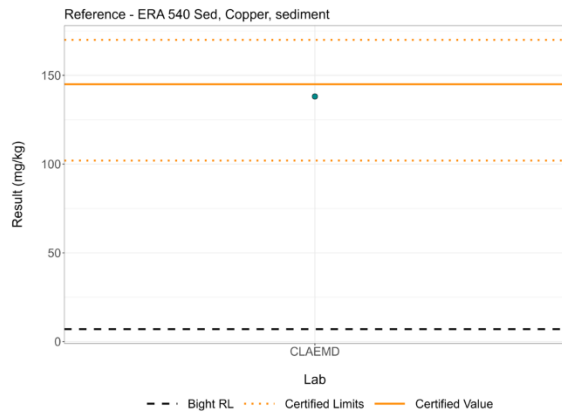
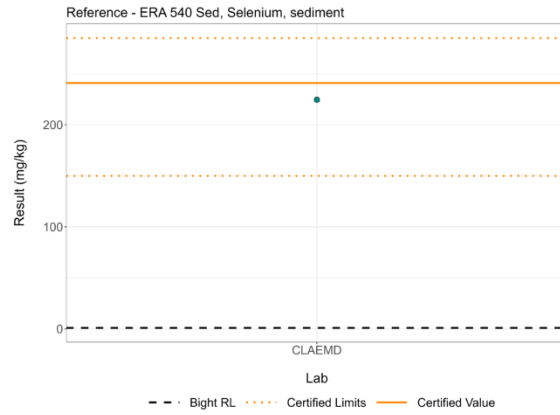
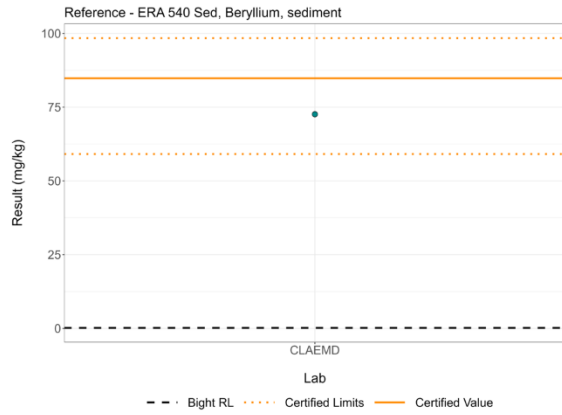
FRM metals





Third-Round Intercalibration Results for Core Individual Analytes

CRM metals: ERA 540 (analyzed as unknown by laboratory)



APPENDIX B. DETAILS ON QUALITY ASSURANCE/QUALITY CONTROL FOR BIGHT '23 SEDIMENT CHEMISTRY ANALYSIS

Holding Times

The vast majority of the data met holding time requirements (Table 17). The 92% holding time for mercury was because one laboratory had analyzed some mercury results within the holding time, but used a non-Bight CRM. This laboratory re-analyzed those samples with a Bight CRM after the holding period. Results for both sets of analyses, based on database measurements, were comparable (Figure 23), with all data points except for four within the 1:1 line (Re-analysis = $1.02 \times \text{original analysis} + 0.006$, $r^2 = 0.77$). In all four such cases, measured concentrations in the earlier analysis, compliant with the holding time requirement, were greater than those in the re-analysis. Accordingly, results from the first set of analysis were retained. Although the wrong CRM was used in the first analysis with this batch, the laboratory QA for this CRM met DQO requirements.

Table 17: Achieved holding times. Percent success is based on the number of samples meeting the required holding time.

Analyte class	Required holding time (days)	Holding time range (days)	Success (%)
Grain size	182.5	2-194	99.4-100
Total organic carbon (TOC)	365	29-415	93.6
Total nitrogen (TN)	NA	8-365	100
Mercury	182.5	6-376	92
Other metals	365	9-212	100
Organochlorine pesticides	365	12-254	100
PAHs	365	7-254	100
PCBs	365	12-254	100
Neonicotinoids	365	97-338	100
Pyrethroids	365	135-268	100
Tire-wear compounds	365	97-268	100
PFAS	365	153-626	39

Chlordane had 100% holding time success. While there were two non-Bight samples in the database that were analyzed past the holding time DQO because they were put with another batch by that laboratory at short notice; that batch had acceptable QA data compliant with the holding time DQO.

Grain size results depended on the size fraction, and those that did not meet the holding time DQO were due to a laboratory performing analyses just outside the 6-month window. There is no evidence that this contributed to measurement bias, since other monitoring programs, such as the Surface Water Ambient Monitoring Program (SWAMP), utilize one year holding times for grain size. The same was true for TOC. It is unlikely that the scattered exceedances for holding time DQOs affected the results of the chemistry analysis, as none of the analytes in question would likely have changed significantly over the times in question.

PFAS holding time success rates were low, as one laboratory had logistical issues with regards to their analysis that were disclosed to the Chemistry Technical Committee ahead of time. This is unlikely to affect results given the recalcitrance of PFOS and PFOA; intercalibration results are in keeping with this, as losses of PFAS due to long holding times are not reflected in the success rates observed (Table 17).

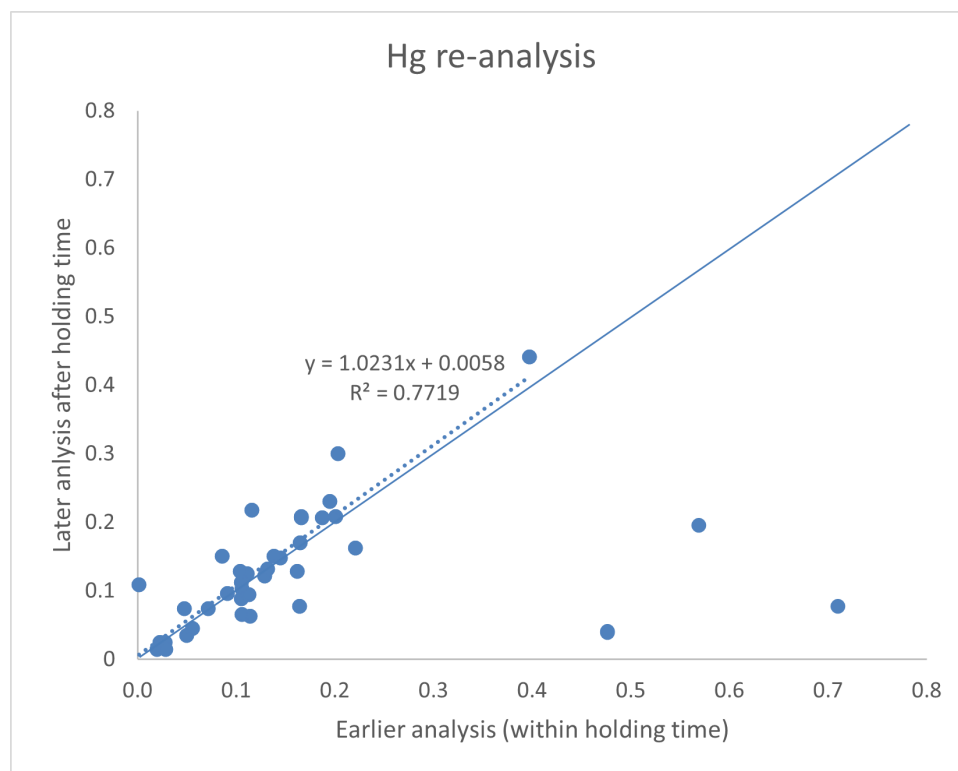


Figure 23: Comparison of results between original and repeated analysis for mercury.

Reporting Limits

As noted, the majority of the Bight '23 sediment chemistry data met RL requirements. Exceptions are as follows. Elevated reporting limits for some metals such as silver (Ag), and zinc (Zn), noted by two laboratories, were due to detections in a single procedural blank in a batch. These batches typically had multiple procedural blanks that otherwise had non-detectable levels of that metal. These laboratories reported that their long-term baseline for procedural blanks was stable and very low, so the elevated reporting limits noted (Table 18) are conservative, and affect only Ag (14 samples at most, or 4% of the overall dataset) and Zn (23 samples at most, or 6.7% of the dataset). Mercury reporting limits were elevated in one laboratory, affecting 33 samples or 10% of the overall dataset; however, all sample results from that laboratory were above the reported RLs and thus this issue did not affect data quality. The elevated reporting limits for some organochlorine pesticides, PCBs and PAHs, as reported by two laboratories, were adjusted due to the actual amount of sample used, and the moisture content of those samples, and affected approximately 10% of the total dataset of some of the analytes in each of these classes of contaminants (e.g., only PCB 8 for one laboratory, and for the other laboratory 7 of 12 organochlorine pesticides, 10 of 24 PAHs, and 33 of 43 PCB congeners).

The RLs among the laboratories generally varied by two orders of magnitude (Table 18). Some laboratories elected to use the required RL (e.g., one laboratory for PFAS, and tire-wear compounds), even if they were capable of improved sensitivity. Other laboratories, however, elected to use the lowest RL they could achieve. Since the data from the laboratories are combined, there should ideally be a narrower range of RLs. One future option is to require laboratories to report data only using the current RL (to not utilize lower RLs). This has the advantage of straightforward comparison to historical data acquired with similar RLs, but has the potential disadvantage that such censored data may confound further analysis and interpretation efforts. Alternatively, in a coordinated effort all laboratories could utilize lower RLs. This has the advantage of keeping methods state-of-the-art and continuing to detect and quantify legacy contaminants as they decrease in environmental concentration, with the limitation that lower program-wide RLs may not be technically possible, or logistically feasible.

Table 18: Achieved reporting levels in sediment (dry weight). Percent success is based on the number of samples meeting the required reporting level (RL). N/A=not applicable as a required reporting level was not set. *RL is for each individual analyte.

Analyte (analyte class)	Units	Required reporting level (RL)	Achieved range of RLs	Percent success
Aluminum (metal)	µg/g	N/A	2.5-3990	N/A
Antimony (metal)	µg/g	10	0.05-10	100%

Analyte (analyte class)	Units	Required reporting level (RL)	Achieved range of RLs	Percent success
Arsenic (metal)	µg/g	1.6	0.05-1.6	100%
Barium (metal)	µg/g	N/A	0.05-12	N/A
Beryllium (metal)	µg/g	0.2	0.05-0.2	100%
Cadmium (metal)	µg/g	0.09	0.005-0.09	100%
Chromium (metal)	µg/g	16	0.005-16	100%
Copper (metal)	µg/g	7	0.005-7	100%
Iron (metal)	µg/g	N/A	2.5-3990	N/A
Lead (metal)	µg/g	9.3	0.005-9.3	100%
Mercury (metal)	µg/g	0.03	0.00002-0.07	91%
Nickel (metal)	µg/g	4.2	0.02-4.2	100%
Selenium (metal)	µg/g	1	0.05-1	100%
Silver (metal)	µg/g	0.2	0.02-0.5	96%
Zinc (metal)	µg/g	30	0.05-40	94%
Polycyclic aromatic hydrocarbons (PAHs): 1,6,7-trimethylnaphthalene, 1-methylnaphthalene, 1-methylphenanthrene, 2,6-dimethylnaphthalene, 2-methylnaphthalene, acenaphthene, acenaphthylene, anthracene, biphenyl, fluorene, naphthalene	ng/g	20	0.5-35.1	91%
PAHs: benz[<i>a</i>]anthracene, benzo[<i>a</i>]pyrene, benzo[<i>b</i>]fluoranthene, benzo[<i>e</i>]pyrene, benzo[<i>g,h,i</i>]perylene, benzo[<i>k</i>]fluoranthene, chrysene, dibenz[<i>a,h</i>]anthracene, fluoranthene, indeno[1,2,3- <i>c,d</i>]pyrene, perylene, pyrene	ng/g	80	0.5-81	95%
Polychlorinated biphenyls (PCBs)*	ng/g	3	0.1-14	91%
Organochlorine pesticides (OCs)*	ng/g	0.5	0.5-64	91%

Analyte (analyte class)	Units	Required reporting level (RL)	Achieved range of RLs	Percent success
Polybrominated diphenyl ethers (PBDEs)*	ng/g	0.1	0.1	100%
Pyrethroid pesticides*	ng/g	0.5	0.5	100%
Neonicotinoid pesticides*	ng/g	5	0.25-4	100%
Tire-wear compounds*	ng/g	10	0.1-5	100%
Per- and polyfluorinated alkyl substances (PFAS) (PFOS and PFOA only)*	ng/g	1	0.08-0.5	100%
Total organic carbon	% by wt.	0.1	N/A	N/A
Moisture content	% by wt.	0.1	N/A	N/A

QA/QC for Analyte Classes

Total Organic Carbon

Two reference materials were used to test method accuracy: SRM 1944 New York/New Jersey Waterway Sediment, and SRM 1941b Organics in Marine Sediment. The latter was included as SRM 1944 became unavailable commercially during Bight '23 due to the dwindling supply of remaining stock. Laboratories were required to obtain concentrations within 20% of the reference value, as well as precision for sample duplicates of RPD < 30%. This was achieved by all laboratories except one, for which its subcontracting laboratory evaluated blank spikes and duplicates instead at 0.82% by weight, with an accuracy within 1% of the spiked value and an RPD of 0%. In comparison, other participating laboratories met accuracy requirements except for one batch from one laboratory, affecting 3 samples total; and had sample duplicate RPDs between 1.3-10%. These deviations from QC goals are minor.

Metals

ERA Certified Reference Material 540 Metals in Soil, Lot D099-540 was used to test method accuracy. Laboratories are required to obtain concentrations within PT performance acceptance limits of certified value for all 15 analytes. Blank spikes and duplicates into pure laboratory water or Ottawa sand were used to assess precision, instead of matrix spikes and duplicates as was used in Bight '18. This was done to address issues with variable recoveries and duplicate reproducibility previously observed, from high background levels of metals that were natively present, as well as inhomogeneous samples.

