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Contaminants of Emerging Concern in San Francisco Bay: A Strategy for Future Investigations
2020 Update

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About the Update

The Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) has investigated contaminants of emerging concern (CECs) since 2001. CECs can be broadly defined as synthetic or naturally occurring contaminants that are unregulated or inadequately regulated, not commonly monitored in environmental matrices, and have the potential to enter the environment and cause adverse ecological or human health impacts.

The RMP Emerging Contaminants Workgroup (ECWG), established in 2006, includes representatives from RMP stakeholder groups, regional scientists, and an advisory panel of expert researchers that work together to address the workgroup's guiding management questions (MQs), which are:

- MQ1: Which CECs have the potential to adversely impact beneficial uses in San Francisco Bay?
- MQ2: What are the sources, pathways, and loadings leading to the presence of individual CECs or groups of CECs in the Bay?
- MQ3: What are the physical, chemical, and biological processes that may affect the transport and fate of individual CECs or groups of CECs in the Bay?
- MQ4: Have the concentrations of individual CECs or groups of CECs increased or decreased in the Bay?
- MQ5: Are the concentrations of individual CECs or groups of CECs predicted to increase or decrease in the future?
- MQ6: What are the effects of management actions?

The overarching goal of the ECWG is to develop cost-effective strategies to identify and monitor CECs to minimize impacts to the Bay.

To this end, the RMP first published a CEC Strategy in 2013 (Sutton et al., 2013). A significant revision of the CEC Strategy was completed in 2017 (Sutton et al., 2017), and an Update that described the RMP strategy for monitoring CECs in pathways was published in 2018 (Lin et al., 2018). The Strategy is a living document that guides RMP special studies on CECs, assuring continued focus on the issues of highest priority to protecting the health of the Bay. A key focus of the Strategy is a tiered risk-based framework that guides future monitoring proposals. The Strategy also features a multi-year plan indicating potential future research priorities.

This 2020 CEC Strategy Update is a brief summary document that describes the addition of recently monitored CECs to the tiered risk-based framework. Reviews of findings relevant to San Francisco Bay are provided, as is a discussion of the role of environmental persistence in classifying CECs within the framework.

The 2020 Update also outlines the RMP strategy for evaluating the potential toxicological risks of data-poor contaminants. Identifying toxicological thresholds for CECs should first use the best available *in vivo* data, then *in vitro* data when *in vivo* data are unavailable, and model predictions when *in vitro* data is unavailable. Predictive *in vitro* screening of environmental samples may help identify when mixture effects and/or additional, unmeasured contaminants are a concern for aquatic life.

The Update concludes with the current multi-year plan for RMP special studies on CECs. According to this multi-year plan, a full revision of the CEC Strategy is anticipated in 2021.

RMP Tiered Risk-based Framework: 2020

The RMP assigns CECs monitored in Bay water, sediment, and wildlife to tiers in a tiered risk-based framework (framework in Table 1; CEC tier assignments in Table 2). The degree of concern associated with a particular chemical or chemical class guides both RMP monitoring activities and external management actions, as outlined in Table 1. The criteria listed below were used for placement in each tier (Sutton et al., 2017).

High Concern – Bay occurrence data suggest a high probability of a moderate or high level effect on Bay wildlife (e.g., frequent detection at concentrations greater than the EC_{10}[1]).

Moderate Concern – Bay occurrence data suggest a high probability of a low level effect on Bay wildlife (e.g., frequent detection at concentrations greater than the PNEC[2] or NOEC[3] but less than the EC_{10} or another low level effects threshold).

Low Concern – Bay occurrence data suggest a high probability of minimal effect on Bay wildlife (i.e., Bay concentrations are well below toxicity thresholds and potential toxicity to wildlife is sufficiently characterized).

Possible Concern – Uncertainty in toxicity thresholds suggests uncertainty in the level of effect on Bay wildlife. Bay occurrence data exist; in some cases, they may be constrained by analytical methods with insufficient sensitivity.

Secondary factors may impact tier assignments. In particular, trends in use of the chemical or trends in Bay concentrations may affect the relative level of concern about a CEC. When Bay monitoring data are sufficient to indicate whether levels are declining or increasing over time, this is indicated in the tiered risk-based framework via ▼ and ▲ symbols, respectively. Modified symbols ▽ and △ are used when contaminants are expected to be declining or increasing over time based on information other than monitoring data.

Another secondary factor relevant to classification is the potential for cumulative impacts, including additive or synergistic impacts resulting from exposure to contaminant mixtures. For example, co-exposure to multiple endocrine disrupting compounds that trigger similar impacts to an endocrine pathway can result in cumulative (additive or synergistic) impacts to an organism. Cumulative impacts may arise following co-exposure to members of the same class of CECs, or to commonly co-occurring contaminants from different chemical classes with the same mode of action. Greater levels of concern may be warranted when cumulative impacts are relevant to the evaluation of the relative risk of a CEC or class of CECs.

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[1] EC_{10}, effect concentration where 10% of the population exhibits a response
[2] PNEC, predicted no effect concentration
[3] NOEC, no observed effect concentration

Persistence as a Secondary Factor in Risk Evaluation

Persistence in the environment is one of the secondary factors that may be relevant to classification of a CEC or CEC class within the tiered risk-based framework. Highly persistent contaminants, for example those with half-lives of six months or more across various abiotic matrices (e.g., water, sediment), will accumulate in the environment under a scenario of continuous use. Increasing levels of contamination will lead to an increasing probability of impacts to wildlife, many of which may be unanticipated due to limited toxicity information. Once adverse impacts have been observed and relevant management actions implemented, recovery may take decades or longer.

Persistence is a widely recognized hazard criterion in chemical regulations around the world (Cousins et al., 2019; Matthies and Beulke, 2017). Based on the universal implications for contaminant behavior in the environment, and in particular the likelihood that accumulation of highly persistent contaminants will eventually lead to known and unknown adverse impacts, some have made the argument that high persistence alone is an indicator of risk (Cousins et al., 2019). An important qualification is that exposure is required to elicit adverse impacts; thus, a contaminant must be both highly persistent and bioavailable to wildlife for risk to be present. Biotic metabolism and elimination rates are also important to consider, but persistence or residence time in abiotic media is usually much greater than in biota. Persistence combined with slow rates of elimination by biota, especially top predators, is a worst case situation. Degradates must be evaluated for persistence as well, as in some cases, a degradate may be far more persistent than its parent compound. For example, some polyfluorinated compounds are not persistent themselves, but are precursors that degrade in the environment to highly persistent transformation products including perfluorinated compounds (Sedlak et al., 2018).

Formal consideration of persistence as a secondary factor within the tiered risk-based framework is directly relevant to the class of CECs described by the RMP as “Other PFAS,” consisting of short-chain perfluoroalkyl and polyfluoroalkyl substances (PFAS). Previous review of this class of contaminants focused on the more limited toxicity data relative to long-chain perfluoroalkyl substances. As a result, while long-chain PFAS like perfluorooctanesulfonic acid (PFOS) and perfluorooctanoic acid (PFOA), were classified as Moderate Concern for the Bay, all other PFAS were previously classified as Possible Concern (Sedlak et al., 2018).

However, the high persistence of PFAS as a class suggests this differentiation of potential risks may not be appropriate. For example, while short-chain perfluoroalkyl substances have received less study relative to long-chain versions, these contaminants are still highly persistent and expected to accumulate in the environment. Short-chain compounds are also highly mobile, not amenable to water treatment, and are known to bioaccumulate in plants, though not typically in animals (Brendel et al., 2018). In Europe, there is movement toward regulating PFAS as a class or as subgroups, rather than by individual compounds (Kwiatkowski et al., 2020). Some states are also beginning to regulate and investigate members of the “Other PFAS” class. The State of Michigan is now implementing drinking water standards for seven PFAS, including three

short-chain compounds: perfluorobutane sulfonate (PFBS), perfluorohexanoic acid (PFHxA), and a perfluoroalkyl ether (HFPO-DA, often referred to as GenX). California's State Water Board is now conducting a statewide assessment to determine the scope of contamination by PFAS in water systems and groundwater; sampling will include discharges from wastewater treatment facilities, and a number of short-chain PFAS are required analytes.

Review of the RMP classification of "Other PFAS" with consideration of persistence as a secondary factor suggests that reclassification to the Moderate Concern tier is appropriate. This designation would be consistent with the rapidly evolving scientific and regulatory response to PFAS as a broad class of priority compounds for management actions. Reclassification of all PFAS as Moderate Concern was agreed upon during the 2020 ECWG meeting, and is reflected in the discussion of PFAS subsequently in this Update.

For future use of persistence as a reason to elevate contaminants within the tiered risk-based framework, it is suggested that the RMP use established criteria for defining persistence (e.g., ECHA's definition and weight-of-evidence approach for identifying persistence (ECHA, 2017)). Other factors, such as production and use, should also be considered. In cases where a contaminant or contaminant class is elevated due to persistence, different monitoring and management approaches may be more appropriate than those used for contaminants in the same tier that exceed toxicity thresholds. Persistence may also be a useful characteristic to identify new candidates for monitoring.

Table 1. The RMP Tiered Risk-based Framework for San Francisco Bay. See Sutton et al. 2017 for more information.

	Risk Level Description*	Monitoring Strategy	Water Quality Management Actions
High Concern	Bay occurrence data suggest a high probability of a moderate or high level effect on Bay wildlife.	Studies to support TMDL or alternative management plan.	303(d) listing.** TMDL or alternative management plan.** Aggressive control/treatment actions for all controllable sources.
Moderate Concern	Bay occurrence data suggest a high probability of a low level effect on Bay wildlife.	Consider including in Status and Trends monitoring. Special studies of fate, effects, sources, pathways, and loadings.	Action plan/strategy. Aggressive pollution prevention. Low-cost control/treatment actions.
Low Concern	Bay occurrence data suggest a high probability of no effect on Bay wildlife.	Discontinue or conduct periodic screening level monitoring in water, sediment, or biota. For CECs previously considered Moderate Concern, maintain Status and Trends monitoring for at least two cycles. Periodic screening level monitoring for chemical(s) detected in wastewater or stormwater to track trends.	Low-cost source identification and control. Low-level pollution prevention. Track product use and market trends.
Possible Concern	Potential for concerns or uncertainty in measured Bay concentrations or toxicity thresholds suggest uncertainty in the level of effect on Bay wildlife.	Screening level monitoring to determine presence in water, sediment, or biota. Screening level monitoring for presence in wastewater or stormwater.	Maintain (ongoing/periodic) effort to identify and prioritize emerging contaminants of potential concern. Track international and national efforts to identify high priority CECs. Develop biological screening methods and identify available analytical methods.

*Secondary factors such as persistence, trends in use or presence in the Bay, and potential for cumulative impacts, may affect tier assignments.
**Subject to Regional Water Quality Control Board action with public review.

RMP CEC Tier Assignments: Recent Findings

Recent findings relating to contaminants assigned to the RMP tiered risk-based framework for CECs are summarized below. These include newly designated Moderate Concern contaminants bisphenols, organophosphate esters, and imidacloprid; previously established Moderate Concern contaminants PFOS, PFOA and long-chain carboxylates; Low Concern contaminant galaxolide (a member of the class described as personal care and cleaning product ingredients); as well as Possible Concern contaminants quaternary ammonium compounds (QACs), indole, and 4-methylphenol. A discussion of the designation of microplastics as a Moderate Concern contaminant is provided in the RMP Microplastic Strategy Update (Sedlak et al., 2019). In addition, a rationale for screening level monitoring of brominated and chlorinated azo dyes is provided. These groups represent the only significant changes to the tiered risk-based framework since the 2018 CEC Strategy Update (Lin et al., 2018).

The tier assignments for each CEC in this report were based on available information and will be updated annually as new information on the levels or potential risk of the CEC becomes available. The rationale behind the assignments of these CECs, as well as previously evaluated and assigned CECs, are provided in Table 2.

At this time, no CECs are considered to be a High Concern for the Bay. For information on contaminants assigned to tiers but not discussed in this Update, see Sutton et al. (2017) and Lin et al. (2018).