Success rates were high (>86%) for the accuracy of method blanks, reference materials, and blank spike and duplicate accuracy, as well as for blank spike duplicate precision (Table 5). Several laboratories did conduct matrix spikes as part of their regular QA/QC processes, and in fact, one laboratory performed spikes and duplicates with sample sediments instead of blank spikes. The latter reason is why the success rate for metal sample duplicate frequency and precision is low (84% and 46% respectively, Table 8). An analysis of precision among blank spikes, matrix spikes, and sample duplicates for metals (Table 5) shows that variability between duplicates was similar across laboratories for both blank spikes and matrix spikes. This observation suggests that both these types of spikes are suitable measures of precision for metals, despite potential confounding factors such as differences in native metal concentrations and spike levels (e.g., precision will be better with greater concentrations). Both these types of spikes had better precision than that observed for sample duplicates, which was expected given the very low concentrations in many samples (e.g., at or near laboratory MDLs) as well as potential sample inhomogeneity. Many such samples with less precision in sample duplicates were sandy sediments with very low concentrations, and hence inherently worse precision.

The metals blank spike and duplicate acceptability frequency (78%) was due to several issues. One laboratory had two batches of metals analyses that failed the blank spike and duplicate DQOs for aluminum and iron. However, these two metals, being natural constituents of the inorganic components of sediments, are not further discussed in Bight sediment evaluations as evidenced by the lack of system-wide RLs for them (Table 19). A different laboratory had ten batches of blank spikes, affecting 83 samples total, that were systematically lower than the spiked value by 40% for beryllium only. There were no issues with blank spike duplicate precision for this metal, and other metals were unaffected. Accordingly, these issues are likely to be minor.

Table 19: Averages and ranges for RPDs for matrix spike duplicates, blank spike duplicates, and sample duplicates for metals. Weck did not analyze metals for Bight '23. Matrix spikes and matrix spike duplicates were not required for metals.

Laboratory	Matrix spike RPD average	Matrix spike RPD range	Blank spike RPD average	Blank spike RPD range	Sample duplicate RPD average	Sample duplicate RPD range
CLAEMD	NA	NA	1.6%	0 – 5.4%	3.6%	0 – 31%
CSD	1.1%	0 – 17%	0.5%	0 – 2.4%	4.2%	0 – 11%
LACSD	1.8%	0 – 6.4%	1.7%	0 – 5.7%	4.5%	0 – 35%
OC San	NA	NA	4.9%	1 – 10%	3.2%	0 – 15%
Physis	NA	NA	2.3%	0 – 15%	16	0 – 91%

Organics

The same two CRMs used for TOC (SRM 1944 and SRM 1941a) were also used for organics to evaluate method accuracy for those analytes with certified or reference values: organochlorine pesticides, PAHs, and PCBs. Laboratories are required to obtain concentrations within 40% of the certified or reference value for 70% of the compounds of each of these classes except for PAHs, for which 80% of the compounds needed to be within that concentration window.

Success rates for accuracy and precision were >81% for the classes of analytes noted above (Tables 6 and 7) and are similar to those for Bight '18 (87-100%) for organochlorine pesticides and PAHs (Du et al. 2020), suggesting comparable levels of data quality. One laboratory had consistently low recoveries for PAHs using SRM 1941b for reasons unknown. However, this laboratory met Bight '23 QA goals with an alternate reference material (Phenova) for PAHs, and had comparable performance for PAHs with other laboratories in the intercalibration exercise using SRM 1944 (Table 6). Matrix spike accuracy and precision can be affected by high background levels of PCBs and PAHs in some samples, particularly at contaminated sites such as Los Angeles/Long Beach harbor. This may be reflected in the success rates (Tables 6-7) for these analytes for accuracy (85 and 83%, respectively) and precision (87 and 70%, respectively). Accordingly, laboratories flagged data in which background levels for matrix spikes were greater than 1/3 of the spike level, a typical threshold for EPA Methods. Co-elution issues were also a reason for poor recoveries in reference materials and matrix spikes for a few analytes. This affected *cis*-nonachlor in matrix spikes for one laboratory, for example. Other co-elution issues were manifested in calibration curves, as noted by initially poor recoveries with one laboratory for PCBs 153, which co-eluted with PCB 168. The latter congener is typically not

found in the environment, but was present in that laboratory's calibration mixture. Recoveries were acceptable after the laboratory recalibrated without PCB 168. Finally, it should be noted that close elution times could also create issues with integration and quantification of some analytes, such as larger PAHs, although these analytes were typically not detected in Bight '23 samples.

The low values for matrix spikes for the PFAS compounds PFOS and PFOA (Table 7) were due to the fact that one of the laboratories analyzing for many of the PFAS samples had performed matrix spikes for only one batch of samples out of 13 total. The accuracy and precision of matrix spikes in this batch met Bight '23 QA/QC requirements not only for PFOS and PFOA as described in Table 10, but also for all other PFAS analytes submitted by this laboratory (i.e., those in EPA Method 1633) except for 8:2 FTS (both spike recoveries at 160%). The low success rate for PFAS matrix spike accuracy and precision was due to one laboratory performing these measurements for selected batches only. Given these QA results, it is likely that the matrix spike accuracy and precision results are representative.

APPENDIX C. GEOGRAPHIC DISTRIBUTION AND MAGNITUDE OF ANALYTES

The following maps (Figures 24-49) show the 2023 Southern California Bight regional monitoring survey's geographic distribution of sediment contaminant concentration, in $\mu\text{g/g}$ dry weight for metals, ng/g dry weight for organic analytes, and percent by weight for total organic carbon (TOC) and total nitrogen (TN). All concentration data is binned into the following groups: non-detected values (i.e., below the analysis lab's method detection limit), minimum to 25th percentile, 25th percentile to median, median to 75th percentile, 75th percentile to 90th percentile, and 90th percentile to maximum (plotted separately to highlight locations with the greatest concentrations). Numbers in parentheses after the concentration ranges are the number of samples in each bin. For analytes that are summed values (e.g., total PCBs), non-detects are treated as a concentration of zero.

For a rough comparison, background metal dry-weight sediment concentrations in the 1998 Bight cycle (Noblet et al. 2003), calculated using iron-normalization and regression to establish baseline levels (Schiff and Weisberg, 1999), ranged from zero up to 5-15 $\mu\text{g/g}$ for arsenic, 0.13-0.72 $\mu\text{g/g}$ for cadmium, 12-110 $\mu\text{g/g}$ for chromium, 5-50 $\mu\text{g/g}$ for copper, 6-32 $\mu\text{g/g}$ for lead, 0.06-0.36 $\mu\text{g/g}$ for mercury, 20-80 $\mu\text{g/g}$ for nickel, 0.1-0.6 $\mu\text{g/g}$ for silver, and 13-200 $\mu\text{g/g}$ for zinc. Background levels of aluminum, antimony, barium, beryllium, and selenium from the first two Bight cycles (Schiff and Gossett 1998, Noblet et al. 2003) are not available, although Bight '03 did note that relatively high levels of antimony and nickel in Channel Island sediments may be suggestive of natural abundances (Schiff et al. 2006). These concentrations are locale-specific, as they depend on iron sediment concentrations and can be invalidated under

anaerobic conditions that would remobilize particulate-phase sedimentary iron (Schiff and Weisberg, 1999, Noblet et al. 2003). These 1998 levels do not necessarily reflect current background metal concentrations in the SCB, the determination of which is beyond the scope of this report and was not assessed from Bight '08 onwards.

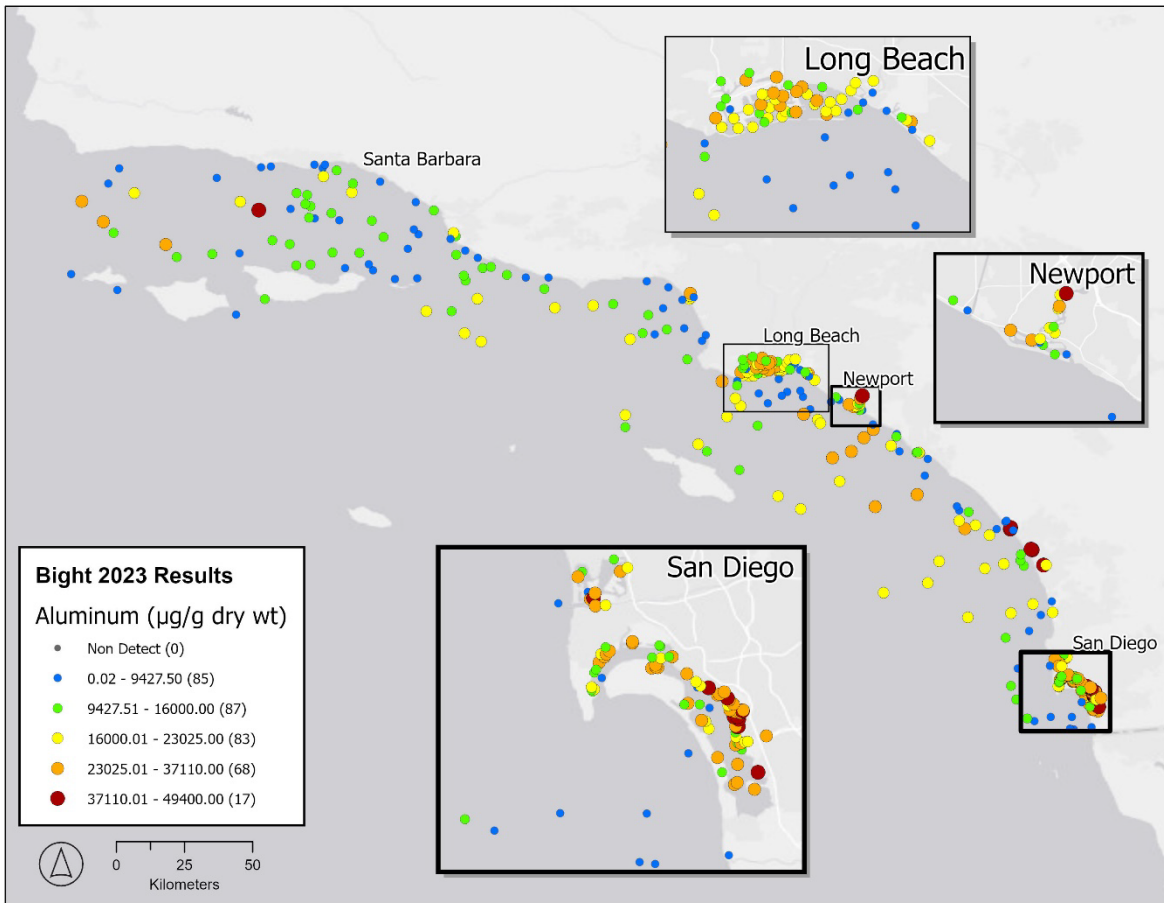


Figure 24: Map of aluminum concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

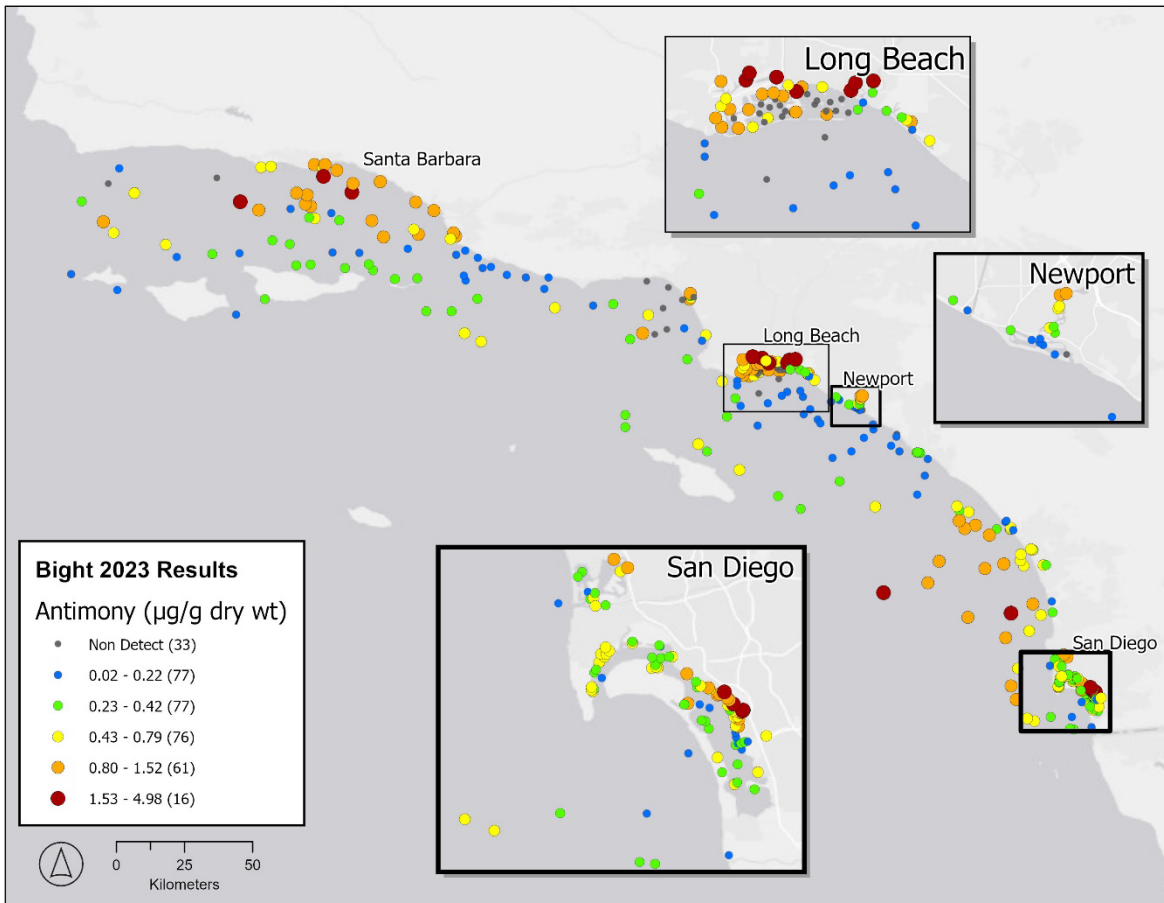


Figure 25: Map of antimony concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