Table 2A. Current status of CECs in the tiered risk and management action framework for San Francisco Bay (Moderate Concern).

	Contaminant Class	Trend[1]	Current Bay Data
Moderate Concern	Alkylphenols and Alkylphenol Ethoxylates		Ubiquitous in Bay water, sediment, bivalves, fish, bird eggs, with concentrations below most toxicity thresholds; possible impacts on larval barnacle settlement; possible synergistic effects with pyrethroids; estrogenic activity; previously high volume use in laundry detergent may be decreasing following phase-out.
	Bisphenols	△	BPA and BPS detected in Bay water in the range of BPA toxicity threshold; potential adverse estrogenic effects and potential for cumulative impacts; projected increase in production and use worldwide for BPA and alternatives.
	Fipronil	▽	Sediment concentrations in the range of toxicity thresholds for degradates; use has increased over the last several years and is high in urban areas; however, mitigation measures to reduce outdoor use in California were announced in 2017 and are expected to result in significant declines in environmental contamination.
	Imidacloprid	△	Lower South Bay water concentrations in the range of toxicity thresholds; significant discharges via Bay Area stormwater and wastewater; limited chronic toxicity data in marine species; increasing use in urban areas.
	Microplastics	△	Ubiquitous in Bay water, sediment, prey fish, bivalves; high concentrations in stormwater, also observed in wastewater; uncertainty in toxicity to Bay wildlife; extremely persistent in environment and difficult to clean up; increasing plastic use and discharge globally; EU proposes considering microplastics a non-threshold contaminant, meaning any discharge poses risk.
	Organophosphate Esters		Detection of several in water, sediment, and tissue; levels of TDCPP in water comparable to or exceeding conservative ecotoxicity threshold; endocrine disrupting properties; potential for cumulative effects; high volume and potentially increasing use as PBDE replacements, though California's AB 2998, which bans flame retardants in foam furniture in 2020, may cause decline.
	PFOS	▲	Bird egg concentrations have exceeded a PNEC and are currently in the range of concentrations linked to reproductive effects in wild birds; possible risks to humans who frequently eat Bay sport fish; high concentrations in seal blood; high volume use of precursors; recent monitoring suggests a decline in bird eggs.
	PFOA and Long-chain Perfluoro-carboxylates		While PFOA and some long-chain carboxylates appear to be declining in bird eggs, others are not; declines have not been observed in seals; concentrations in seal blood and bird eggs from South Bay are in the range of concentrations linked to disruption in gene functions in Russian seals and potential reproductive effects in birds; cumulative effects expected; further information on PFAS in stormwater and wastewater pathways is expected.

[1]Solid triangles indicate sufficient Bay monitoring data to indicate a temporal trend, while hollow triangles indicate an expected temporal trend based on information other than monitoring data, such as changes in use or new regulations.

Table 2B. Current status of CECs in the tiered risk and management action framework for San Francisco Bay (Low Concern).

	Contaminant Class	Trend[1]	Current Bay Data
Low Concern	HBCD	▽	Low concentrations measured in sediment, bird eggs, fish, harbor seals, bivalves; reduction in use anticipated globally; clear declines not yet observed in the Bay; monitoring deprioritized.
	PBDD/Fs	▽	Low concentrations; synthetic sources declining with PBDE phase out; natural sources not expected to change; monitoring deprioritized.
	PBDEs	▼	Concentrations in Bay wildlife and sediment have been thoroughly studied and decreased over time, with detections now typically below thresholds of potential concern; tern egg concentrations are below reproductive toxicity threshold; sport fish concentrations are below protective human health thresholds for fish consumption; uncertainty regarding impacts on harbor seals; production and use phased out in US; Status & Trends monitoring is recommended for at least two more cycles in sediment, sport fish, and bird eggs to track declines.
	Personal Care and Cleaning Products (10+ monitored e.g., antimicrobials, galaxolide and other fragrances)	△	Concentrations below toxicity thresholds, toxicity to aquatic species sufficiently characterized; levels expected to increase with population; many other personal care analytes have yet to be monitored; periodic monitoring is recommended, particularly for as-yet unevaluated ingredients such as sunscreens.
	Pharmaceuticals (100+ monitored e.g., ibuprofen, sulfamethoxazole)	△	Concentrations below toxicity thresholds; toxicity to aquatic species sufficiently characterized; levels expected to increase with population; many other pharmaceutical analytes have yet to be monitored; periodic monitoring is recommended.
	Pyrethroids		Detected infrequently and at low levels in open Bay and margin sediment; of High Concern in watersheds, as tributary sediment concentrations exceed toxicity thresholds; lower impact professional application methods have been prescribed via state regulations; Bay monitoring deprioritized.

[1]Solid triangles indicate sufficient Bay monitoring data to indicate a temporal trend, while hollow triangles indicate an expected temporal trend based on information other than monitoring data, such as changes in use or new regulations.

Table 2C. Current status of CECs in the tiered risk and management action framework for San Francisco Bay (Possible Concern).

	Contaminant Class	Trend[1]	Current Bay Data
Possible Concern	4-methylphenol		Detected in sediment samples below reporting limits in the South Bay Margins; uncertainties in available toxicity data; periodic monitoring in conjunction with other fragrance ingredients is recommended.
	Additional Contaminants Identified via Non-targeted Analysis		Tissue study [2010]: 2,2'-dichlorobenzil, dichloroanthracenes; Water & wastewater study [2016]: triphenylphosphine oxide, erucamide Detected in Bay samples; uncertainties in concentrations and toxicity data.
	Alternative Flame Retardants - Hydrophobic Brominated Compounds	△	Detection of several in sediment and tissue; limited toxicity data for aquatic species; additive/synergistic exposure effects unknown; high volume and potentially increasing use as PBDE replacements; a future Special Study is recommended.
	Alternative Flame Retardants - Hydrophobic Chlorinated [Dechlorane] Compounds		Detection of Dechlorane Plus and a few related compounds in sediment and tissue; limited toxicity data for aquatic species; additive/synergistic exposure effects unknown; high volume use; monitoring deprioritized.
	BZT-UVs		Water and sediment concentrations below available toxicity thresholds; uncertainties in toxicity data; awaiting stormwater data from multi-year CECs screening study.
	Indole	△	Sediment concentrations in the South Bay Margins at levels greatly exceeding toxicity thresholds, though there are uncertainties in available thresholds; uncertainties in toxicity to Bay wildlife; levels expected to increase with population; periodic monitoring in conjunction with other fragrance ingredients is recommended.
	PCB 11 (non-Aroclor PCB)		Ubiquitous contaminant and has been detected in Bay water, urban runoff, sediments; not bioaccumulative (minor congener in small fish and bivalves); uncertainty in toxicity thresholds; monitoring via PCB Status and Trends, Special Studies is recommended. Additional PCBs with non-Aroclor sources that may be appropriate to evaluate include PCB 47, 51, 68, 206, 207, 208, 209.
	Other Current Use Pesticides		Concentrations below toxicity thresholds with the exception of carbendazim in margin sediment; uncertainty in toxicity to Bay wildlife; future Special Study should include wet season monitoring.

Possible Concern	Other PFASs – Short-chain Perfluoroalkyl Substances, Polyfluoroalkyl Substances	△	Detection of several compounds in Bay matrices; indications of contamination with as-yet unidentified PFASs; potential for increased use as alternatives to PFOS and PFOA; toxicity to aquatic species not sufficiently characterized; monitoring via Status and Trends and Special Studies is a high priority.
	Plastic Additives - Phthalates (e.g., DEHP,BBZP)		Sediment concentrations in the same range as low apparent effects threshold (but threshold not directly linked to DEHP); BBZP sediment concentrations exceed low apparent effects threshold (threshold not directly linked to BBzP or effects in macrobenthos); future Special Study recommended.
	Polyhalogenated Carbazoles		Ubiquitous contaminants detected in Bay sediment, bivalves, fish, birds, and seals; uncertainty in toxicity thresholds.
	QACs	△	Several detected in Bay sediment; sediment core shows use of some staying roughly the same while others are decreasing over time; uncertainty in toxicity to Bay wildlife; expected to increase with population and response to the COVID-19 outbreak; periodic monitoring is recommended.
	Rare Earth Elements, Gadolinum	▲	Detected at low concentrations in ambient Bay sediment and water; gadolinium only detected in water; concentrations below limited available toxicity thresholds; gadolinium concentrations increasing over time; awaiting pro bono data on water and effluent.
	Roadway Contaminants		Identified via non-targeted analysis; uncertainty in toxicity thresholds; awaiting stormwater data from multi-year CECs study.
	Short-chain Chlorinated Paraffins		Concentrations below toxicity thresholds; uncertainties in toxicity data; high volume use; monitoring deprioritized; medium- and long-chain versions are yet to be characterized and may be prioritized in the future.
	SDPAs		Water and sediment concentrations below available toxicity thresholds; uncertainties in toxicity data; awaiting stormwater data from multi-year CECs screening study.
	Siloxanes		Detected in bivalves; uncertainty in bioaccumulation potential; some sediment and water toxicity thresholds available; awaiting pro bono data on sediment and effluent.
	Single-walled Carbon Nanotubes		Not detected; analytical limitations; toxicity information not available; monitoring deprioritized.

[1]Solid triangles indicate sufficient Bay monitoring data to indicate a temporal trend, while hollow triangles indicate an expected temporal trend based on information other than monitoring data, such as changes in use or new regulations.

Bisphenols (Moderate Concern)

Bisphenols are a group of synthetic compounds containing two hydroxyphenyl functional groups joined by a carbon or sulfur bridge. Bisphenols are stabilizing agents and are widely produced in Asia, Europe, and the US, primarily for use as plastic additives, thermal reactants, and color developers in industrial and consumer products (Björnsdotter et al., 2017; Wu et al., 2018). Such products include polycarbonate plastics used in medical devices, car parts, water bottles, tableware, and cookware; epoxy resins, varnishes, and food can linings; additives in textiles, clothing, printing inks, and flame retardants; and in paper products including receipts, food packaging materials, paper towels, and currency (Björnsdotter et al., 2017; Chen et al., 2016; Heemken et al., 2001; Xue et al., 2017). The tendency for bisphenols to migrate from products into wastewater and stormwater has been demonstrated by bisphenol detections in these pathways (Björnsdotter et al., 2017; Boyd et al., 2004; Fairbairn et al., 2018; Jackson and Sutton, 2008; Vidal-Dorsch et al., 2012).

Due to their durability, non-corrosivity, stability, and many other desirable chemical properties, their manufacture has steadily increased since their industrial popularity began in the 1930s (Rochester and Bolden, 2015). Historically, bisphenol A (BPA) has had the highest global production volume of all bisphenols, followed by bisphenol S (BPS). Bisphenol A and S are produced and imported in the US at volumes of billions and millions of pounds, respectively (USEPA [United States Environmental Protection Agency], 2016). In recent years, BPS and other bisphenols have become substitutes for BPA as BPA toxicity findings have spurred public concern and policy changes such as the US Food and Drug Administration's 2012 ban of BPA in baby bottles.

Serving as home to over seven million inhabitants and a breadth of industry, the San Francisco Bay Area is an excellent location to screen for the presence of bisphenols in the environment. As part of the RMP biennial 2017 water cruise, 22 water samples were collected throughout the Bay in the dry season for analysis of 16 bisphenols. These included bisphenols A, AF, AP, B, BP, C, C-dichloride, E, F, G, M, P, PH, S, TMC, and Z.

Two of the 16 bisphenols, BPA and BPS, were detected in both the dissolved and particulate phase. While BPF was detected in some samples as well, contamination of the field blank limited confidence in the data, so these quantifications were censored. Other bisphenols were not detected. Total concentrations of BPA (sum of particulate and dissolved contributions) were detected at 91% of the 22 sites (<0.7–35 ng/L) with increased concentrations in Lower South Bay. Total concentrations of BPS were detected at 41% of the 22 sites (<1–120 ng/L) and typically found at lower levels than BPA. Summed concentrations of BPA and BPS across all sites ranged from <MDL–160 ng/L. Bisphenol A is imported into and manufactured in the US at approximately 1000 times the amount of BPS (USEPA, 2016), but BPS was found at somewhat comparable levels which is likely due to the higher persistence of BPS in aquatic environments (Danzl et al., 2009).