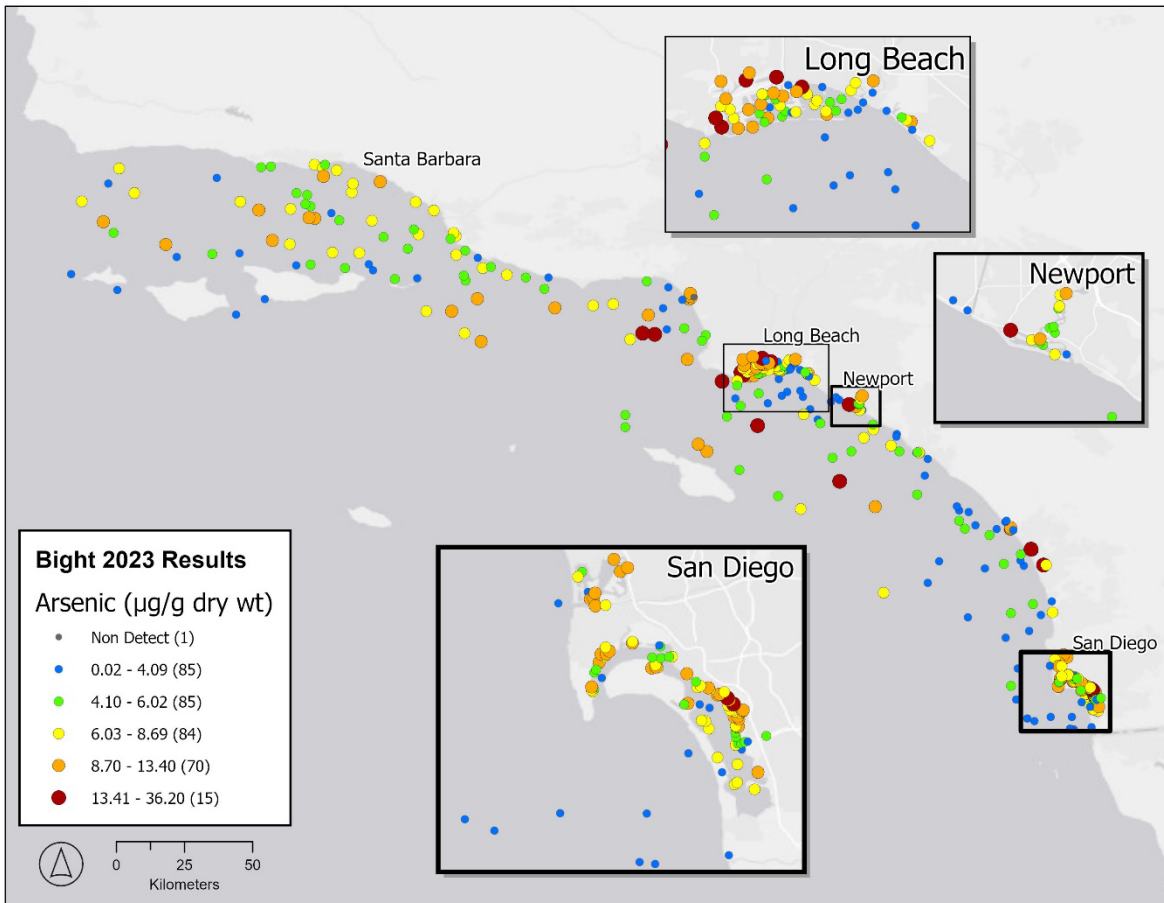


Figure 26: Map of arsenic concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

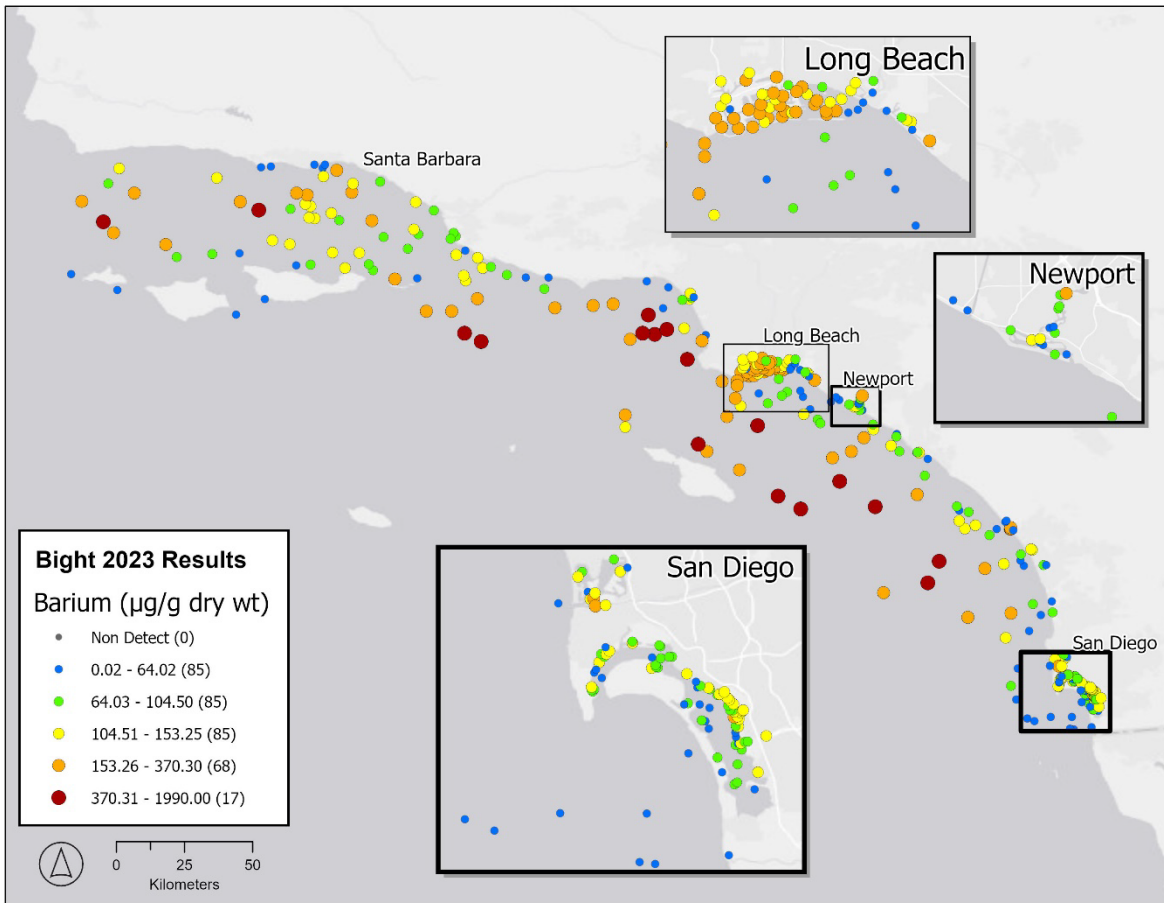


Figure 27: Map of barium concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

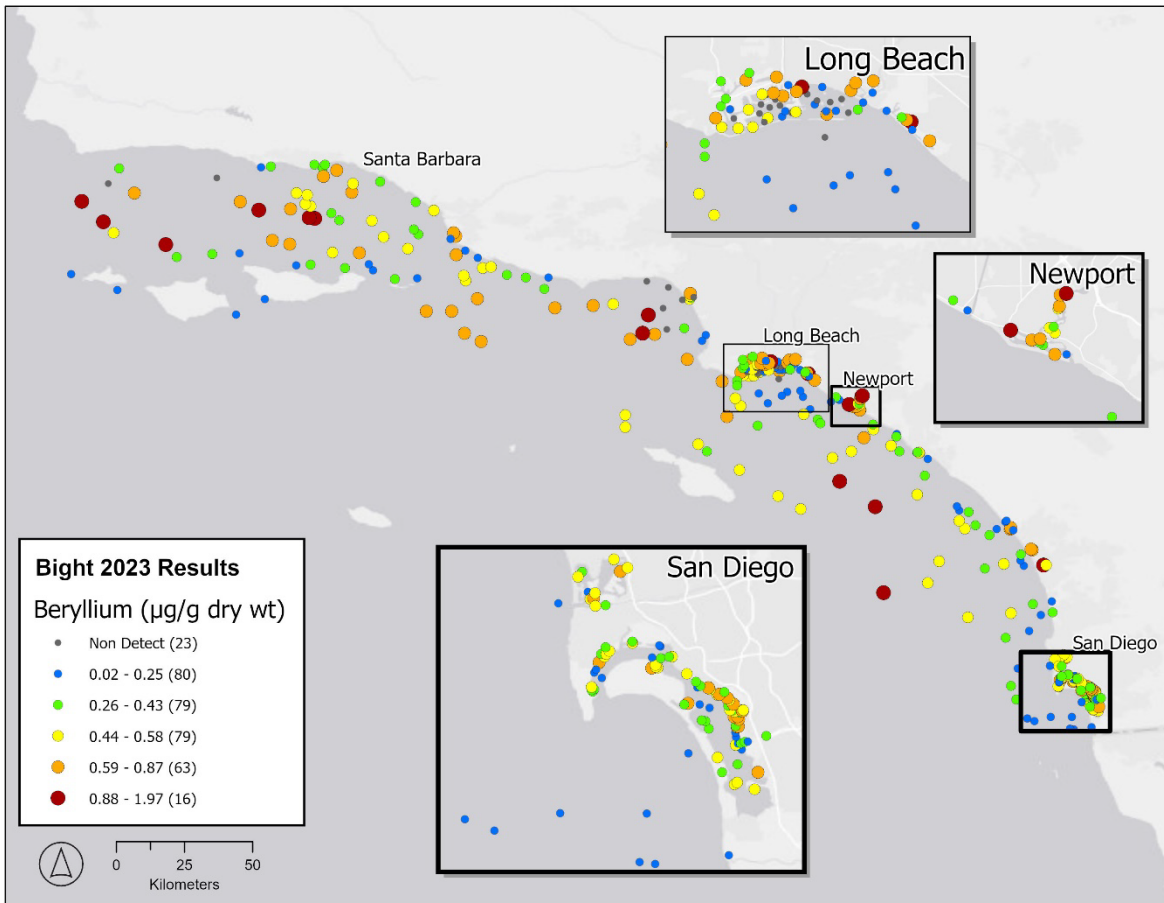


Figure 28: Map of beryllium concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

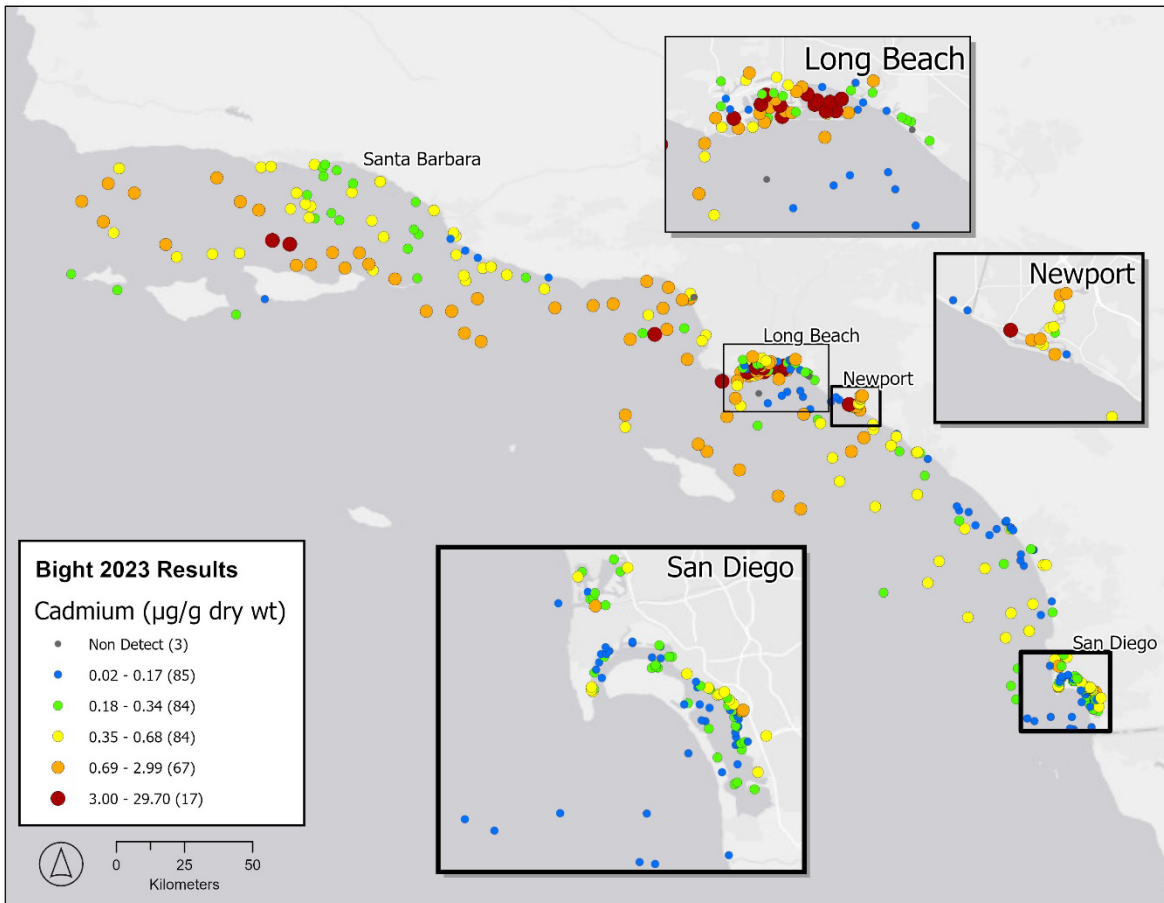


Figure 29: Map of cadmium concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

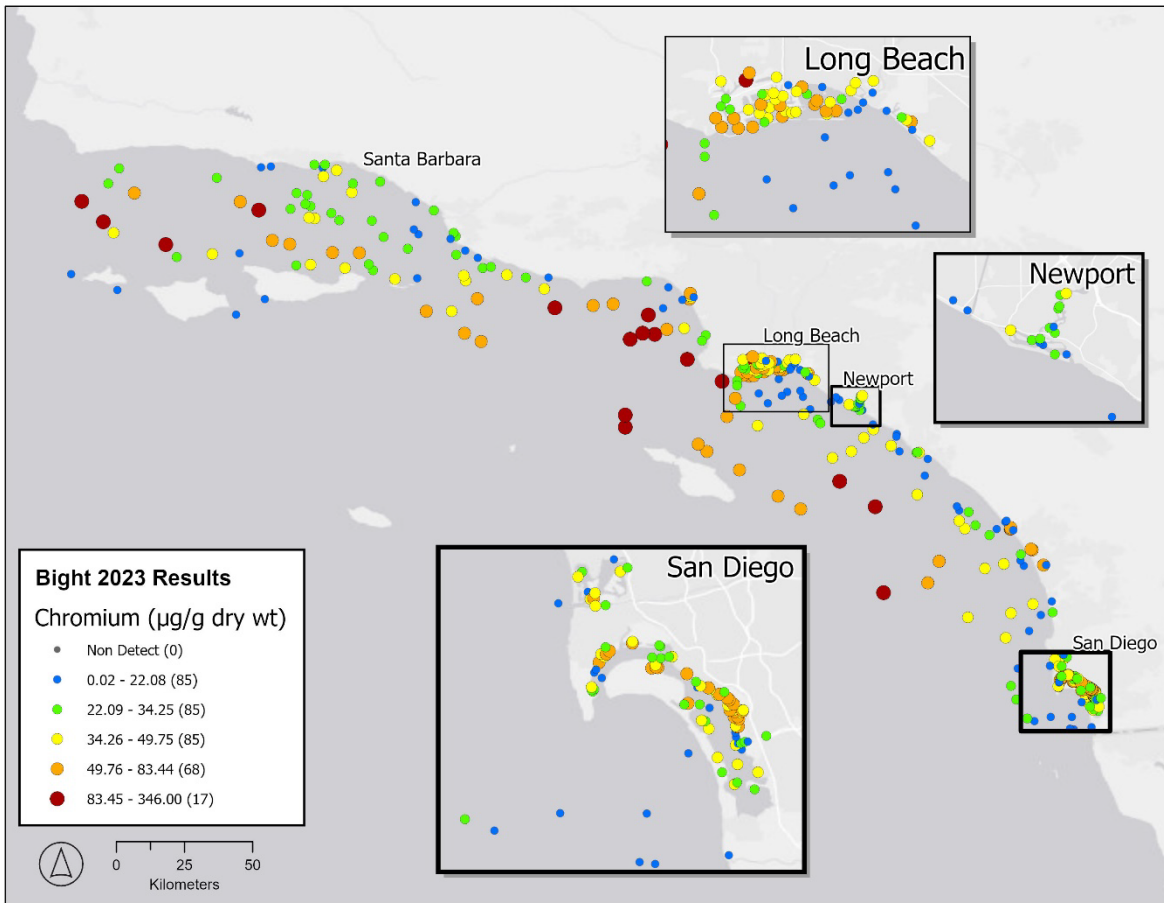


Figure 30: Map of chromium concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

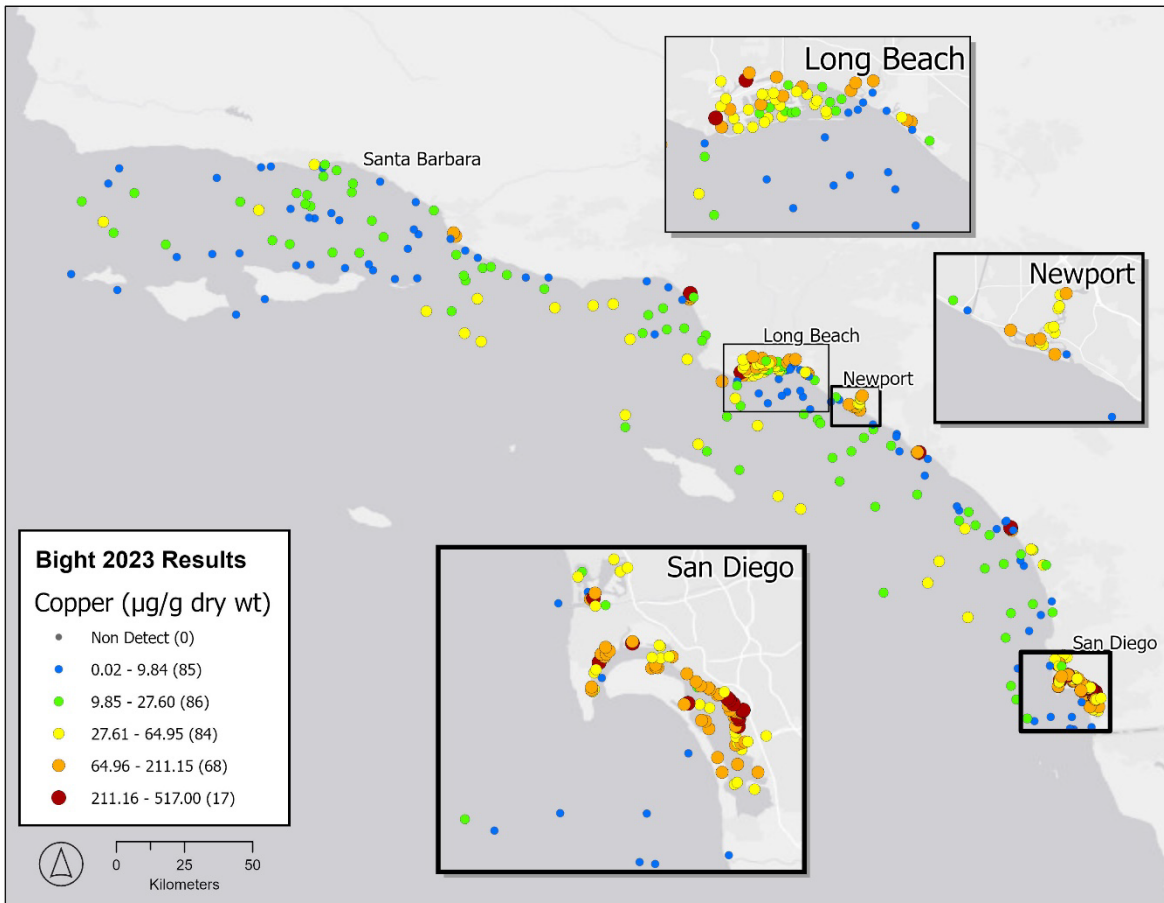


Figure 31: Map of copper concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