Levels of BPA in San Francisco Bay were within the broad range of concentrations observed in several estuarine and marine environments, a range that varies by four to five orders of magnitude. A review of 32 North American studies from 1996-2004 found the 95th percentile for BPA to be 24 ng/L in marine waters (Staples et al., 2018). Environmental monitoring data for other bisphenols in similar environs were scarce.

Studies have shown that BPA demonstrates a multitude of adverse estrogenic effects and, in some cases, with potencies comparable to naturally occurring hormones (Björnsdotter et al., 2017; Chen et al., 2016; Héliès-Toussaint et al., 2014; Rosenmai et al., 2014). In addition to endocrine-disrupting effects, BPA has also been linked to cytotoxicity, genotoxicity, mutagenicity, neurotoxicity, cancer, obesity, reproductive and developmental effects, miscarriages, and immunological effects (Björnsdotter et al., 2017; Carlisle et al., 2009). It is listed on California's Proposition 65 List for developmental toxicity and female reproductive toxicity (OEHHA [Office of Environmental Health Hazard Assessment], 2019). Although numerous toxicity studies have been conducted for BPA, conflicting data, questionable data curation, and non-monotonic dose-response curves have led to the generation of multiple predicted no effect concentrations (PNEC) in the literature. The European Chemicals Association's determination of a marine PNEC of 150 ng/L (ECHA, 2011) excluded the most sensitive species (Oehlmann et al., 2008). In response, Wright-Walters et al. (2011) derived a more sensitive PNEC of 60 ng/L. More recent derivation of PNECs using more advanced statistical techniques and both traditional and non-traditional (e.g., responses at the cellular biomarker and genome levels) endpoints have, perhaps surprisingly, resulted in higher PNECs (Wang et al., 2018).

Popular "BPA free" disclaimers on products imply that other BPA alternatives are, presumably, "safer" alternatives. However, they are structurally similar to BPA, and some have demonstrated links to the same array of toxic effects at similar, and sometimes greater, potencies (Chen et al., 2016; Mu et al., 2018; Naderi et al., 2014; Rochester and Bolden, 2015; Rosenmai et al., 2014). Preliminary marine predicted no effect concentrations have also been derived for BPAF (520 ng/L), BPC (240 ng/L), BPS (27,000 ng/L), and BP-TMC (500 ng/L) as part of REACH registrations (ECHA, 2018a, 2018b, 2012, 2010a, p. 4). Comparison of these thresholds with the 2017 water cruise results seems to indicate these compounds are presently a lower concern than BPA. However, these PNECs are based on limited freshwater data and therefore have inherently high uncertainty. Considering their structural similarities, bisphenols are expected to have overlapping modes of action and possibly exhibit additive toxicity; a protective read-across approach of applying Wright-Walter et al.'s (2011) weight-of-evidence PNEC is therefore warranted. Predicted no effect concentrations have not been derived for any of the other bisphenols analyzed by the RMP.

Prior to 2019, bisphenols had been classified as a Possible Concern for the San Francisco Bay within the RMP tiered risk-based framework. Analysis of 2017 Bay water revealed presence of BPA and BPS at levels comparable to Wright-Walter et al.'s (2011) weight-of-evidence PNEC for BPA of 60 ng/L, indicating an appropriate classification for this chemical class is Moderate

Concern for the Bay. Secondary factors relevant to this classification include observed increases in bisphenols in environmental matrices elsewhere (Xu et al., 2015), and a projected increase in production of all bisphenols (Rochester and Bolden, 2015).

RMP Monitoring Strategy: Continued observations of bisphenols in Bay water through regular monitoring is recommended to track trends in light of projected increases in use across the class. In addition, temporal trends might reveal changes in the mixtures of bisphenols in the environment, which might be anticipated based on observations of substitution of other bisphenols for BPA in some products.

However, the anticipated presence of bisphenols in both stormwater and wastewater pathways suggests that the current Status and Trends monitoring design for water samples, in which sample collection occurs during the dry season, may not provide representative concentration data for these contaminants. A pilot study to assess occurrence during both dry and wet seasons is recommended.

Screening of bisphenols in wastewater and stormwater is now underway to fill data gaps in our conceptual understanding of the relative importance of these pathways in contributing loads of these contaminants to the Bay.

Screening of bisphenols in South Bay margin sediment is also underway. The partitioning and persistence of bisphenols in environmental matrices varies based on their unique individual chemical properties and by matrix. The logK_{ow}s of the 16 bisphenols monitored by the RMP range from 1.2 (BPS) to 7.2 (bisphenol PH) (EPIWEB 4.1), suggesting a wide range in tendency to adsorb to sediment. Some bisphenols, such as BPA, have been shown to photo- and biodegrade readily in marine environments, while others like BPS have demonstrated moderate persistence (Björnsdotter et al., 2017). Some, such as BPA-DGE and BPAF, are predicted to have higher persistence in water, according to the US EPA's Persistent, Bioaccumulative and Toxic Profiler (PBT Profiler). Similarly, lab experiments have demonstrated slow degradation for BPA, BPE, BPB, and BPS under anaerobic conditions, such as in anoxic estuarine sediment (Ike et al., 2006; Voordeckers et al., 2002) and the US EPA's PBT Profiler predicted BPA-DGE, BPAF, BPAP, BPB, BPC, BPF, BPAP, BPS to be "persistent" or "very persistent" in sediment. Thus, while only BPA and BPS were observed in water monitoring, the occurrence and distribution of bisphenols in sediment may be quite different.

An exploratory screening of bisphenols in biota is also recommended. While some bisphenols, such as BPA, are known to show low potential for bioaccumulation, studies have found elevated potential for some bisphenols to bioaccumulate into food webs (Wang et al., 2017). This may become increasingly relevant as BPA replacements continue.

Organophosphate Esters (Moderate Concern)

Organophosphate esters (OPEs) are a group of synthetic, mobile, and water-soluble compounds, each composed of a different combination of alkyl-, aryl-, or halogen-containing functional groups. Organophosphate esters are known primarily for their use as flame retardants, and also provide other functions as plastic additives. Since the 1970s, manufacturers have abided by various voluntary and regulatory flammability standards, incorporating flame-retardant additives in a wide array of consumer and industrial products, including upholstered foam furniture, synthetic textiles, foam building insulation, and plastic housings for electronics and appliances (Cooper et al., 2016; Klosterhaus et al., 2012; Rodgers et al., 2018). In the 2000s, use of OPEs as flame retardants products increased with the country-wide phase-out of specific mixtures of polybrominated diphenyl ethers (PBDEs) (Cooper et al., 2016; Rodgers et al., 2018; Stapleton et al., 2012).

In addition to flame retardants, OPEs are used as plastic and hydraulic-fluid additives, antifoaming agents, and lacquer and floor polish ingredients (Li et al., 2017; Rodgers et al., 2018). Globally, they are produced at high volumes, with 2015 US aggregate production and import volumes for TCPP and TPhP (acronyms defined in Table 3) in the tens of millions and millions of pounds, respectively (USEPA, 2016). Use of OPEs has drastically increased in recent years and is projected to continue expanding (Greaves and Letcher, 2017).

Due to their ubiquitous presence in a variety of consumer products, OPEs are frequently found in indoor dust and air (Rodgers et al., 2018; Stapleton et al., 2012; Wei et al., 2015). They can enter the environment via multiple routes, including volatilization, particle abrasion, or leaching from consumer products like textiles, insulation, and building and decorative materials that, when washed or exposed to rainwater, result in discharges of OPEs to sewers and storm drains, respectively (Li et al., 2017; Saini et al., 2016; Wei et al., 2015). Hydrophilic compounds like OPEs are often more difficult to remove during wastewater treatment processes and can also be transported to the Bay through stormwater; OPEs have been detected in Bay wastewater treatment plant effluent and stormwater (Sutton et al., 2019). Organophosphate esters are highly mobile contaminants and have been detected in remote locations, indicating their capacity for long-range transport (Li et al., 2017; McDonough et al., 2018).

In 2013, a screening study of flame retardants in surface water, sediment, bivalves, and harbor seal blubber was conducted to gain a snapshot of environmental exposure to brominated, chlorinated, and organophosphate ester flame retardants (Sutton et al., 2019). Eleven of 13 OPEs analyzed were detected in 12 water samples (sum of OPE concentrations [including dissolved and particulate contributions] 170–5100 ng/L).

In 2017, more extensive monitoring was conducted on samples collected from 22 Bay sites. Thirteen of 22 OPEs were detected in the dissolved phase, and 12 of 22 were detected in the particulate phase. TnBP, TEP, TiBP, TDCPP, TCPP, and TCEP were detected in all samples. The sum of all 22 OPEs ranged from 35–290 ng/L (median 100 ng/L) across all Bay sites. TCPP

was observed at the highest levels across the Bay, with total concentrations (sum of particulate and dissolved contributions) ranging from 15 to 150 ng/L. The spatial variability for many OPEs was similar, with higher levels in Lower South Bay. Contributions of OPEs via wastewater and stormwater pathways disproportionately influence Lower South Bay due to the smaller water volume relative to discharges and longer residence time.

Concentrations of individual OPEs detected in the Bay in 2017 were on the higher end of available monitoring data in estuarine or marine environments. In a comparison of Bay concentrations to results from 12 similar water bodies, Bay TCPP concentrations (15–150 ng/L, mean 65 ng/L) were only exceeded by the Elbe River Estuary (31–310 ng/L; mean 93 ng/L; Bollmann et al., 2012), and TDCPP concentrations (2.8–23 ng/L; mean 9.0) were similar to those seen in Long Island, New York (8.9–25 ng/L; mean 16 ng/L; Kim and Kannan, 2018), and in Maizuru Bay, Japan (12–25 ng/L; mean 18 ng/L; Harino et al., 2014).

Table 3. Twenty-two OPEs analyzed in 2017 ambient Bay Water.

Abbreviation	**Organophosphate Ester**
V6	Tetrakis(2-chloroethyl) dichloroisopentyl diphosphate
TCEP	Tris(2-chloroethyl) phosphate
TCPP	Tris(1-chloro-2-propyl) phosphate
TDCPP	Tris(1,3-dichloro-2-propyl) phosphate
TDBPP	Tris(2,3-dibromopropyl) phosphate
BPA-BDPP	Bisphenol A bis(diphenyl phosphate)
RBDPP	Resorcinol bis(diphenyl phosphate)
T2iPPP	Tris(2-isopropylphenyl) phosphate
T35DMPP	Tris(3,5-dimethylphenyl) phosphate
BPDPP	Tertbutylphenyl diphenyl phosphate
2iPPDPP	2-isopropylphenyl diphenyl phosphate
CrDPP	Cresyl diphenyl phosphate
TCrP	Tricresyl phosphate
IDDPP	Isodecyl diphenyl phosphate
EHDPP	2-Ethylhexyl-diphenyl phosphate
TPhP	Triphenyl phosphate

TBEP	Tris(2-butoxyethyl) phosphate
TEHP	Tris(2-ethylhexyl) phosphate
TiBP	Triisobutyl phosphate
TPrP	Tripropyl phosphate
TnBP	Tri-n-butyl phosphate
TEP	Triethyl phosphate

Previously, OPEs had been classified as a Possible Concern for the San Francisco Bay within the RMP tiered risk-based framework. Water, sediment, and tissue monitoring for OPEs in 2013 revealed the presence of OPEs in the Bay at levels suggesting some concern might be warranted for TDCPP and TPhP with respect to European Chemicals Agency PNECs for marine waters (ECHA (European Chemicals Agency), 2019a, 2019b). In some cases, total TDCPP concentrations in water (14–450 ng/L; median 33 ng/L) exceeded its marine PNEC (20 ng/L) (ECHA, 2019a) by more than an order of magnitude. Likewise, total TPhP concentrations in water (41–360 ng/L; median 90 ng/L) approached its marine PNEC (370 ng/L) at a few sites (ECHA, 2019b).

More extensive monitoring in 2017 revealed that total concentrations of TDCPP (2.8–23 ng/L; median 6.2 ng/L) met or exceeded the 20 ng/L PNEC at three sites in Lower South Bay.

Based largely on the presence in the Bay of TDCPP at levels comparable to or exceeding conservative ecotoxicity thresholds, OPEs are now considered a Moderate Concern for the Bay. Secondary factors relevant to this classification include widespread and increasing use of OPEs, and concerns around cumulative toxic effects caused by simultaneous exposure to multiple members of the OPE class. Though OPE toxicity is not well understood, endocrine-disrupting effects have been demonstrated at environmentally relevant levels (Bollmann et al., 2012; Harino et al., 2014; Venier et al., 2014). Exposure to mixtures of endocrine-disrupting compounds with the same modes of action can result in additive or even synergistic impacts to organisms. In addition, OPEs have also been linked to cancer, neurotoxicity, and adverse effects on fertility (Wei et al., 2015), and three OPEs—TCEP, TDCPP, TDBPP—are listed as carcinogens on California's Proposition 65 List (OEHHA, 2019).

RMP Monitoring Strategy: Inclusion of OPEs in routine RMP Status and Trends water monitoring would provide useful information on spatial and temporal trends. Use of OPEs as flame retardants in foam furniture increased dramatically following a statewide ban and nationwide phase-out of PBDEs (Stapleton et al., 2012; Cooper et al., 2016; Rodgers et al., 2018); however, with the later passage of a state law (AB 2998) banning flame retardants in foam furniture taking effect in 2020, this use should decline. Trend monitoring will provide

information as to whether this management action results in a meaningful reduction in Bay concentrations.

However, the documented presence of organophosphate esters in both stormwater and wastewater pathways (Sutton et al. 2019) suggests that the current Status and Trends monitoring design for water samples, in which sample collection occurs during the dry season, may not provide representative concentration data for these contaminants. A pilot study to assess occurrence during both dry and wet seasons is recommended, as with bisphenols.

Screening of OPEs in stormwater is now underway, intended to fill a major data gap identified in the RMP's synthesis and strategy concerning this class of compounds (Lin and Sutton, 2018).

Imidacloprid (Moderate Concern)

Neonicotinoids are a group insecticides developed as alternatives to organophosphates and carbamates, and are currently the most widely used class of insecticides worldwide (Bass et al., 2015; California Department of Pesticide Regulation (CDPR), 2016). Of the many neonicotinoids produced today, imidacloprid was the first of the class to be registered in 1991, and is now one of the most widely used insecticides in the world (Friends of the Earth (FOE) and Responsible Purchasing Network (RPN), 2017).

Neonicotinoids are widely used in urban and agricultural applications because of their efficacy; with a similar structure to nicotine, they are highly toxic to invertebrates through a systemic mechanism that affects the central nervous system. The chemicals irreversibly bind to nicotinic acetylcholine receptors in insects, which suggests that besides causing mortality, low-level continual exposure may also result in sublethal, cumulative effects (Hook et al., 2018; Maloney et al., 2017).

Imidacloprid has a wide range of urban and residential uses like indoor and outdoor pest control (e.g., ants and termites), residential landscape maintenance, as well as pet treatments for fleas and ticks (Sadaria et al., 2016; Ensminger et al., 2013). Manufacturers also incorporate imidacloprid into construction materials, such as polystyrene insulation, vinyl siding, adhesives, sealants, and pressure-treated wood (Sadaria et al., 2016; FOE and RPN, 2017).

Two significant pathways of neonicotinoid contamination to the Bay are treated wastewater and urban runoff. Because of its high solubility and low affinity for soils, imidacloprid is highly mobile. Urban and agricultural runoff is expected to transport imidacloprid to the Bay from surrounding urban and agricultural land. Independent surveys in urban areas in northern and southern California confirm that imidacloprid is a commonly detected insecticide in urban runoff (Ensminger et al., 2013; Murray, 2015; CDPR, 2016).

Effluent discharged by municipal wastewater treatment plants can transport residential uses of imidacloprid, particularly topical applications on pets that can be washed off and discharged to

municipal wastewater facilities (Sadaria et al., 2017; Teerlink et al., 2017). Previous RMP monitoring of Bay Area wastewater influent and effluent in 2015 detected imidacloprid in 100% of samples at levels up to 310 ng/L. In contrast, air is not expected to be a pathway for this contaminant because it is non-volatile and has a low soil adsorption coefficient, indicating a low potential to be dispersed via air-borne soil particles (Fossen, 2006).

In the summer of 2017, open Bay water samples were collected during the RMP Status and Trends Water Cruise to assess the presence and levels of neonicotinoids. The only neonicotinoid detected was imidacloprid; it was found at a single site in Lower South Bay at a level of 4.2 ng/L. This value is within the range of concentrations found in a separate study in water samples collected from the South Bay margins in 2017 (Heberger et al., 2020). In this study, imidacloprid was detected at 3 of 12 of the margin sites at levels between 3.9 and 11 ng/L.

No comparable studies of neonicotinoids in estuarine environments in the U.S. have been reported. However, a recent study from Hook et al. (2018) investigated the impact of several pesticides, including imidacloprid, on shellfish aquaculture along the east coast of Australia. Imidacloprid was detected at five of seven sites at concentrations ranging from 2.6 - 415 ng/L (site mean concentrations: 13.6, 14.6, 345, 2.8, 2.9 ng/L). San Francisco Bay imidacloprid concentrations were within the lower end of the range of the Australian study. Unlike the Australian study, the San Francisco Bay samples were not taken during a wet weather period when concentrations are likely to be elevated from increased urban stormwater inputs.

Aggregated open Bay and margins monitoring data indicated that imidacloprid concentrations in Lower South Bay were comparable to or greater than protective thresholds. These include the European Union predicted no effect concentration (PNEC) of 4.8 ng/L derived from chronic toxicity (EC_{10}) for *Cloeon dipterum* in freshwater (European Commission, 2015); the USEPA freshwater aquatic life benchmark (10 ng/L) for chronic invertebrate exposure in freshwater (US EPA, 2018); and an annual average freshwater environmental quality standard (AA-EQS) of 8.3 ng/L, which is based on chronic toxicity data and aims to protect freshwater ecosystems against adverse effects resulting from long-term exposure (Smit et al., 2015). All of these toxicity thresholds are based on freshwater species data; comparable thresholds designed to protect marine and estuarine species are not available.

While there are some studies that explore acute imidacloprid toxicity for marine species, there is a significant lack of chronic toxicity studies (Anatra-Cordone and Durkin, 2005; Cavallaro et al., 2017; Gibbons et al., 2015; Osterberg et al., 2012; Song et al., 1997). This is a concern because imidacloprid lethality and toxicity has been observed to be significantly greater when pollinators are exposed for long periods of time (Pisa et al., 2015).

The comparison between Bay monitoring results and toxicity benchmarks, in combination with widespread and increasing use of this pesticide in urban settings resulted in the classification of imidacloprid as a Moderate Concern for the Bay.

RMP Monitoring Strategy: At this time, there is no urgent need for ambient Bay monitoring of imidacloprid due to ongoing monitoring of select California streams by the Department of Pesticide Regulation.

However, two ongoing studies of stormwater will provide additional monitoring data on imidacloprid concentrations entering the Bay. MS4 stormwater permittees are now measuring imidacloprid as part of their permit compliance monitoring. Imidacloprid is also an analyte in a current multi-year RMP screening study monitoring CECs in Bay Area stormwater. Wet weather monitoring of current-use urban pesticides in ambient Bay water, suggested as a future priority, would provide useful information for imidacloprid as well.

Per- and Polyfluoroalkyl Substances (Moderate Concern)

PFAS are a broad class of fluorine-rich specialty chemicals. Perfluorooctane sulfonate (PFOS), perfluorooctanoic acid (PFOA), and many other PFAS possess thermal stability, non-reactivity, and surfactant properties, making them useful for many different types of applications. More than 4,700 PFAS are used in consumer, commercial, and industrial applications, including food packaging materials, waterproof textiles, stain-resistant carpets and furniture, fire-suppression foams, processing aids for the production of fluoropolymers like Teflon, mist suppressants in metal-plating, and hydraulic aviation fluids.

In the US, production of PFOS was phased out by 2002, and production of PFOA was phased out by 2015. This federal action was part of a broader international collaboration to reduce human and environmental risks associated with exposure to these compounds. PFOS was restricted under the global Stockholm Convention in 2009 as an Annex B chemical, allowing some specific exceptions to a total ban. PFOA, a compound with eight carbons (C8), as well as other PFAS compounds with C9 through C14 are on the European Candidate List of High Concern compounds due to their persistence, bioaccumulation risk, and toxicity. The goal of the Candidate List is to restrict the use of these chemicals and encourage industry to move to safer alternatives. Although restricted in North America and Europe, PFOS and PFOA production continues in some countries, such as China and India. In addition, global production of related replacements, including the short-chain perfluoroalkyl substances and polyfluoroalkyl substances, means continuing use of and exposure to compounds that may potentially pose similar risks.

Due to their diverse uses, PFAS have been entrained in Bay Area stormwater and wastewater discharged to the Bay. Analysis from multiple years of study found wastewater concentration trends consistent with manufacturing and use trends, with average levels of PFOS and PFOA decreasing (not statistically significant) and average levels of short-chain perfluoroalkyl substances increasing (statistically significant). Stormwater has been monitored less frequently, with the most recent study, completed in 2010-2011, finding detectable levels of PFAS

especially in more urbanized areas. To fill this data gap, PFAS have been included as one of five classes of analytes in the RMP's ongoing screening study of CECs in urban stormwater.

Within the Bay, the RMP monitors PFAS in bird eggs and sport fish as part of Status and Trends monitoring. Here, we describe recently reported data from a 2018 collection of cormorant eggs, a valuable indicator of regional patterns in contamination of the open Bay food web (SFEI, 2019). PFOS concentrations in cormorant eggs have been higher in South Bay than in Central Bay (Richmond Bridge) or Suisun Bay (Wheeler Island). South Bay concentrations have varied considerably, falling from over 1200 ng/g ww (wet weight) in 2006 and 2009 to approximately 400 ng/g ww in 2012, rising to around 600 ng/g ww in 2016, and then falling again to 250 ng/g ww in 2018. Meanwhile, the concentration in Central Bay in 2018 (27 ng/g ww) was substantially lower than in previous years. A Suisun Bay colony could not be sampled in 2018. Concentrations of PFOA and other long-chain carboxylates are present at lower concentrations than PFOS (Figure 1). While PFOA concentrations may be declining, other long-chain carboxylates do not demonstrate this trend.

PFAS concentrations in cormorant eggs in South Bay may be of concern. A field study indicated a 50% reduction in hatching success of tree swallows in Minnesota and Wisconsin at a PFOS concentration of 500 ng/g ww in eggs (Custer et al., 2013), a level similar to that observed in South Bay cormorant eggs. However, a followup study of a highly exposed tree swallow population in Michigan did not support a connection between PFOS exposure and reproductive or physiological responses, indicating uncertainty regarding this threshold (Custer et al., 2019).

Additional studies of wild bird populations suggest emerging concerns relating to exposure to long-chain carboxylates including those monitored by the RMP (Figure 1). A study of Arctic kittiwakes linked plasma levels of PFDoDA (perfluorododecanoate, C12) to reduced hatching (Tartu et al., 2014). The concentrations in plasma observed in this study are comparable to the concentrations observed in Bay cormorant eggs; concentrations in these two different tissue types are not readily comparable. In addition, higher exposure to PFDoA and other long-chain carboxylates has been linked to protein damage in populations of kittiwakes and great tits (Costantini et al., 2019; Lopez-Antia et al., 2019). These findings are of particular interest because the levels of PFDoDA in Bay bird eggs have remained relatively consistent for nearly a decade, indicating sustained exposure despite management actions (Figure 1).