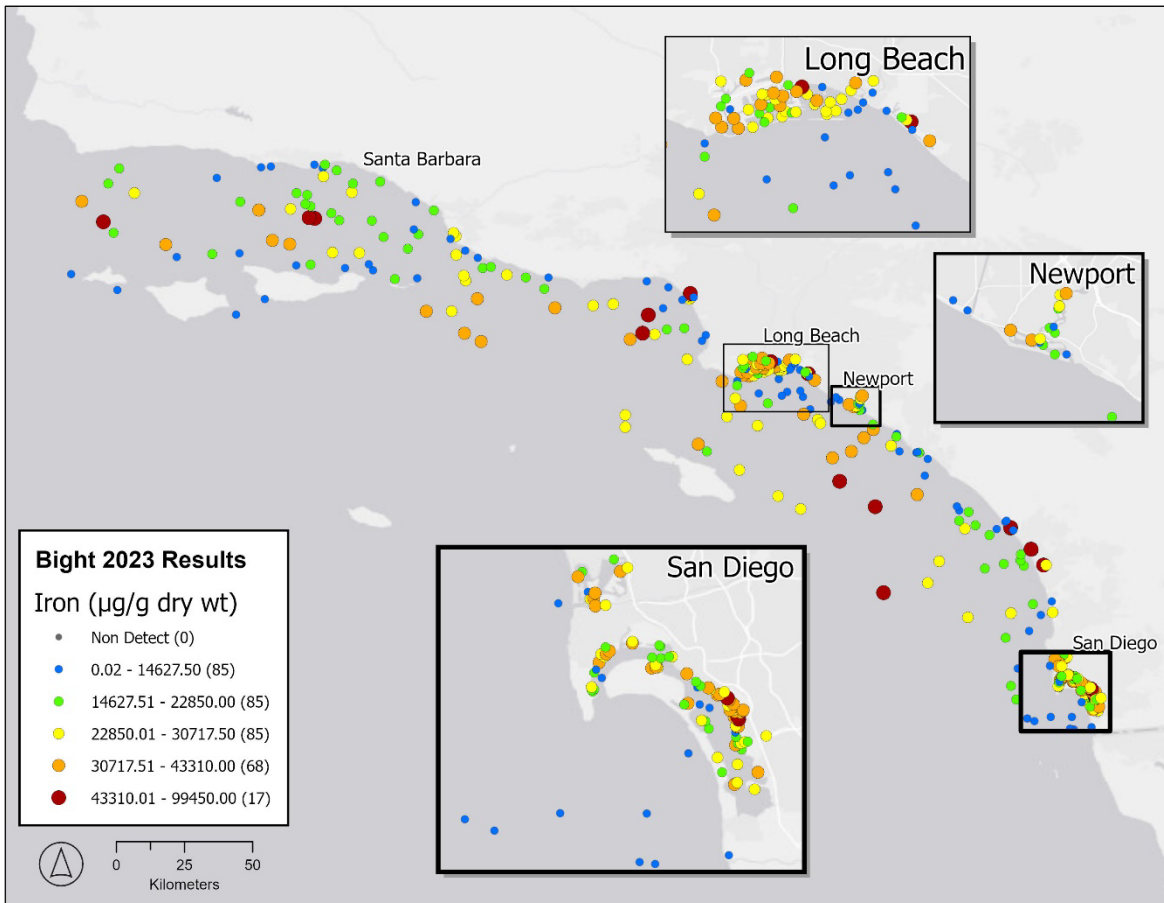


Figure 32: Map of iron concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

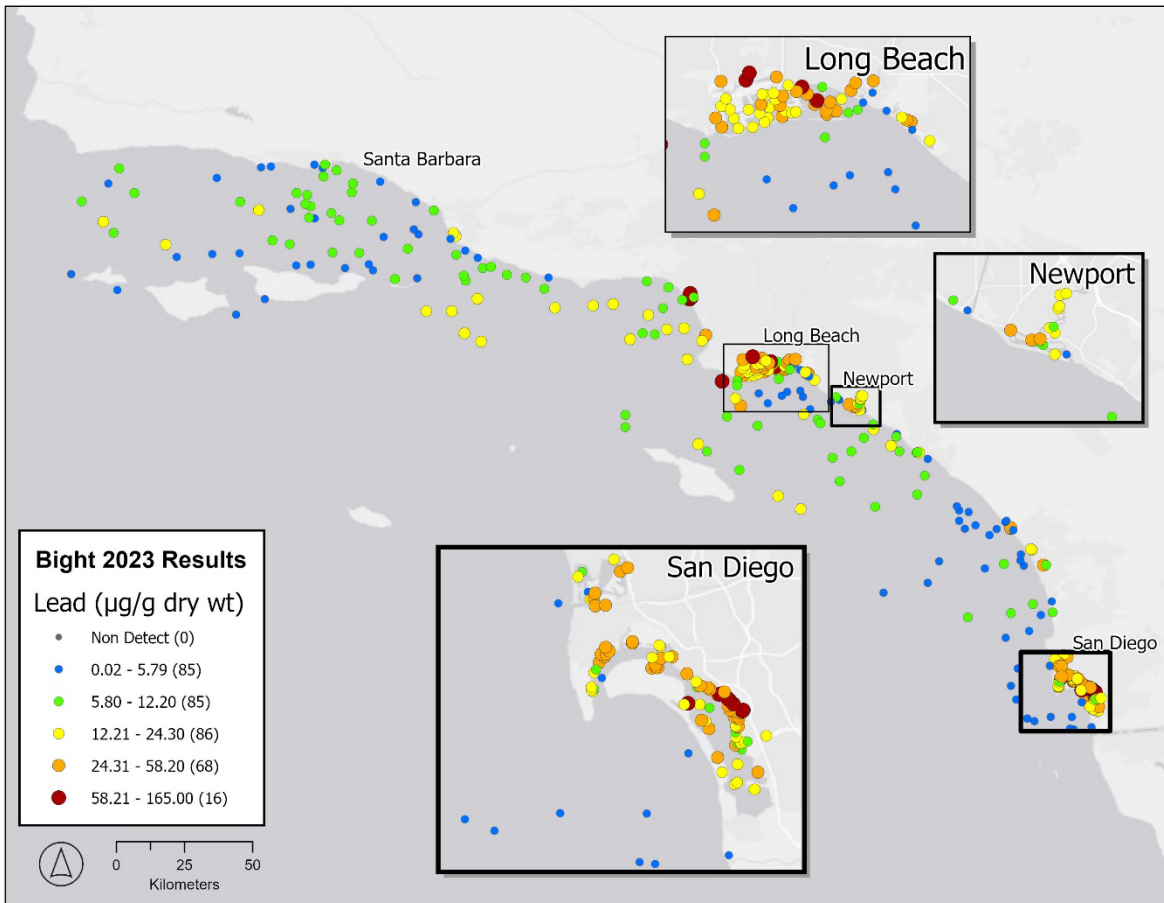


Figure 33: Map of lead concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

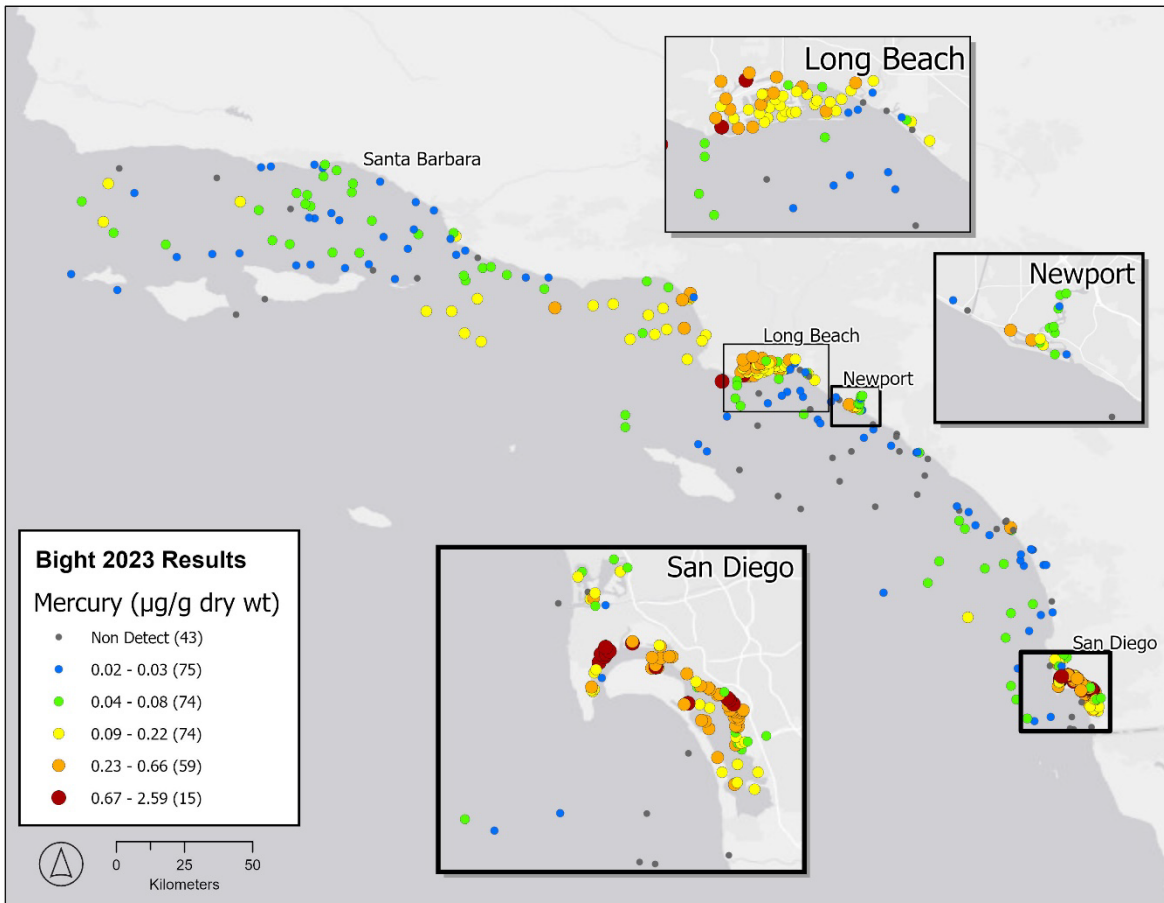


Figure 34: Map of mercury concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

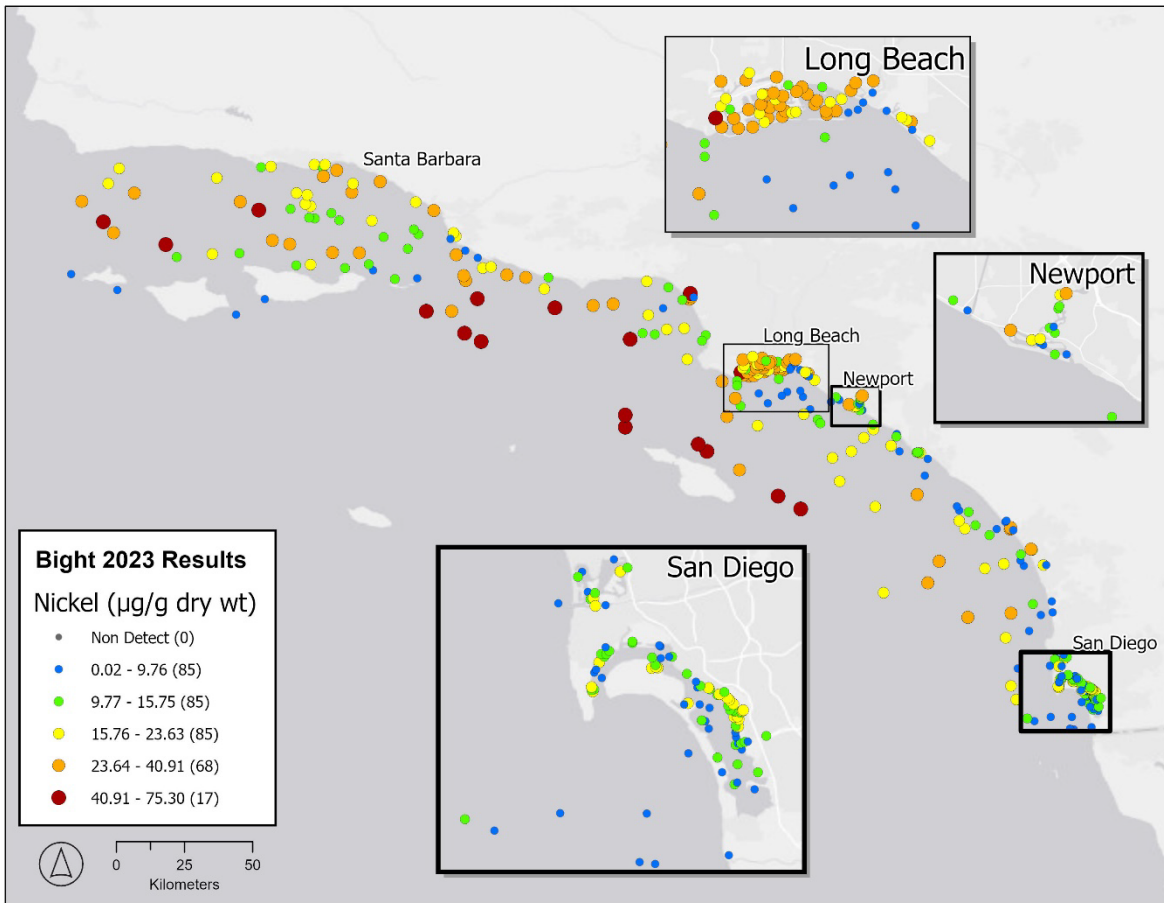


Figure 35: Map of nickel concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

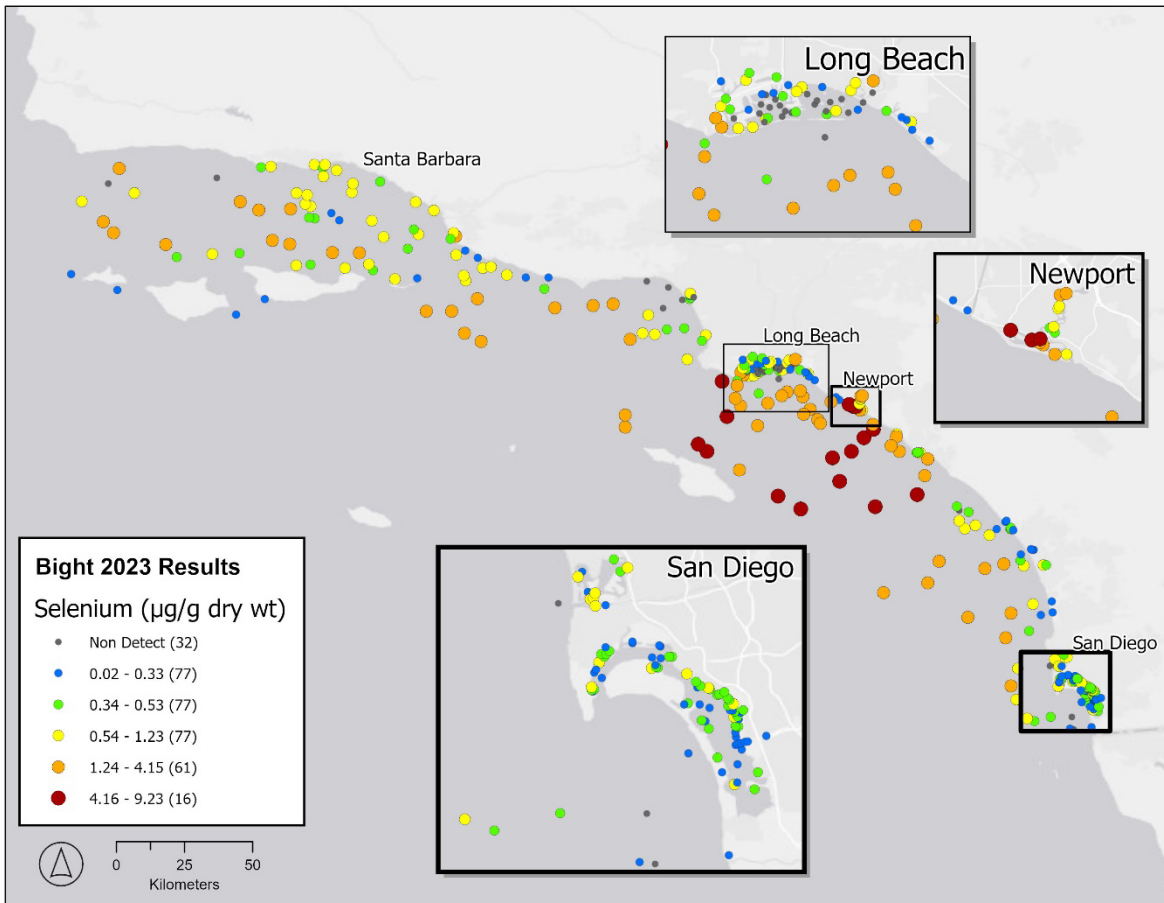


Figure 36: Map of selenium concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

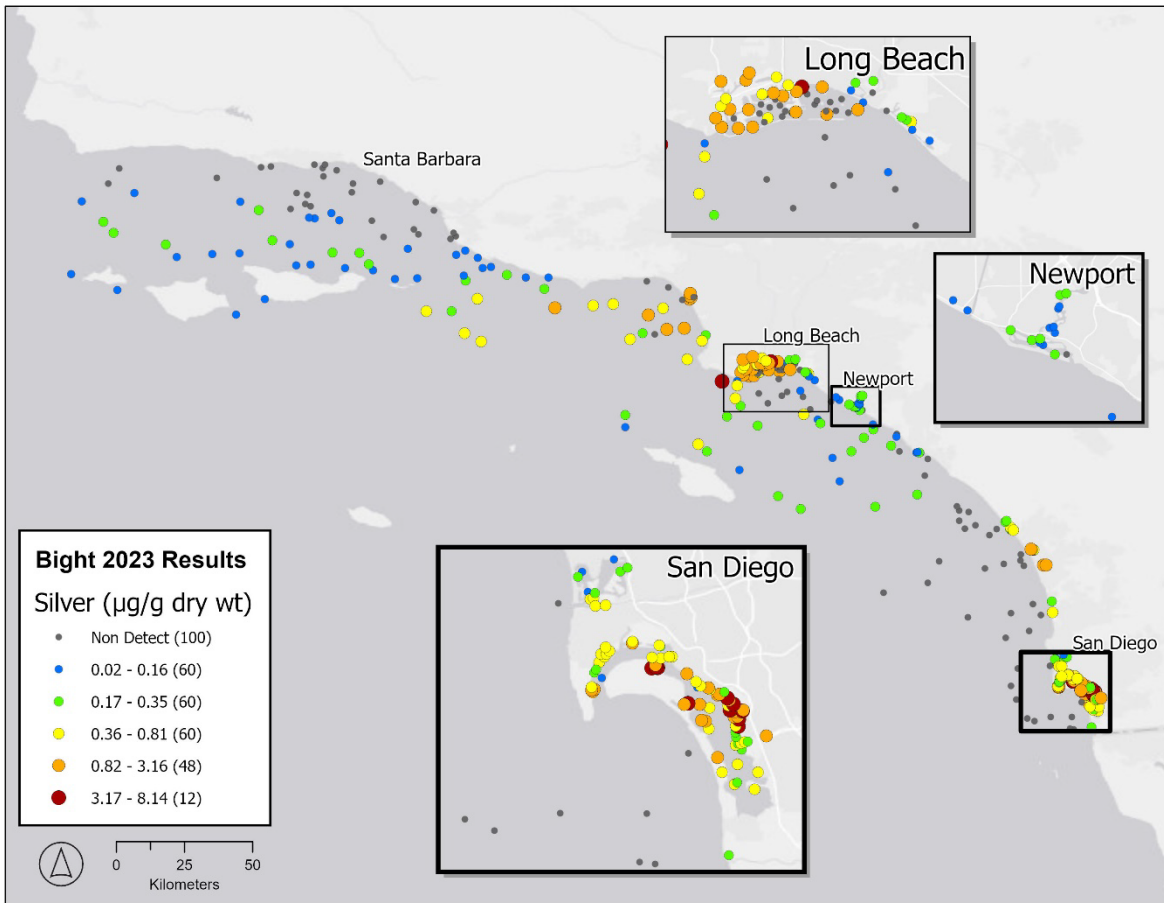


Figure 37: Map of silver concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

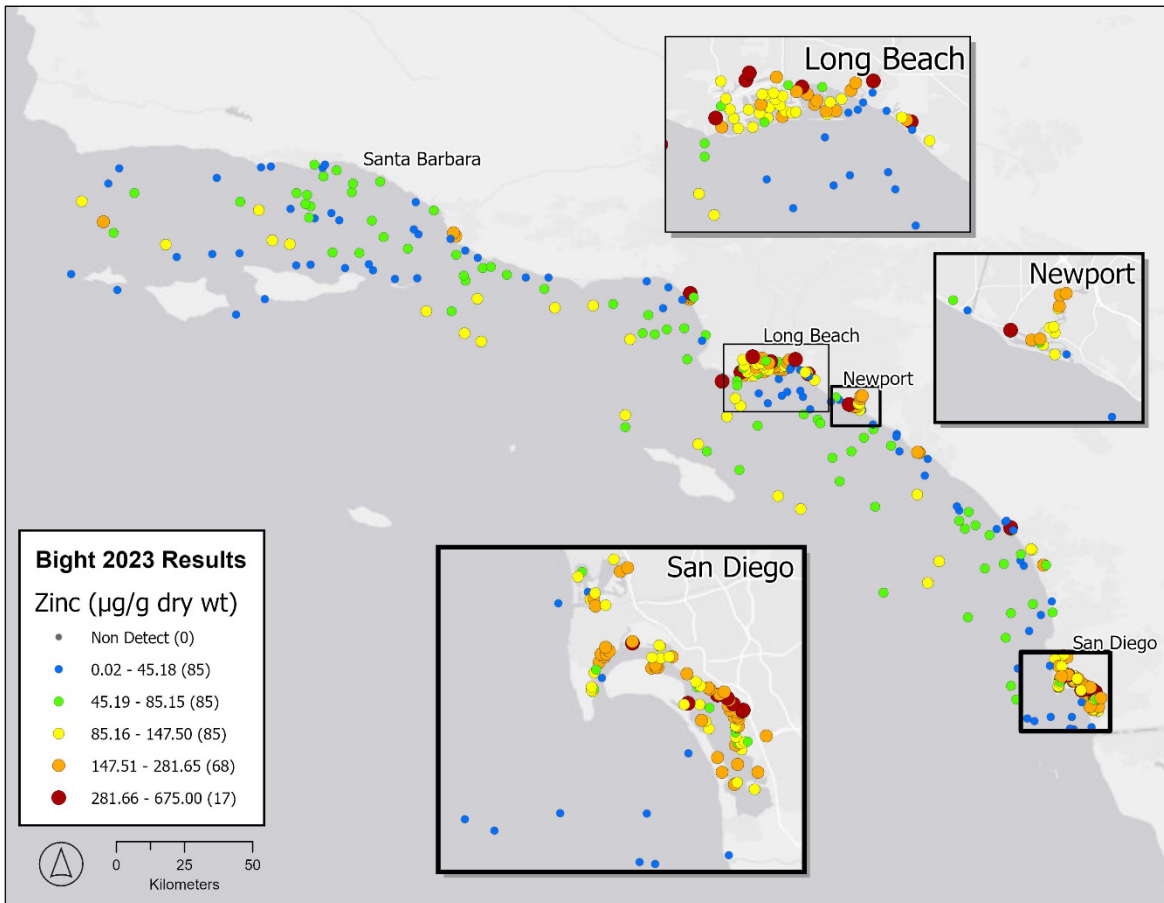


Figure 38: Map of zinc concentrations in $\mu\text{g/g}$ dry weight in Bight '23 sediments.

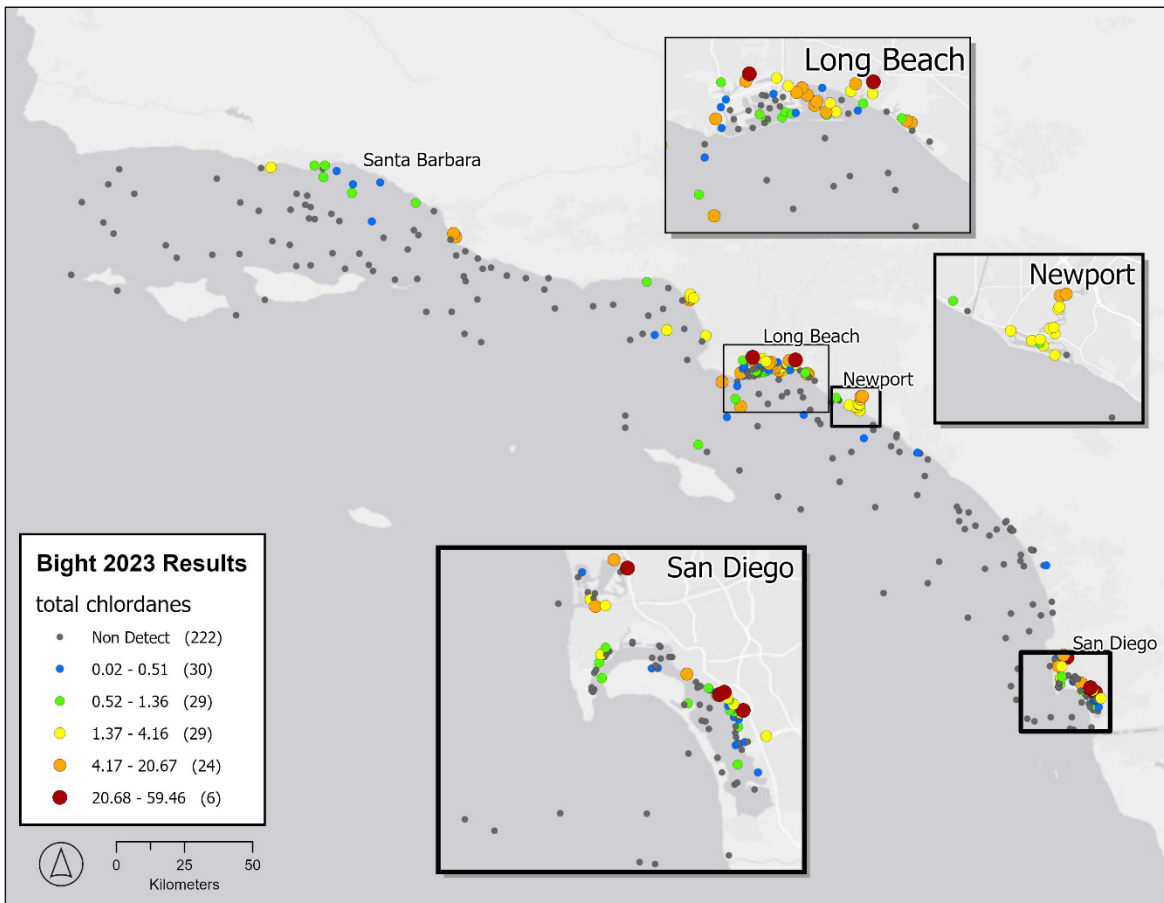


Figure 39: Map of total chlordane concentrations in ng/g dry weight in Bight '23 sediments.

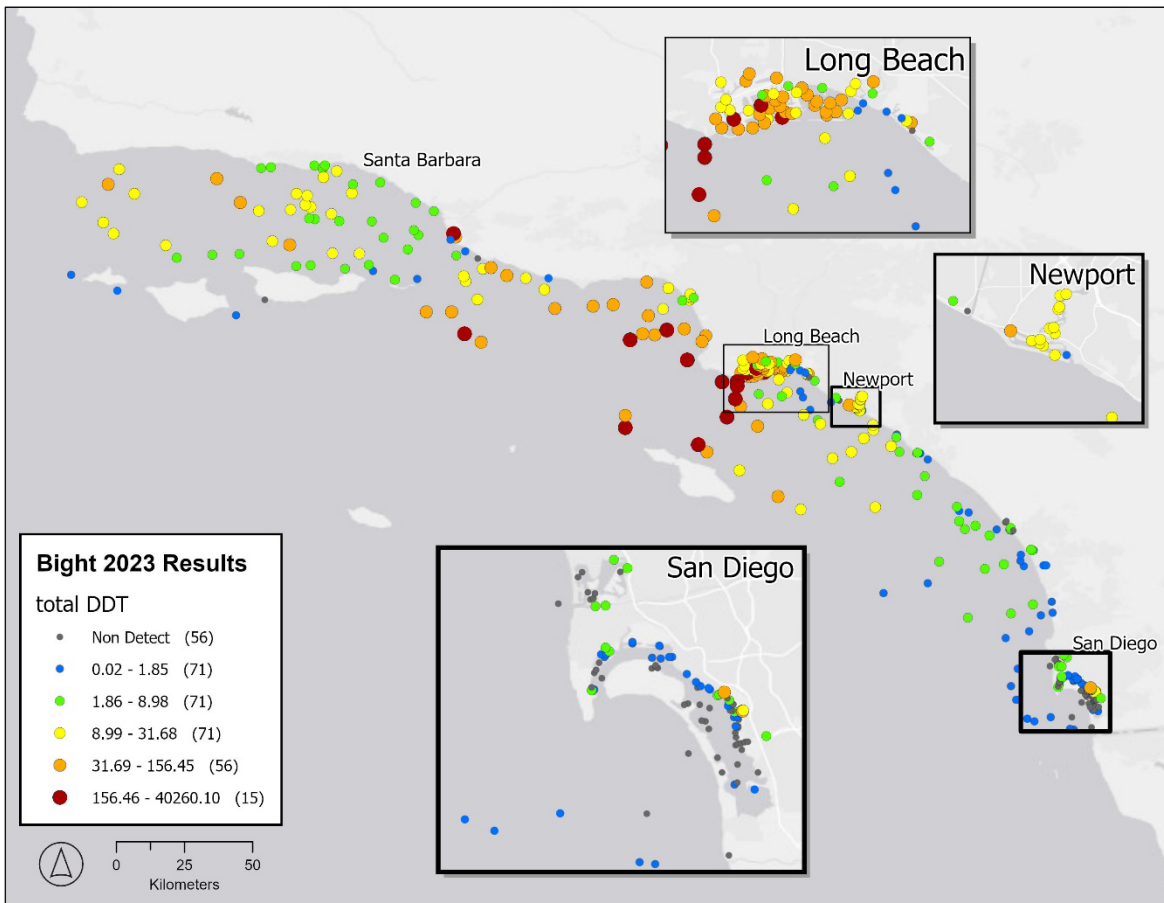


Figure 40: Map of total DDT concentrations in ng/g dry weight in Bight '23 sediments.

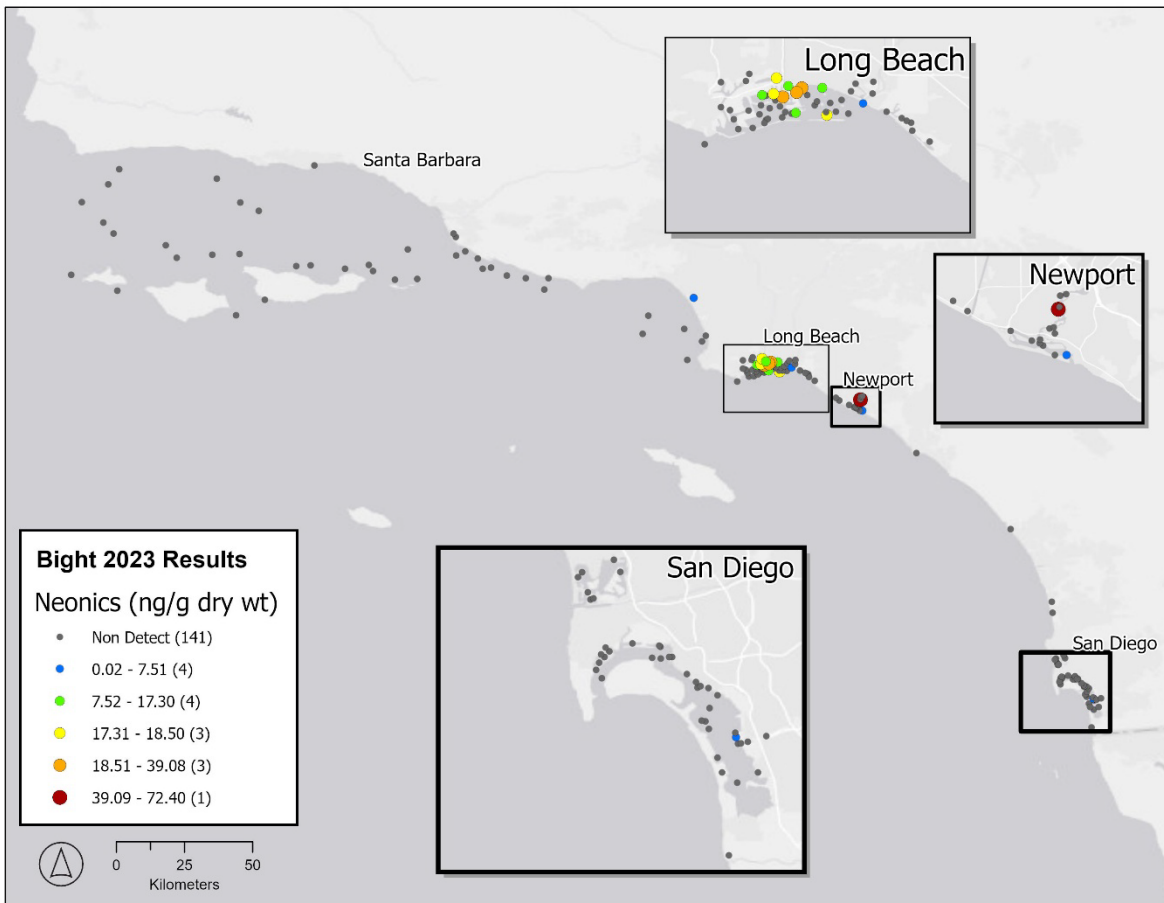


Figure 41: Map of total neonicotinoid pesticide concentrations in ng/g dry weight in Bight '23 sediments.