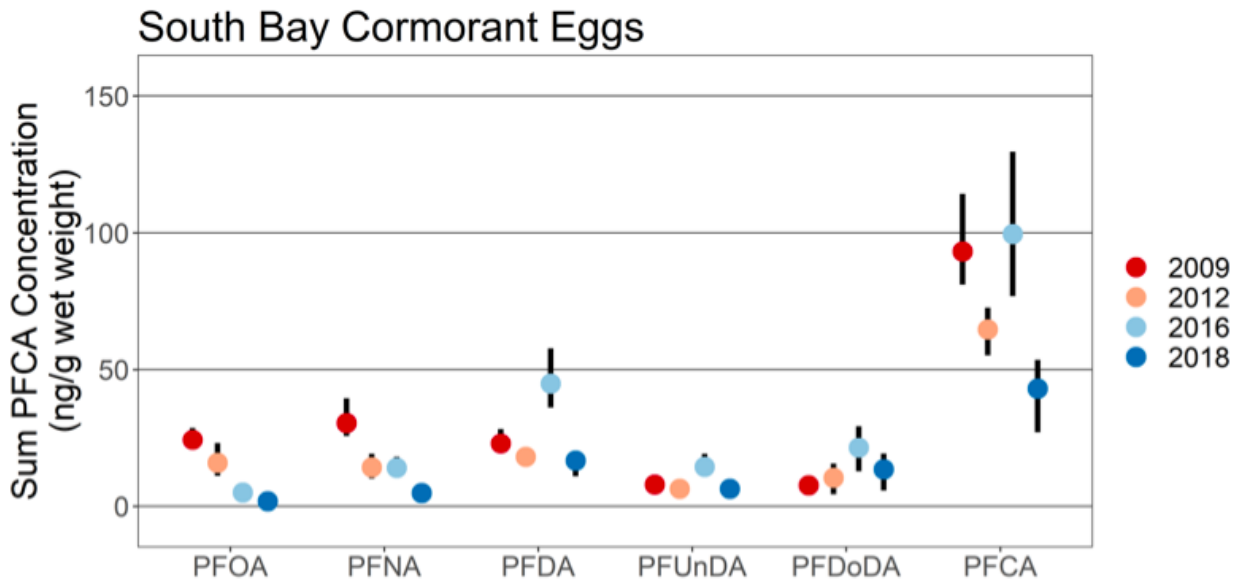

Figure 1. PFOA and other long-chain perfluorinated carboxylates (or PFCA) in cormorant eggs from the South Bay. Circles represent average concentrations, while bars range from minimum to maximum detection; ng/g wet weight is equivalent to parts per billion (ppb).

To date, much of the regulatory focus has been on the long-chain perfluoroalkyl chemicals (generally containing at least seven to eight carbons) such as PFOS and PFOA, in part based on their extensive toxicity profiles, their multi-year half-lives in human blood, past production volumes, and their bioaccumulation in aquatic food webs. In addition, these compounds show extremely high persistence in the environment, as noted previously. Within the US, the USEPA has developed a chemical action plan for the long-chain perfluoroalkyl substances. California's State Water Board recently announced new drinking water response levels (RLs) of 10 ng/L for PFOA and 40 ng/L for PFOS. Under a new California law (AB 756), if a drinking water system receives a State Water Board order for testing and finds that PFOA or PFOS concentrations exceed their RL, the system must take the water source out of service, provide treatment, or notify their customers. Last year, the State Water Board set drinking water notification levels (NLs), health-based concentrations that indicate the need for notification and further monitoring and assessment, of 5.1 ng/L for PFOA and 6.5 ng/L for PFOS. California's Office of Environmental Health Hazard Assessment (OEHHA) is in the process of developing public health goals (PHGs) for PFOA and PFOS, the next step in the State's process of establishing regulatory standards, known as maximum contaminant levels (MCLs), in drinking water. As noted previously, the State Water Board is also now conducting a statewide assessment to determine the scope of contamination by PFAS in water systems and groundwater; sampling will include discharges from wastewater treatment facilities.

This year, San Francisco Bay Regional Water Quality Control Board staff also developed interim final Environmental Screening Levels (ESLs) for PFOS and PFOA in groundwater (direct exposure human health risk levels), as well as in aquatic habitats for ecotoxicity (PFOS: 75

ng/L, PFOA: 4,400 ng/L) and human health via seafood ingestion (PFOS: 4.7 pg/L, PFOA: 22 pg/L). As with all the ESLs, the PFOS and PFOA ESLs are guidance intended to assist Regional Water Board staff currently overseeing the investigation and cleanup of PFAS spills and releases, so their use is not mandatory. However, they represent additional useful thresholds for comparison with environmental concentrations. Although 2009 water concentrations exceed human health via seafood ingestion ESLs, these thresholds are based on lifetime one in one million cancer risk to the 95th percentile upper bound estimate of fish ingestion rate, and are thus highly conservative.

To evaluate the potential impacts of existing management actions on levels of PFOS and PFOA in the Bay, the RMP participated in a pro bono project to develop a multi-box mass balance model to predict the long-term distribution and concentrations of these contaminants in water and sediment (Sánchez-Soberón et al., 2020). The model was based on similar, one-box models developed previously by the RMP for other contaminants (Davis, 2004; Greenfield and Davis, 2005; Oram et al., 2008; Yee et al., 2011). It included rate constants as well as time-varying water inputs, meant to capture the expected continuing declines in discharge due to the US phase-outs. A multi-box model was seen as particularly advantageous for modeling of PFOS and PFOA because it can calculate levels of these compounds in water and sediment for each subregion, allowing for a more spatially differentiated assessment of risks. PFOS and PFOA are generally observed at higher levels in South and Lower South Bays, and wildlife exposures in these subembayments have been the focus of the RMP risk evaluation.

The base case scenario for the model describes conditions in 2009, as this is the year for which the majority of PFAS monitoring data are available. In particular, Bay water has not been monitored for PFAS since 2009. The model predicts significant declines to nearly stable PFOA concentrations in water and sediment would be reached after 50 years, while PFOS needed close to 500 years to stabilize in sediment. Afterwards, concentrations stabilize between 4 and 23 pg/g in sediment, and between 0.02 and 44 pg/L in water, depending on compound and region. South Bay had the greatest final concentrations of pollutants, regardless of compartment.

To better understand the potential risks to humans from the consumption of fish, the model was linked to a model that calculated PFAS concentrations in fish tissue (Larson et al., 2018). Fish consumption was considered safe for most scenarios based on consumption thresholds for low to moderate fish diets available in other states. The European Food Safety Authority's draft scientific opinion on the risks to human health related to the presence of perfluoroalkyl substances in food (Schrenk et al., 2020) established a group tolerable weekly intake (TWI) for PFOA, PFNA, PFHxS and PFOS of 4.4 ng/kg body weight. For a 70 kg adult, this TWI would translate to 308 ng. Compared to the model's stabilized predictions of between 7 and 104 pg PFOS and PFOA per g ww fish, and assuming one serving of fish is 75 g, one meal per week may exceed this threshold, depending on region.

This modeling exercise identified the need for additional monitoring data to reduce uncertainty. Data for Bay water were limited to a single year, 2009, and as such, model predictions beyond the base case scenario in 2009 could not be compared to measured concentrations in this matrix. In addition, data were limited for PFAS in sediment and in precipitation, and could be improved in wastewater and stormwater as well.

RMP Monitoring Strategy: Major improvements in standardized analytical methods for PFAS have resulted in the expanded ability to characterize not just PFOS and PFOA, but also a broad range of alternatives in increasing use, including short-chain perfluorinated compounds and polyfluorinated compounds. A high priority should be placed on monitoring of Bay water with these new methods. Bay water has not been monitored since 2009; new data would allow an initial analysis of trends in PFOS and PFOA concentrations, and provide information on additional PFAS of potential concern. Characterization of PFAS in Bay water would provide information that could inform the State Water Board investigation of PFAS statewide.

Monitoring Bay margin sediment may also improve our understanding of the occurrence and risks associated with PFAS. In 2017, a handful of South and Lower South Bay margin sediment samples were archived for PFAS analyses. This region of the Bay is of particular concern given higher exposures in resident wildlife. Additional margin sediment sample collection is planned for the North Bay; this region is of interest given due to the presence of refineries and other industries that store and use PFAS-containing aqueous film forming foams (AFFF) for fire suppression. In October 2019, a fuel fire at a Crockett fuel tank facility led to deployment of AFFF (T. Mumley, personal communication). Emergency personnel took action to prevent discharge of AFFF to the storm drain systems and collect waste liquid for disposal. Facility management submitted a sampling plan to the Regional Water Board to investigate potential impacts; data are not yet available. An incident like this emphasizes the PFAS sources present in the North Bay region and suggests monitoring may be needed.

PFAS monitoring in sport fish and bird eggs is conducted as part of RMP Status and Trends monitoring. Data to date has indicated declines in PFOS and, in some cases, PFOA, while concentrations of other analytes have not shown clear temporal trends. Status and Trends monitoring for both tissue types is recommended to continue, particularly now that analytical methods have improved and can provide information on a broader array of PFAS.

In addition to targeted analyses on routinely collected species, the RMP may consider a more exploratory, non-targeted and suspect screening approach to determine whether unknown or unexpected PFAS may be present in wildlife. Archived tissue samples for Bay harbor seals are a suitable matrix for a non-targeted investigation of both known and unknown PFAS. A recent targeted and nontargeted analysis of marine mammal tissues collected from across the northern hemisphere found that while PFOS typically dominated, an additional 33 PFAS not yet included in targeted methods were also present (Spaan et al., 2019). In the Bay, results from harbor seal samples collected over different time periods could even suggest temporal trends in use and environmental presence.

For Moderate Concern contaminants like PFOS and PFOA, the RMP typically investigates presence in pathways. PFAS is being characterized in Bay Area stormwater as part of the RMP multi-year CEC screening study. While the RMP is not currently conducting a similar study on wastewater, there is interest among Bay Area wastewater agencies in conducting a collaborative PFAS monitoring study as a proactive response to State Water Board efforts to characterize the presence of this class of contaminants in various sources and pathways statewide. Thanks to prior wastewater monitoring supported by the RMP, any newly developed data could be used to further assess temporal trends. A sewershed study to help identify PFAS sources could be part of this effort, or could be conducted as a followup study by the RMP.

A study of air deposition is also currently recommended, given concerns relating to the volatility and long-range transport of polyfluorinated precursors that degrade to perfluorinated alkyl substances. However, the California Air Resources Board has just proposed adding a large number of PFAS to the list of pollutants required to be monitored under the Air Toxics Hot Spots program, an effort that could help to fill the present data gaps.

Galaxolide (Low Concern)

Galaxolide (or HHCB) is a polycyclic musk used as a fragrance ingredient in various personal care and cleaning products including cosmetics, detergents, fabric softeners, perfumes and soaps. Galaxolide is a high production volume chemical in the US, produced or imported in quantities between one and ten million pounds in 2015 (US EPA, 2016). The widespread use of galaxolide has highlighted it as a key fragrance ingredient of concern to monitor (ECHA, 2008).

Usage of galaxolide in all kinds of consumer products has led to high concentrations in wastewater treatment plants, especially in the US where reported levels have been the highest at 3.7 µg/L (Clara et al., 2011; ECHA, 2008; Reiner et al., 2007). The high lipophilicity (5.3-6.3) means adsorption to sludge is the major removal pathway in wastewater treatment, though a large range of removal efficiencies, from 50 to 95%, have been reported (Clara et al 2011; ECHA 2008). As a result of incomplete removal, galaxolide is likely contained in wastewater discharge. Surface waters and sediment are likely to be impacted from wastewater discharges, as well as nearby applications of biosolids.

Recently, a study conducted in the South Bay margins found detectable levels of galaxolide in water. This project was motivated in part by a desire to learn whether galaxolide was present at levels of concern in Bay waters. The margins of Lower South Bay are mudflats and shallow regions that receive direct discharges of stormwater and wastewater; as a result, they may have higher levels of urban contaminants than the open Bay (Yee et al., 2019, 2018). Galaxolide, the most frequently detected fragrance ingredient among the panel of analytes, was detected in 25% of South Bay margins water samples (3 of 12 sites). The maximum observed concentration was 30 ng/L.