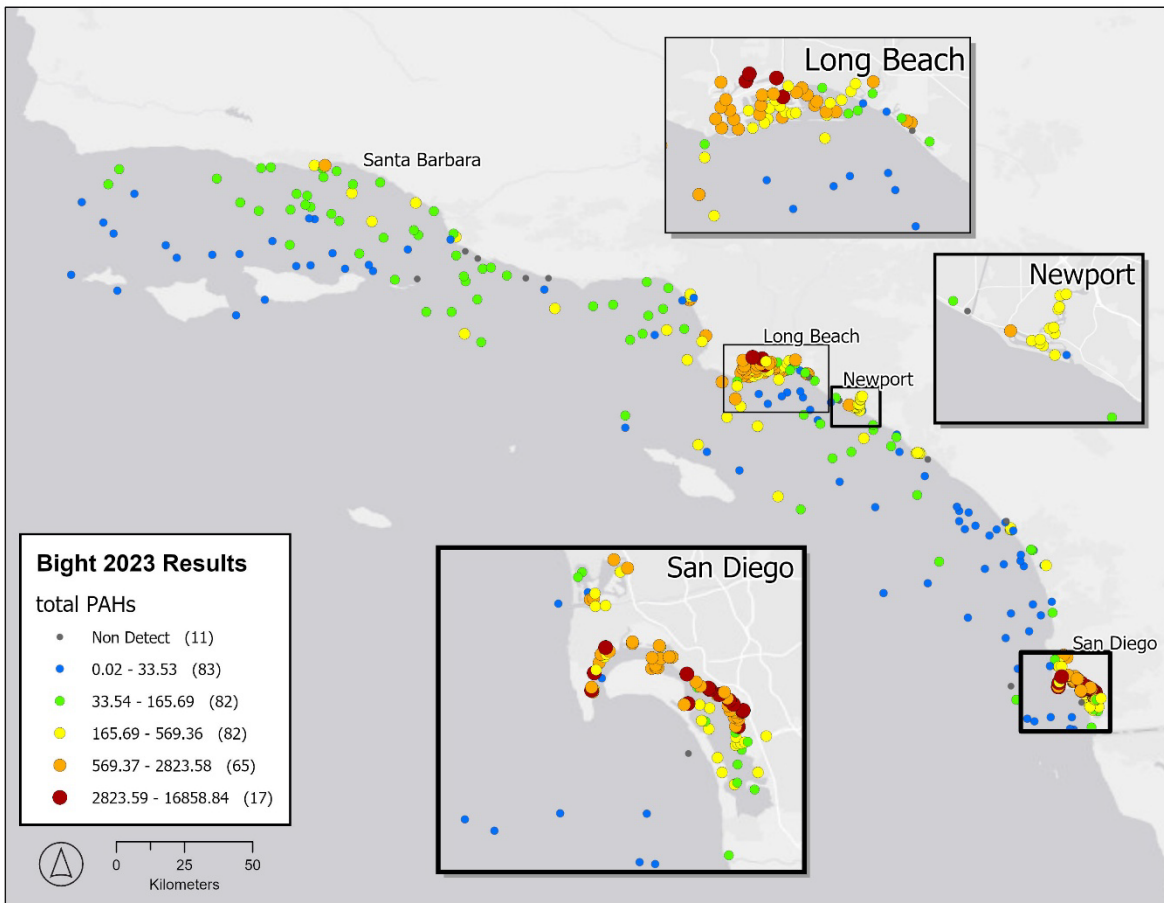


Figure 42: Map of total polycyclic aromatic hydrocarbon (total PAH) concentrations in Bight '23 sediments.

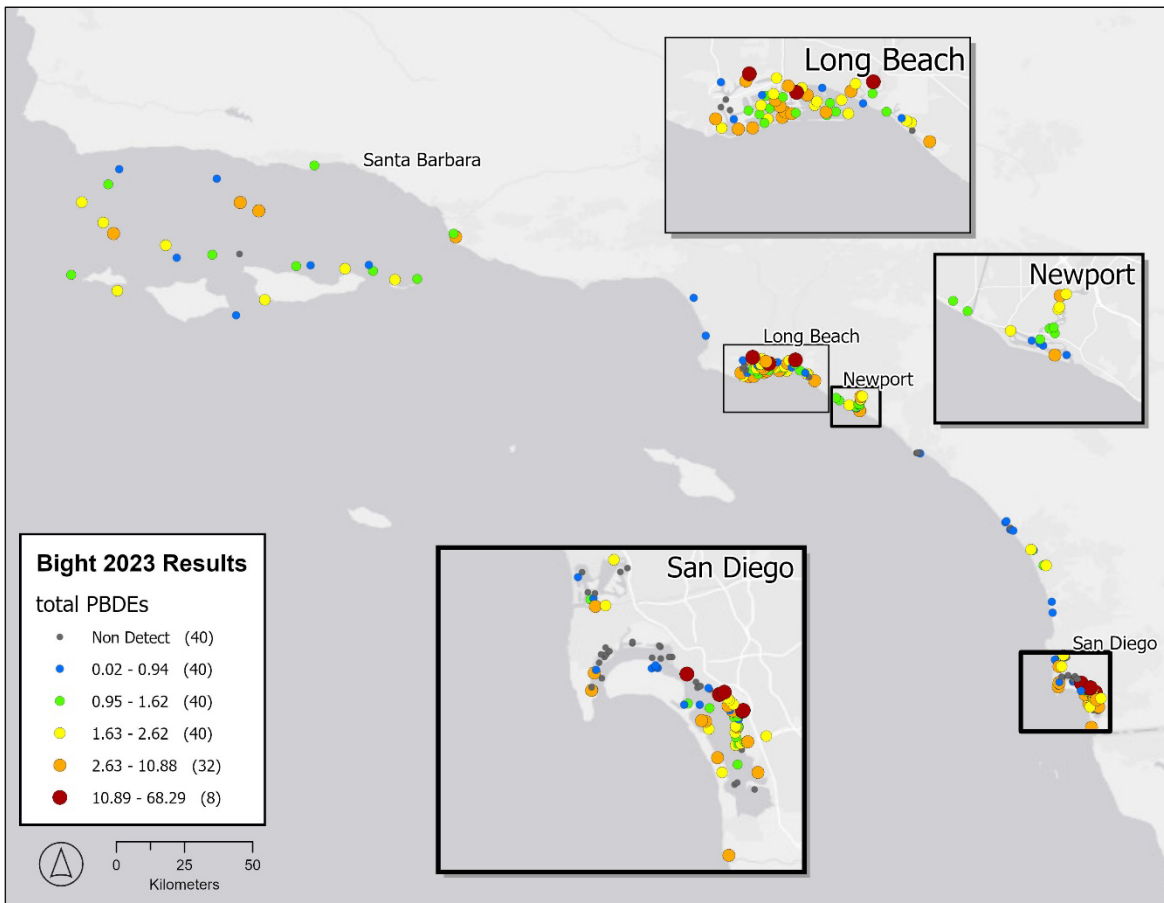


Figure 43: Map of total polybrominated diphenyl ether (total PBDE) concentrations in Bight '23 sediments.

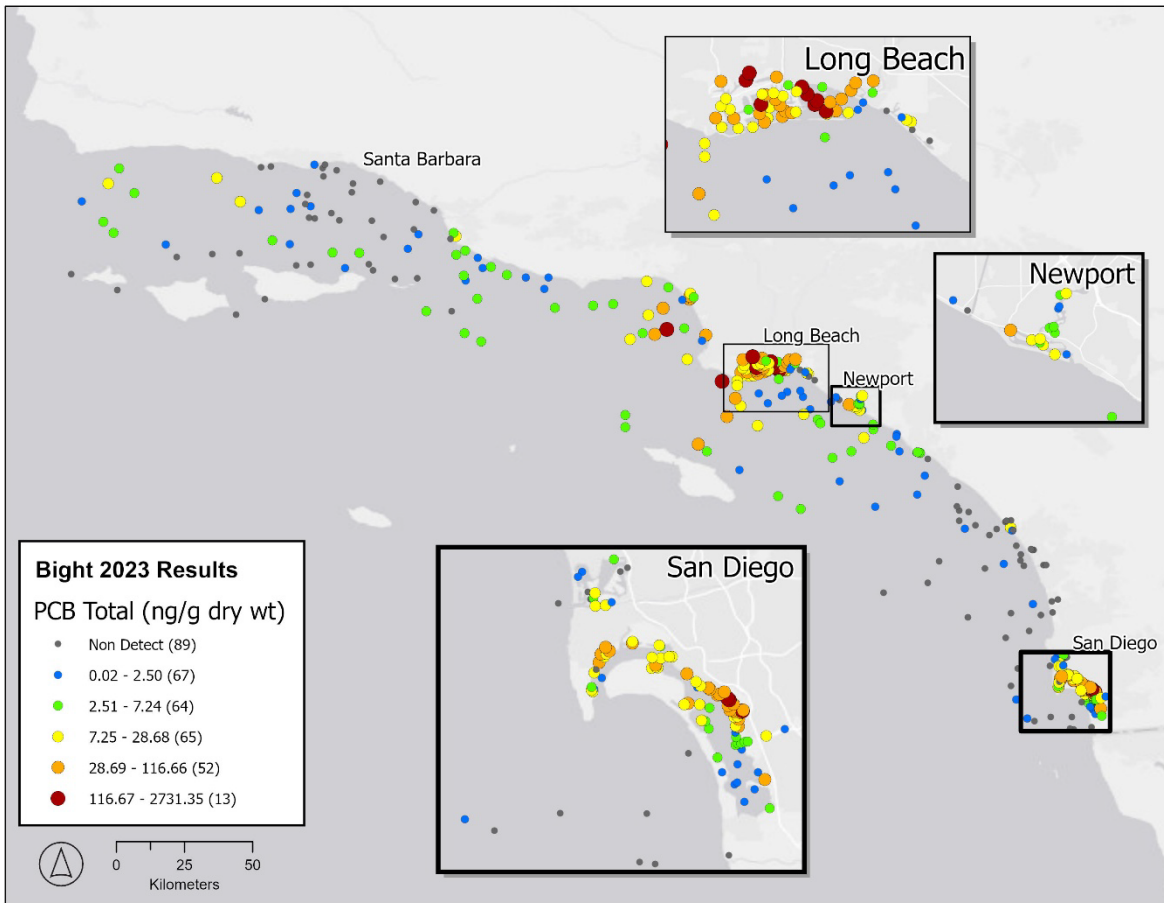


Figure 44: Map of total polychlorinated biphenyl concentrations (total PCBs) in ng/g dry weight in Bight '23 sediments.

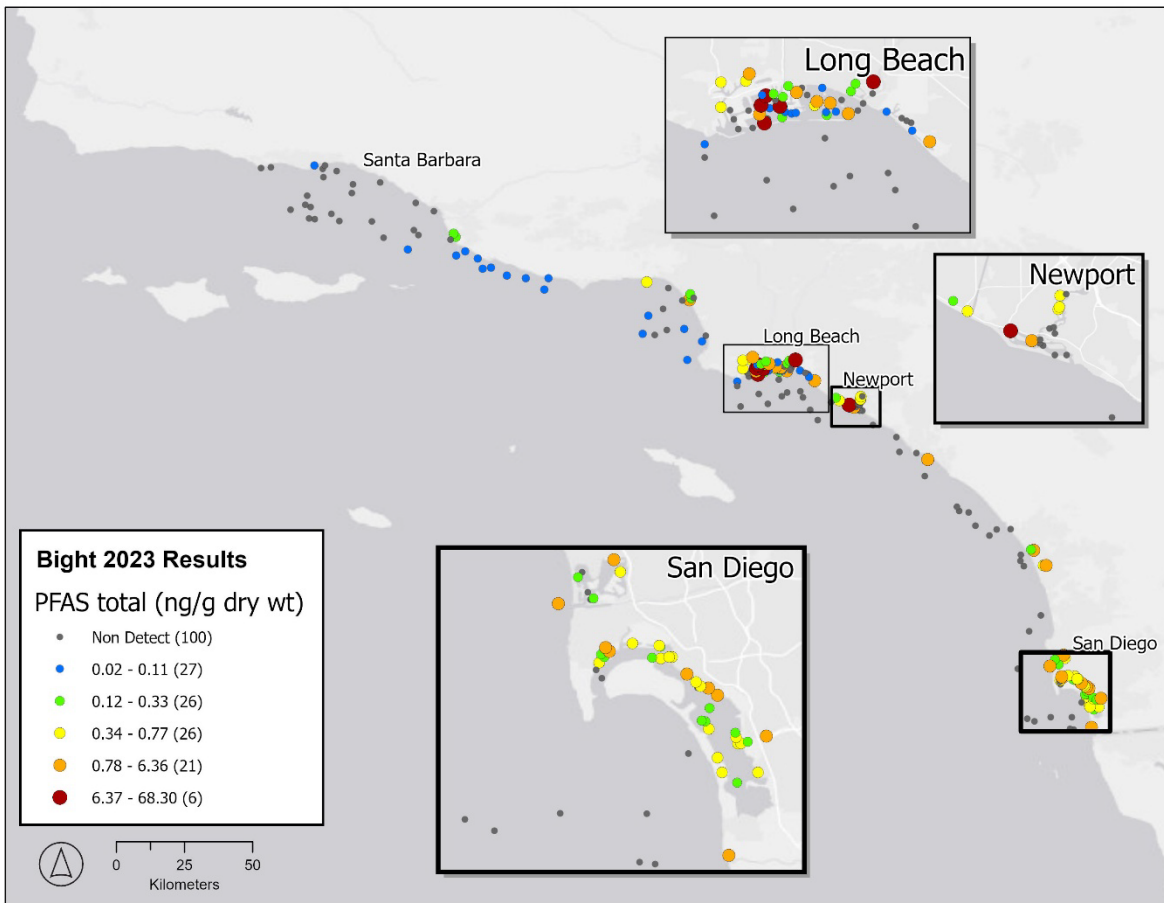


Figure 45: Map of total per- and polyfluorinated alkyl substance (total PFAS) concentrations in ng/g dry weight in Bight '23 sediments.

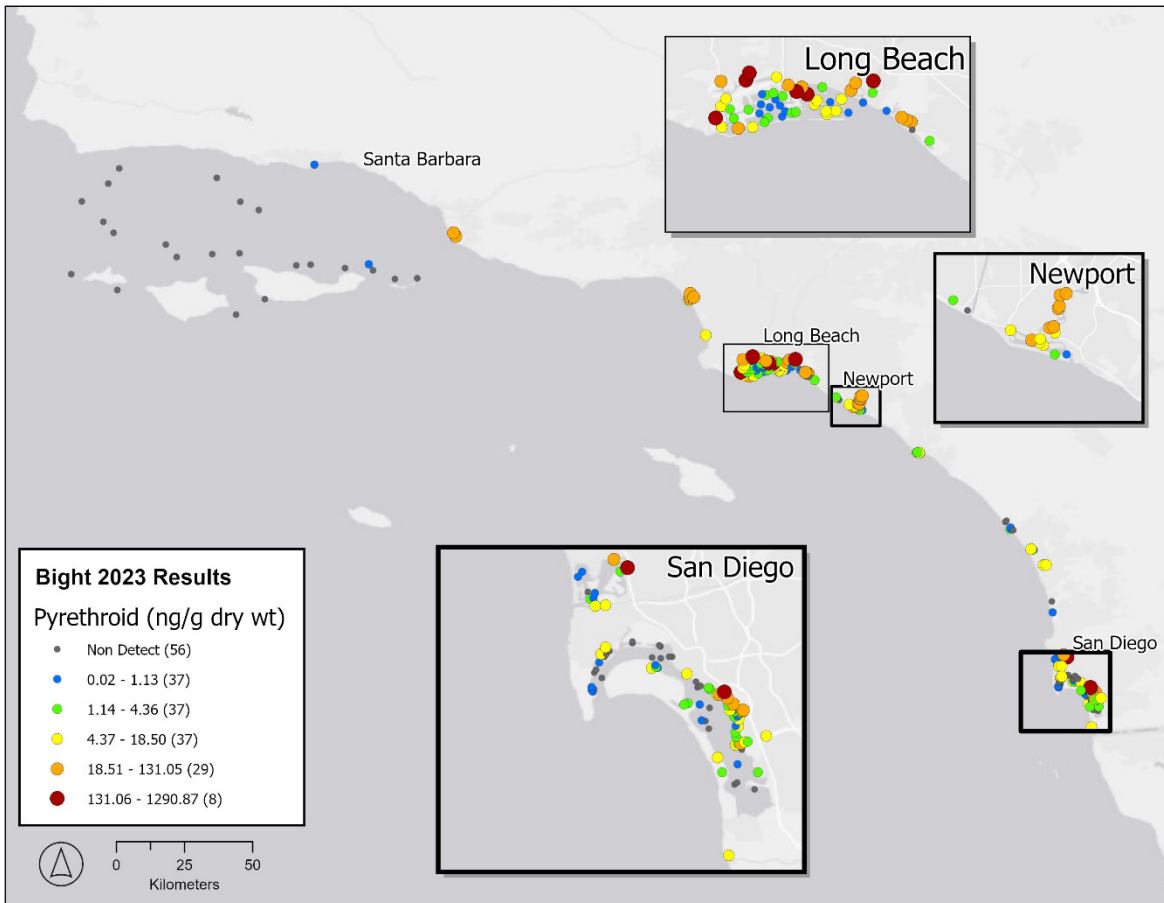


Figure 46: Map of total pyrethroid insecticide concentrations in ng/g dry weight in Bight '23 sediments.

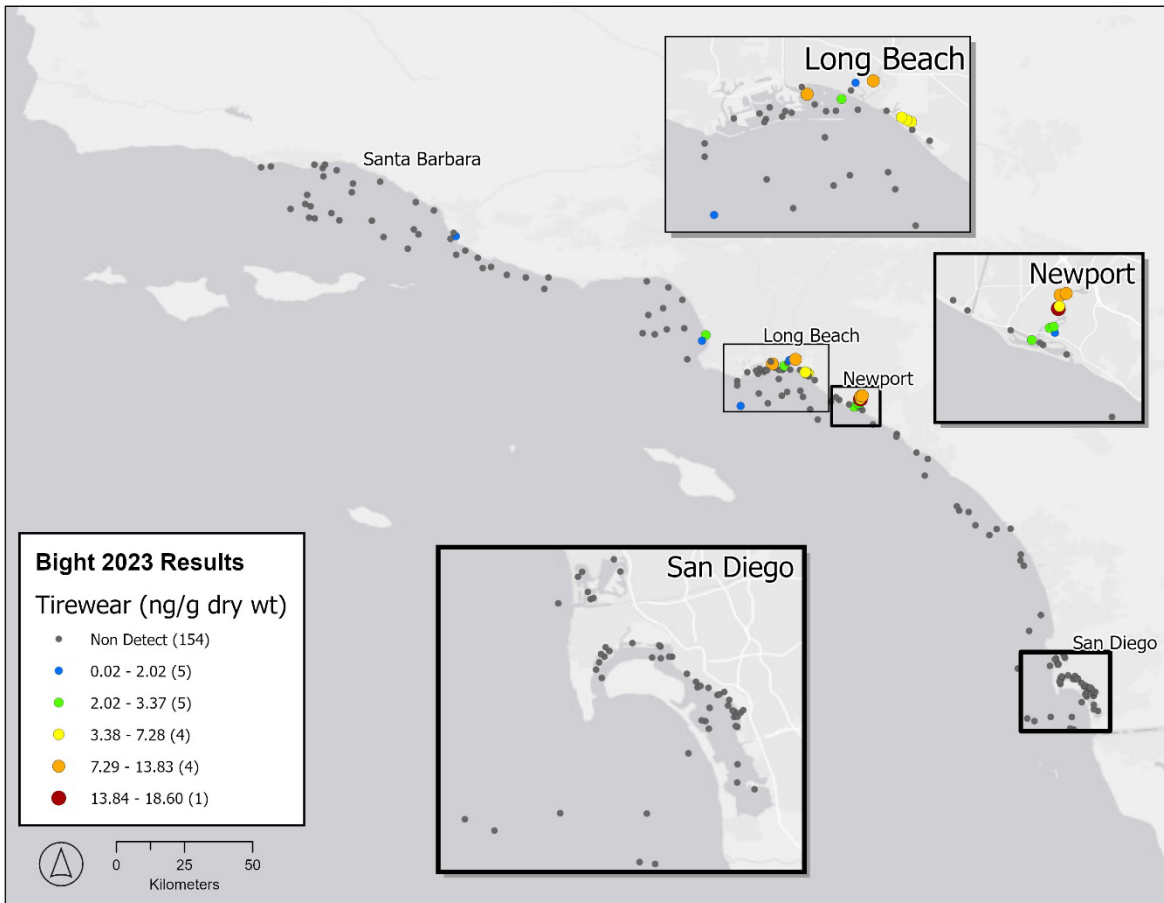


Figure 47: Map of tire-wear compound concentrations (6-PPD-quinone) in ng/g dry weight in Bight '23 sediments.

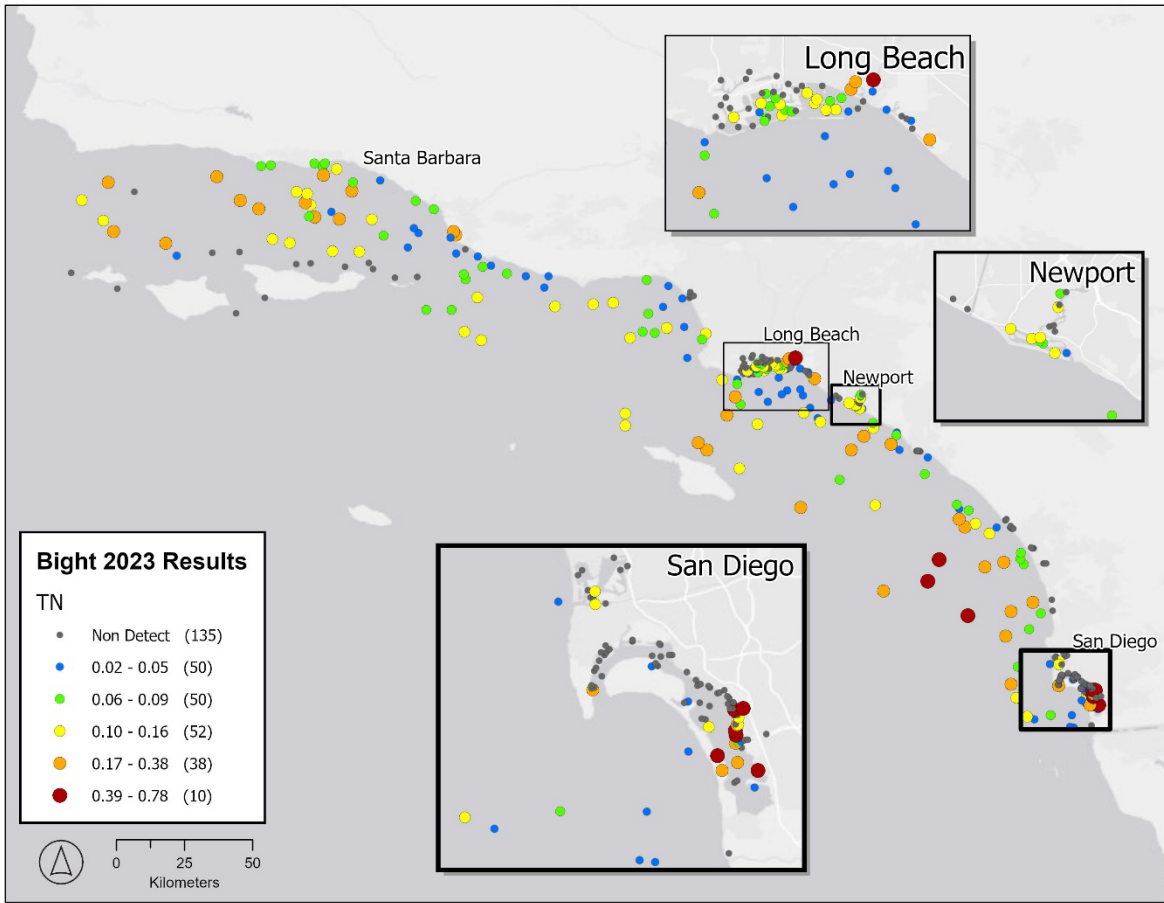


Figure 48: Map of total nitrogen (TN) concentrations as % by weight in Bight '23 sediments.

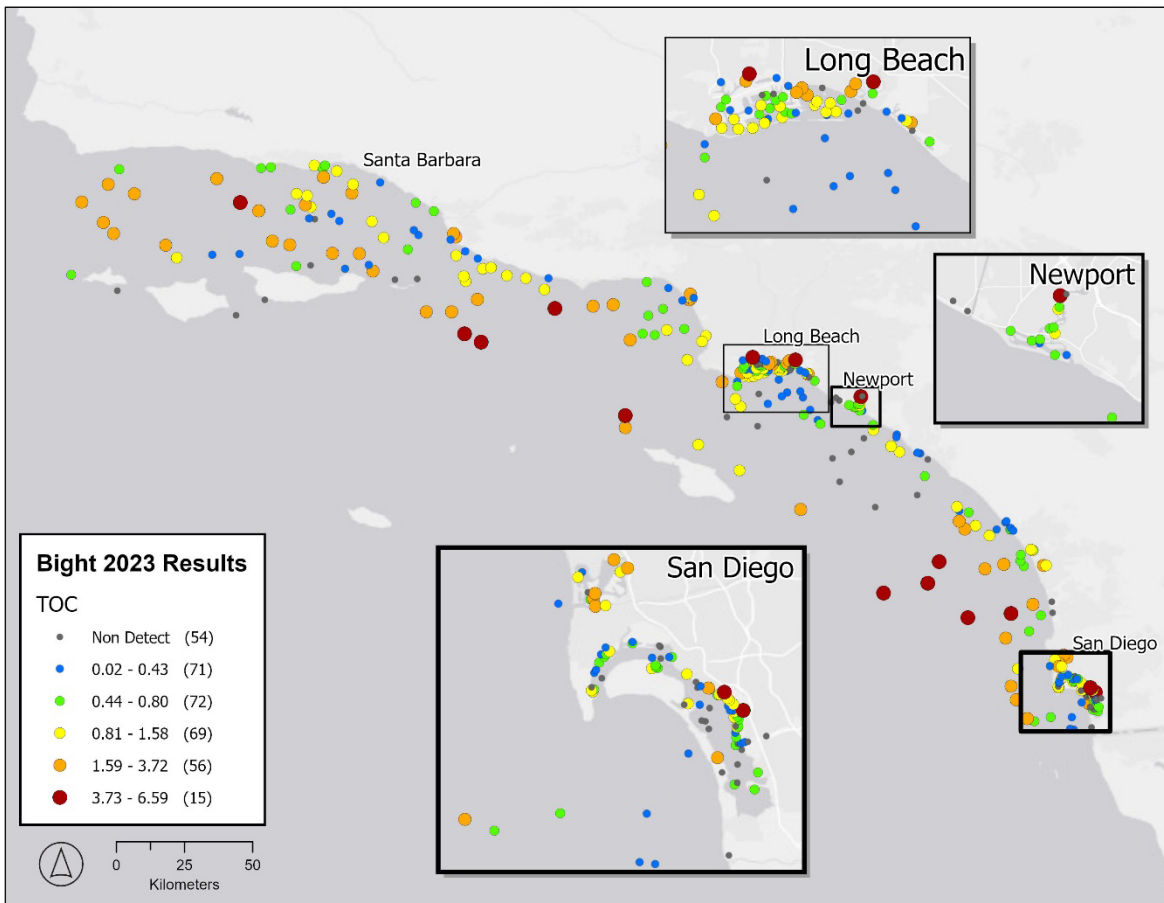


Figure 49: Map of total organic carbon (TOC) concentrations as % by weight in Bight '23 sediments.

APPENDIX D. CUMULATIVE DISTRIBUTION FREQUENCY DIAGRAMS OF CHEMICAL CONCENTRATIONS BY PERCENT OF BIGHT-WIDE AREA

The following plots depict cumulative distribution frequencies for concentrations of Bight '23 analytes and analyte classes by area-weighted percentages (i.e., how much of the area of the overall Bight has AWM concentrations lower than the concentration of interest). Only those analytes which were sampled and analyzed across the entire Bight are included (i.e., neonicotinoids, PBDEs, PFAS, pyrethroids, and tire-wear compounds excluded).