The same study also found detectable levels of HHCB in margin sediment samples. Galaxolide was detected at a maximum concentration of 12 µg/kg dw [est.] in 2 of 12 samples. Previous monitoring by the RMP found detectable levels of galaxolide in bivalves (855 µg/kg dw) and bird eggs (1 µg/kg wet weight) collected in 2002 through 2004 (Klosterhaus et al., 2013).

In contrast, a recent study of effluent-dominated rivers in southern California reported average galaxolide levels of 2,260 and 2,410 ng/L for the Los Angeles and San Gabriel Rivers, respectively (Sengupta et al., 2014). Water samples from the Russian River contained up to 370 ng/L HHCB (Maruya et al., 2018). However, laboratory blank contamination of 120 ng/L galaxolide was also reported in this study. Other studies of galaxolide in US water bodies include sites along the Potomac River basin (detection frequency 57%, maximum concentration 27 ng/L; Kolpin et al., 2013) and source waters for drinking water treatment facilities (detection frequency 36%, maximum concentration 110 ng/L; Glassmeyer et al., 2017). Relative to these studies of US freshwater systems, detections of galaxolide in South Bay are not high.

Galaxolide has significant ecotoxicity concerns. A study that evaluated the impacts of subchronic exposure on larval development of the marine copepod *Nitocra spinipes* established a no observed effect concentration (NOEC) of 7,000 nanograms per liter (ng/L) (Breitholtz et al., 2003). Based on this study, a monitoring trigger level (MTL) of 70 ng/L was established for California estuarine waters (Anderson et al., 2012). MTLs are protective benchmarks based on potential ecological and human health risks; exceedances can indicate a need for additional monitoring but do not necessarily indicate a toxicity concern. The MTL is more protective than the ECHA marine aquatic PNEC for galaxolide, 440 ng/L. A more recent study derived a series of ecotoxicity thresholds for galaxolide using data from multiple freshwater species; the most protective was a criterion continuous concentration of 2,220 ng/L, developed using USEPA guidelines (Fan et al., 2019). Benthic crustaceans appeared to be more sensitive than planktonic crustaceans, suggesting more research may be needed with respect to sediment toxicity. The ECHA marine sediment PNEC for galaxolide is 394 µg/kg dw.

In South Bay margins water samples, the median concentration of galaxolide was less than the analytical method reporting limit (RL) of 20 ng/L. The maximum observed concentration was 30 ng/L, below the MTL (70 ng/L), as well as the ECHA PNEC for marine waters (440 ng/L). Likewise, the maximum level of galaxolide in sediment (12 µg/kg dw) was lower than the ECHA PNEC for marine sediment (394 µg/kg dw). Levels of the polycyclic musk galaxolide observed in the margins samples were consistent with placement in the Low Concern tier within the RMP's tiered risk-based framework.

Fragrance ingredients are considered part of a larger class of personal care and cleaning product ingredients. Thousands of chemicals are used in fragrance mixtures and personal care and cleaning products, and the RMP has only monitored the Bay for a small subset of these compounds. Those that have been monitored have been classified as Low Concern for the Bay, with minimal impacts to wildlife anticipated (Sutton et al., 2017).

RMP Monitoring Strategy: Periodic monitoring is recommended in order to determine if levels of galaxolide and other fragrance and personal care product ingredients are increasing or decreasing over time. A rapidly growing Bay Area population suggests that contamination derived from common consumer products could increase in the future. If future monitoring indicates that specific contaminants may be having an adverse impact on Bay aquatic life, followup monitoring is recommended to better understand sources and pathways.

QACs (Possible Concern)

Quaternary ammonium compounds (QACs) are a major class of cationic (positively charged) surfactants with a positive nitrogen head group and four hydrophobic alkyl side chains of varying lengths. The different classes of QACs have important antimicrobial, anti-static, and surfactant properties. QACs are used in a wide swath of consumer, industrial, and medical products including biocides, cosmetics, detergents, disinfectants, fabric softeners, lotions, mouth washes, nasal sprays, and shampoos (Melin et al., 2014; Zhang et al., 2015). Ionic liquids, composed of the same core structure as QACs, are an emerging product for usage as “green solvents.” All QACs are considered high production volume chemicals in the US (USEPA, 2016).

The widespread use of QACs leads to considerable amounts ending up in wastewater. Research on the fate of QACs indicates effluents and sludge from WWTPs as a major pathway for environmental releases (Boethling, 1984; Clara et al., 2007; Li et al., 2014; Pati and Arnold, 2020). Additionally, local point sources to WWTPs like hospitals (Kümmerer, 2001), with widespread use of QACs, have been identified. WWTPs with aerobic treatment systems in Austria exhibited removal rates greater than 90% for QACs, which occurred through adsorption to solids and possibly biodegradation (Clara et al., 2007). The unique cationic and hydrophobic properties of QACs suggest that they adsorb well onto particles with high organic matter and negatively charged surfaces such as sewage sludge, sediment, and soil (Zhang et al., 2015). WWTP effluent discharges containing these contaminants have resulted in significant QAC accumulation within aquatic sediment.

A recent pro bono investigation conducted by Dr. Bill Arnold at the University of Minnesota found detectable levels of several QACs in San Francisco Bay sediment. Eleven samples from sites spanning the entire Bay from Lower South Bay to its northern reaches at the confluence of the Sacramento and San Joaquin Rivers were tested for 14 QACs. Analytes included alkyltrimethyl ammonium compounds (ATMACs; alkyl lengths C10, C12, C14, C18); benzylalkyldimethyl ammonium compounds (BACs; alkyl lengths C10, C12, C14, C18); dialkyldimethyl ammonium compounds (DADMACs; alkyl lengths C12, C16, C18); as well as benzethonium, cetylpyridinium (PYR-C16), and domiphen.

Of the 14 QACs analyzed, seven (C12, C14, C18 BACs; C12, C16, C18 DADMACs; benzethonium) were detected at two or more sites, with the greatest sum detected in Grizzly

Bay (North Bay) with a concentration of 1,200 ng/g dry weight (dw). Additionally, Grizzly Bay was the only site where all seven QACs were detected. Of note, no QACs were detected at quantifiable levels in upstream sites in the Sacramento and San Joaquin Rivers. After Grizzly Bay, the next most contaminated sites were in Lower South Bay, an embayment strongly influenced by municipal WWTP effluent discharges. BAC-C18 was the most widely detected (nine sites), though DADMAC-C18 had the highest concentrations at the eight sites where it was detected despite low rates of recovery. Concentrations of DADMAC-C18 ranged from 42 ng/g dw near Treasure Island to 830 ng/g dw in Grizzly Bay, with a median of 290 ng/g dw. The higher concentrations in Grizzly Bay may suggest a localized source.

A Bay sediment core collected in 2011 from a site in the Central Bay near Richmond (Kerrigan et al., 2015) was analyzed for the same suite of 14 QACs. The core spans roughly 60 years of sediment deposition (approximately 1951-2009). The same seven QACs were detected in two or more portions of the core, with the largest concentration found between 20–22 cm at 3,000 ng/g dw. DADMAC-C18 was found in all layers at the highest concentrations, and averaged 50 times greater than the next most pervasive compound, BAC-C18. Both BAC-C12 and C18 were found in all layers in relatively low concentrations (13 and 33 ng/g dw on average, respectively). The concentrations of DADMAC-C18 ranged from 911 ng/g dw at the surface (0 – 4 cm) to 2,578 ng/g dw at a depth of 20-22 cm (Figure 2). Compared to surface sediment samples from ambient Bay waters, the surface (0 – 4 cm) of the sediment core from a near-shore site had a larger concentration likely due to better sediment build up in its location. The concentration profile with depth suggests a declining temporal trend in sediment, possibly due to declining use of this QAC. The concentration profiles of some other QACs, such as BAC-C18, are not as strongly suggestive of declines (Figure 3).

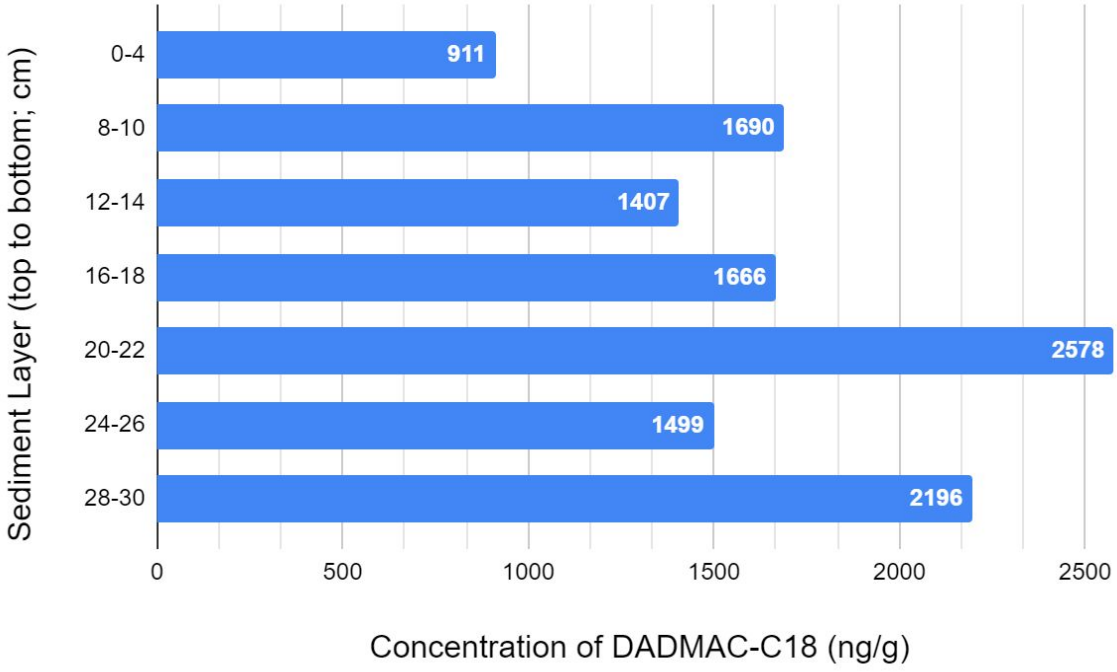

Figure 2. Concentrations of DADMAC-C18 in Central Bay sediment core layers.

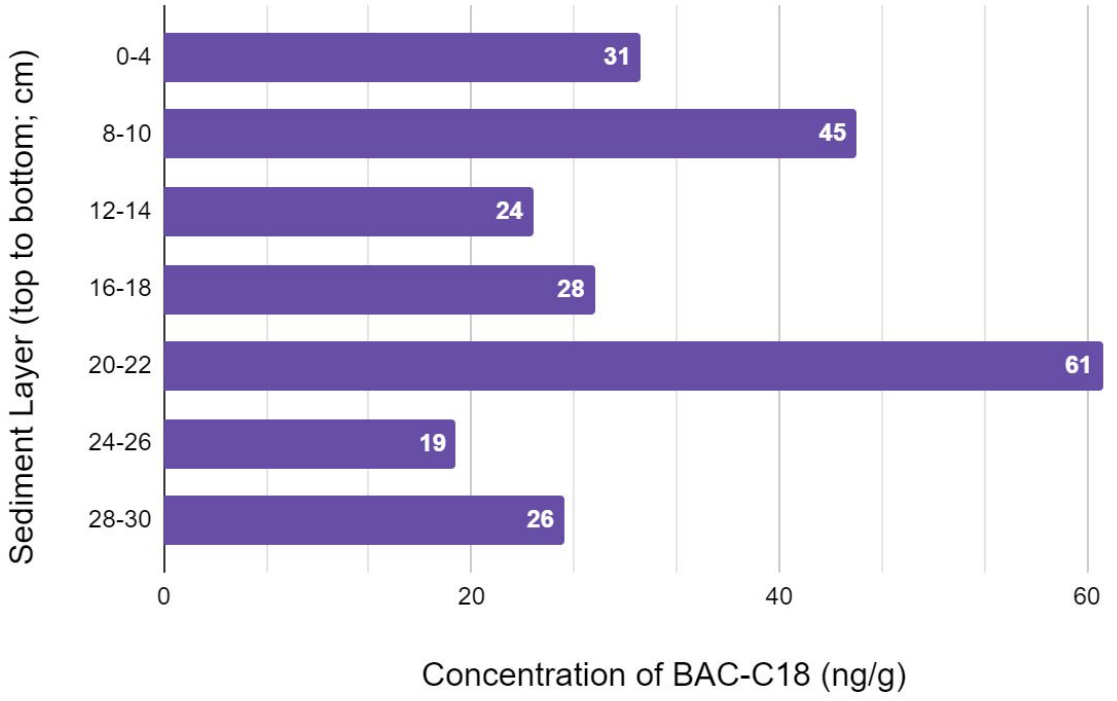

Figure 3. Concentrations of BAC-C18 in Central Bay sediment core layers.

Within the state of New York, studies on estuarine sediment in urban watersheds have shown higher concentrations of QACs (980 to 114,000 ng/g dw) relative to San Francisco Bay, and generally indicated urban WWTP discharges are a major pathway for these contaminants (Li and Brownawell, 2010, 2009). As in the Bay sediment, DADMACs contributed to the total in much higher concentrations (1,700 – 52,000 ng/g dw) than other QACs.

Another study identified three classes of surfactants in surface sediment and dated cores from Jamaica Bay near Long Island (Li et al., 2018). Within surface sediment, concentrations of surfactants were observed in all 21 samples from 18 to >200 µg/g, and were largely composed of cationic surfactants like QACs. In particular, DADMACs with high molecular weights (C14, C16, C18), also known as ditallowdimethylammonium chlorides (DTDMACs), composed 52% to 90% of all surfactants found for each sample. DTDMACs have been voluntarily phased out in Europe, while use may continue in the US. Two sediment cores collected in 1996 and 2008 exhibited subsurface sediment peaks for most compounds in the mid-1960s and 1980s, at maximum summed concentrations of 469 µg/g and 572 µg/g dw. DADMACs, however, showed an increasing temporal trend to the surface of the cores. Further, degradation studies conducted on the cores demonstrated the recalcitrant behavior of DADMACs; their concentrations remained the same over the time span examined (1996-2008).

Most recently, surface sediment and dated core samples from lakes in Minnesota revealed detectable levels of QACs (Pati and Arnold 2020). In Lake Winona, surface sediment samples contained a summed QAC concentration range of 2400 to 4900 ng/g dw, with DADMACs found in the largest concentrations. Sediment core samples from two of the lakes in Minnesota found the highest concentrations of QACs occurring in sediment from the 1950s to the 1970s, with accumulation rates in the largest lake at roughly 6.7 µg/cm^2 yr. These levels decreased sharply after 1980 and stabilized at 1.35 µg/cm^2 yr from the late 1990s onwards. However, for the third lake, QAC levels during the same period were found to be increasing over time, with a maximum QAC accumulation rate of 1.57 µg/cm^2 yr. Overall, the levels detected in samples in San Francisco Bay sediment are significantly lower than the other wastewater impacted aquatic environments studied to date.

Because QACs are present in various environmental matrices, it is critical to understand the risks they may pose to wildlife. QACs are constructed to be biocidal and have been shown to be toxic to a variety of aquatic organisms including algae, daphnids, fish, and microorganisms (Nałęcz-Jawecki et al., 2003; Sandbacka et al., 2000; Zhu et al., 2010). The European Chemicals Agency (ECHA) class-wide aquatic PNEC for marine organisms is 0.62 µg/L (ECHA, 2010b). These studies indicate a range of sensitivities for different species to QACs, illustrating the need for more thorough study to predict toxic effects and estimate protective thresholds.

Although there are a number of toxicological studies examining impacts to aquatic life, more research is needed to understand the potential toxicity of QACs to sediment dwelling organisms. As QACs are more likely to adsorb to particulate matter, as well as negatively charged surfaces common in Bay sediment, their bioavailability in sediment may be significantly limited. The

ECHA class-wide PNEC for marine sediment is 11 mg/kg dw (ECHA, 2010b), which is significantly higher than the highest concentrations measured in Bay sediment. However, this value is based on studies performed before the year 2000, and newer QACs may have differing toxicities. Under these circumstances, QACs should be considered a Possible Concern for the Bay.

RMP Monitoring Strategy: Periodic monitoring of QACs in Bay sediment should be pursued. A rapidly growing Bay Area population suggests that contamination derived from common consumer products could increase in the future. Because QACs have major uses as antimicrobial active ingredients and as fabric softeners, use in the Bay Area could increase. In addition, increased use of antimicrobial products in response to the COVID-19 outbreak could continue into the future. An investigation of QACs in Bay Area influent and effluent is currently underway to characterize occurrence and recent temporal trends related to COVID-19. Wastewater biosolids and stormwater will also be screened to note their significance as a pathway for QACs. A small number of samples of North Bay margin sediment from sites located near Grizzly Bay, where the highest detection was observed previously, will also be monitored.

Current and future monitoring data should be compared to toxicity thresholds for sediment, as they become available. If monitoring indicates that specific contaminants may be having an adverse impact on Bay wildlife, follow up monitoring is recommended to better understand the sources and pathways by which contaminants enter the Bay.

Indole (Possible Concern)

Both a natural and manufactured chemical, indole is used in a variety of products, most frequently as a fragrance ingredient. Indole can also be found in coal tar, cigarette smoke, and petroleum processing wastewater (Ma et al., 2018). It is widely used as a fecal biomarker, as it is produced by gut microbiota in both humans and animals. Over 85 known bacterial species can biosynthesize indole, meaning it is also created naturally in the environment (Ma et al., 2018).

As it is anthropogenically and naturally made, there are many pathways leading to its presence in the environment. Industrial wastewater flows are a large anthropogenic contributor of indole due to its use within various industrial processes and production as a byproduct (Ma et al. 2018). Agricultural wastewater flows may also contribute, mainly due to livestock waste.

A recent study conducted by the RMP of South Bay Margins found indole in 11 of 12, or 92% of sediment samples. The median concentration of indole in bed sediment was 230 µg/kg dw, and the maximum observed concentration was 405 µg/kg dw. However, it was not detected in any water samples (reporting limit 20 ng/L).

A study of fragrance ingredients and other compounds in bed sediment from six stream sites in the Potomac River basin found somewhat similar levels of contamination (Kolpin et al., 2013). Indole was detected in all samples in the Potomac Basin, with a maximum concentration of 197 µg/kg dw. In addition, research on several bioactive contaminants, including fragrance ingredients, in bed sediment from twenty locations (nine stream sites, nine lake sites, and two wetland sites) in remote Rocky Mountain National Park (RMNP) demonstrated much higher levels of contamination than the Bay (Battaglin et al., 2018). All sediment samples in RMNP contained indole with a maximum concentration of 1,150 µg/kg dw, which is significantly greater than the freshwater PNEC (56.6 µg/kg dw). Within surface water, indole was detected in 12% of samples in RMNP at an estimated maximum concentration of 18 ng/L, which is lower than the reporting level (40 ng/L). Overall, indole found in the RMNP was observed in greater concentrations in both bed sediment and surface waters when compared to the Bay RMP study.

German scientists flagged indole as a potential problem in a study of marine water and sediment off the northern coast of Germany (Reineke et al., 2006). The authors linked one case of observed toxicity to a combination of brominated phenols and indoles, and noted that these compounds, “which are assumed to be of biogenic origin, have rarely been discussed so far in the context of ecotoxicologic effects in marine ecosystems.” The ECHA marine PNECs for indole are 5.66 µg/kg dw for sediment and 200 ng/L for water.

Every detection in South Bay sediment was above the marine sediment PNEC of 5.66 µg/kg dw. The median concentration of indole in bed sediment was 230 µg/kg dw, 40 times greater than the PNEC. The maximum observed concentration of indole in bed sediment was 405 µg/kg dw, which exceeds the PNEC by a factor of 70.

However, closer examination of the marine sediment PNEC revealed that it was developed in the absence of any ecotoxicity data relevant to sediment-dwelling organisms and was instead provisionally calculated using sparse freshwater aquatic ecotoxicity thresholds and a standardized equilibrium partitioning method. The higher level of uncertainty associated with this PNEC suggests the need for further testing in benthic species. An examination of the scientific literature revealed only one additional aquatic ecotoxicity study (Eisentraeger et al., 2008) and no sediment ecotoxicity studies.

Given the uncertainties associated with the potential for indole to pose risks to Bay wildlife, the present evidence is consistent with a classification of Possible Concern for the Bay.

RMP Monitoring Strategy: Periodic monitoring in sediment is recommended in order to determine if levels of indole and other fragrance and personal care product ingredients are increasing or decreasing over time. A rapidly growing Bay Area population suggests that contamination derived from common consumer products could increase in the future. An improved understanding of potential indole toxicity is essential for evaluating risks to wildlife. If future information develops to indicate that indole may be having an adverse impact on Bay

wildlife, followup monitoring will be recommended to better understand the sources and pathways by which this contaminant enters the Bay.

4-Methylphenol (Possible Concern)

Like indole, 4-methylphenol (or para-cresol) has both natural and anthropogenic sources. In addition to being a naturally occurring compound produced by mammalian gut bacteria, 4-methylphenol is also used in preservatives (e.g., creosote), to formulate antioxidants, and many other materials such as lubricating oils, motor oils, rubbers, polymers, elastomers, and food products. Creosote-treated pilings and other wooden structures in the Bay are one of the anthropogenic sources of this compound; however, previous extensive mapping has identified relatively few pilings in the South Bay (Werme et al., 2010).

A recent study conducted by the RMP of South Bay margins detected 4-methylphenol in 7 of 12 monitoring locations, or 58% of South Bay margin sediment samples. 4-Methylphenol was not detected in water samples. The median concentration was 40 µg/kg dw, and the highest observed concentration was 150 µg/kg dw. However, all values were below the reporting limit of 250-500 µg/kg dw, and are therefore considered estimates only.

A study of fragrance ingredients and other compounds in bed sediment from six stream sites in the Potomac River basin found higher levels of contamination; 4-methylphenol was found in all sediment samples, with a maximum concentration of 396 µg/kg dw (Kolpin et al., 2013). South Bay concentrations of 4-methylphenol in 4 of 12 sediment samples (33% of samples) exceeded the marine PNEC of 85 µg/kg dw. However, as with indole, the ECHA marine sediment PNEC for methylphenol was provisionally calculated using freshwater aquatic ecotoxicity data rather than data relevant to sediment-dwelling organisms. Likewise, a Canadian screening assessment for methylphenols (including 2-, 3-, and 4-methylphenol) lacked sediment ecotoxicity data, and instead derived a PNEC relevant to sediment using water-only toxicity testing of a sediment-dwelling amphipod (*Gammarus pulex*). The resulting PNEC was 2.1 mg/L (Government of Canada, 2016). Concentrations in this range would undoubtedly have been detected in San Francisco Bay water samples in the recent screening study (reporting limit 40 ng/L); however, 4-methylphenol was not detected in any margin water samples. Clearly, testing in benthic species is needed to establish a more robust sediment toxicity threshold.

Due to uncertainties in the quantification of methylphenol (concentrations below reporting limits) as well as its toxicity, the present evidence supports a classification of Possible Concern for the Bay.

RMP Monitoring Strategy: As with galaxolide and indole, periodic monitoring is recommended in order to determine if levels of 4-methylphenol are increasing or decreasing over time. Improved analytical methods with lower reporting limits in sediment and an improved understanding of potential toxicity to benthic organisms is essential for evaluating risks to

wildlife. If future information develops to indicate that 4-methylphenol may be having an adverse impact on Bay wildlife, followup monitoring will be recommended to better understand the sources and pathways by which this contaminant enters the Bay.

Chlorinated and Brominated Azo Dyes (New Contaminants)

More than 10,000 dyes are used in textile manufacturing, and azo dyes account for approximately 70% of the global industrial demand (Rawat et al., 2016). Azo dyes are the most commonly used synthetic colorants for both natural and synthetic materials for reds, oranges, and yellows. These dyes are not only used in textiles, but also in lacquers and varnishes, printing inks, plastics, and to color cosmetics, waxes (e.g., candles), soaps, leather, and paper. Current EU regulations classify azo dyes based on benzidine, 3,3-dimethoxybenzidine, and 3,3-dimethylbenzidine, as 'substances which should be regarded as if they are carcinogenic to man.' However, brominated and chlorinated azo dyes are not included in this classification and continue to be produced and used.

In addition to their environmental release as part of industry waste, azo dyes may also be released to the environment via the use (e.g., laundering) and disposal of products containing them. Many can pass through municipal wastewater plants nearly unchanged due to their resistance to aerobic treatment, while under anaerobic conditions, they can be cleaved by microorganisms to form carcinogenic aromatic amines. A portion may also be captured in the sludge (de Aragão Umbuzeiro et al., 2005). Because these dyes are used in a wide variety of products, they may also end up in urban runoff due to improper disposal of trash. Potential pathways to San Francisco Bay include wastewater effluent, stormwater, and aerial deposition of urban dust.

Environmental monitoring of halogenated azo dyes remains relatively rare, despite their potential risk to aquatic food webs. The majority of study has been focused on production via industry waste and microbially-mediated environmental degradation. These compounds are structurally diverse, and therefore have diverse environmental fates and toxicities.

Existing studies indicate that halogenated azo dyes may be a concern in highly urbanized areas like San Francisco Bay. Brominated and chlorinated azo dyes have not been previously monitored in the Bay, so their levels are unknown. However, recent studies of indoor dust revealed that nitrogen-containing brominated azo dyes were among the most frequently detected and abundant contaminants, and more common and abundant than commonly detected brominated flame retardants (Dhungana et al., 2019; Peng et al., 2016). For a study of the Piracicaba River in Brazil, samples were collected upstream and downstream of two main discharges: the effluent of a wastewater treatment plant and the tributary Quilombo River, which receives untreated effluent from local industries. The authors found two halogenated azo dyes, Blue 373 and Disperse Violet 93, were the major contributors to sample mutagenicity from both discharges (Vacchi et al., 2017). Another study in Brazil indicated that azo dyes in sediment

downstream of a dye processing plant contributed to mutagenicity of the sediment (Palma de Oliveira et al., 2006).

Many of the newer azo dyes are halogenated (brominated or chlorinated) and mutagenic, genotoxic, or carcinogenic, with the potential to bioaccumulate (Danish Environmental Protection Agency, 1999). Studies of fish exposed to brominated azo dyes have shown bioactivation of these compounds to mutagenic aromatic amines (Han et al., 2020) that can lead to reproductive toxicity and multigenerational effects (Xie et al., 2019). Once released to the environment, microbial metabolism of brominated and chlorinated azo dyes can also produce toxic metabolites (Rawat et al., 2016). Chronic toxicity data show that invertebrates are the most sensitive species, and suggest higher risk for substances with lower molecular weights (less than 360 g/mol) (Government of Canada, 2013). Environment and Climate Change Canada has calculated an aquatic PNEC of 0.0025 mg/L for azo dyes less than 360 g/mol. Available soil and sediment toxicity data indicate that these compounds may be hazardous to sensitive sediment-dwelling organisms at low concentrations in sediment; however, the limited available data are not considered suitable for use as a critical toxicity value and a PNEC has not been calculated for the sediment compartment (Government of Canada, 2013). Thresholds for individual halogenated dyes have not been assessed.

RMP Monitoring Strategy: Halogenated azo dyes have not been previously monitored in San Francisco Bay; monitoring is needed to assess whether and to what extent these contaminants are present. A proposed project would screen archived sediment samples from Lower South Bay for these contaminants. As an add-on option, this project could also include analysis of archived North Bay sediment, to be collected in 2020, in order to begin to assess spatial distribution of azo dyes within the Bay. The goal of this study is to assess Bay sediment samples for halogenated azo dyes using high-resolution mass spectrometry. Concentrations in Bay sediment would be compared to rapidly evolving toxicity data to assign detected chemicals to a tier in the RMP tiered risk-based framework for CECs and determine whether followup study is needed. Should levels of concern be observed, a study of sediment cores could be considered as a followup assessment.

Toxicology Strategy for CECs in the Bay

Prioritization in the tiered risk-based framework is based primarily on occurrence data and the probability of effects on Bay wildlife. However, prioritization of many CECs is hindered by a lack of toxicological data and, therefore, unknown or low-confidence toxicity thresholds.

One way to inform prioritization of these data-poor chemicals, which are classified as Possible Concern in the tiered risk-based framework, is the use of predictive toxicology. The field of predictive toxicology includes both *in vitro* (i.e., test tube) and *in silico* (i.e., computer) models to predict how a chemical will interact with biological systems and what types of adverse effects may result. *In vitro* predictive toxicological methods can also be used to identify which types of organisms may be most sensitive, and therefore, which environmental matrices need further chemical characterization. These methods are generally faster and cheaper than traditional toxicological assays, but are also newer and therefore based on an evolving knowledge of biology and less commonly used in a regulatory context.

In response to the lack of hazard data for many chemicals, the US federal government (including the Environmental Protection Agency (US EPA), National Toxicology Program (NTP) headquartered at the National Institute of Environmental Health Sciences (NIEHS), National Center for Advancing Translational Sciences (NCATS), and Food and Drug Administration (FDA)) collaborative program Toxicology in the 21st Century (Tox21) aims to “transform toxicity testing from a system based on whole-animal testing to one founded primarily on *in vitro* methods that evaluate changes in biologic processes using cells, cell lines, or cellular components” (Krewski et al., 2010). Under this program, thousands of chemicals have been screened using approximately 70 high-throughput *in vitro* assays, covering over 125 important biological processes (US EPA, 2019). Numerous predictive toxicology tools for using this data have been developed, including EPA’s Computational Toxicology Dashboard, CompTox (https://comptox.epa.gov/dashboard).

One way to translate *in vitro* data into predictions of possible organism and population-level effects relevant to risk assessment is through the use of adverse outcome pathways (AOPs), which are conceptual frameworks that portray existing knowledge linking a direct molecular initiating event to an adverse outcome at higher levels of biological organization (Ankley et al., 2010). For example, activation of an estrogen receptor may lead to higher-level effects, such as gonadal tissue differentiation or changes in growth, and changes in the sex ratios of the population, as was demonstrated in a previous RMP study using the inland silverside (*Menidia beryllina*), a Bay species (Denslow et al., 2018, 2017). Many AOPs have been developed and compiled in publicly accessible databases, such as AOPWiki and AOP-KB.

Even with a complete AOP to provide possible organism and population-level effects from a chemical’s cellular activity, understanding the risk to Bay wildlife requires comparison of environmental concentrations with concentrations that produce effects. This comparison is often

calculated as an exposure:activity ratio (EAR). One way of prioritizing chemicals for further study is comparing EARs, calculated as the exposure concentration divided by the concentration for which a high-throughput assay showed a cellular effect (Becker et al., 2015; Blackwell et al., 2017). Several tools for calculating and visualizing EARs using environmental monitoring data are publicly available, such as USGS's toxEval R package. Risk assessment and prioritization of CECs using EARs has been successfully performed in several places, including the Great Lakes region, Colorado, and Oregon (Blackwell et al., 2017; Corsi et al., 2019; Heiger-Bernays et al., 2018).

A complementary approach, which can also be used for chemicals that have not been screened for toxicity using *in vitro* methods, are *in silico* tools that allow prediction of contaminant behavior based on chemical structure. Contaminants with structural similarities (e.g., members of a class) often have similar properties with respect to persistence, bioaccumulation, and/or toxicity. For compounds within a class containing at least one compound with well understood toxicity, read-across, an approach where available data of a data-rich substance are used to predict behavior or toxicity of a similar data-poor substance, can be used. Read-across methods are already commonly used in risk assessment. The US EPA has generated an automated read-across tool called Generalized Read-Across (GenRA), which is included in CompTox. More sophisticated structure-based predictions are also possible using quantitative structure-activity relationships (QSARs). For example, US EPA's Ecological Structure Activity Relationships (ECOSAR) Class Program is a computerized predictive system that estimates aquatic toxicity. The program estimates a chemical's acute (short-term) toxicity and chronic (long-term or delayed) toxicity to fish, aquatic invertebrates (daphnids), and aquatic plants (US EPA, 2015). As with any model, use of *in silico* tools requires caution and knowledge of what they are and are not capable of due to their development and training datasets.

There are many available predictive toxicology tools for assessing the possible risks of compounds known to be present in the Bay that do not have adequate toxicological data, but there may also be other CECs we have not looked for yet that have the potential to harm the health of wildlife and humans. For example, targeted analytical methods rarely capture the occurrence of transformation products, which in some cases can be more toxic than their parent CECs. While non-targeted chemical analysis can help to identify unknowns, these methods are expensive and often result in detections of many more chemicals for which there are no toxicity thresholds or data. Furthermore, Bay organisms are actually exposed to complex chemical mixtures, which may not be accurately represented by single contaminant exposure experiments. Single substances present below their individual effect thresholds may still contribute to combined mixture effects (Kienzler et al., 2019; Silva et al., 2002). In general, contaminants with similar modes of action have predictable additive toxicity, but concern remains for the possibility of synergism or antagonism, and whether fractional approaches to toxicity testing can determine what to expect from chemical mixtures in real world circumstances (Rodea-Palomares et al., 2015).

Bioassay monitoring of environmental samples can detect possible biological effects that may not be predictable solely from chemical analyses of the same samples (Blackwell et al., 2019). With this in mind, the Southern California Coastal Water Research Project and representatives from discharger and regulatory agencies in Southern California co-developed an “enhanced monitoring framework to comprehensively screen for a wide variety of contaminants, including CECs, and to better identify bioactive contaminants with the potential to impact the quality of receiving water environments” (Maruya et al., 2016). This approach starts with cell bioassays (e.g., estrogen receptor binding) complementing traditional targeted chemical monitoring to screen for both known and unknown chemicals according to mode of action. This screening can then be followed up with more traditional toxicity testing of individual contaminants and/or non-targeted chemical analysis to identify potential causative agents. This approach has been successfully used to prioritize sites for further monitoring in Southern California, using endocrine-responsive and aryl hydrocarbon receptor cell assays (Mehinto et al., 2017). A pilot study testing six sites in the Lower South Bay of San Francisco Bay for estrogenic activity detected no activity in water, but was less conclusive for sediment due to concerns around incomplete extraction of contaminants. Although sediment results were inconclusive, the study identified improvements that could be made to the methods for better results in the future (Denslow et al., 2018). A similar approach may be a useful screening tool for unknown CECs in future monitoring and assessment of CECs in San Francisco Bay.

To inform future study design and evaluation of toxicological risk, we have developed a “conceptual model” workflow for identifying appropriate ecotoxicological thresholds for assessment within the tiered risk-based framework (Figure 4). For compounds detected in Bay samples, thresholds based on whole-organism toxicity testing are preferred, if available. If whole organism testing-based thresholds are unavailable, thresholds derived from *in vitro* testing are preferred over those from *in silico* models. While QSARs and read-across methods to estimate toxicity are useful screening tools, they come with a higher degree of uncertainty than biological testing, as toxicity can vary considerably even within chemical classes. Predictive tools may be useful in identifying Possible Concern compounds that are not likely to cause effects in the Bay, and de-prioritizing these contaminants from further study. The quality of thresholds will be an important consideration for classifying CECs within the tiered risk-based framework. If the quality of available toxicity information is poor, a larger margin between occurrence data and ecological thresholds is necessary to identify contaminants as low risk. If available toxicity information is high quality, a smaller margin may be acceptable.

In vitro screening of environmental samples may be performed in cases where mixture effects and/or unknown contaminants are suspected to be a risk to aquatic health. This approach should focus first on major pathways to the Bay (e.g., stormwater), as these waters will have a stronger signal due to their higher concentrations. Focus should also be on the most relevant molecular initiating events and corresponding endpoints for CECs; while estrogenicity is perhaps the most well-understood toxicity pathway, other modes of action such as neurotoxicity or teratogenicity may be more important for Bay contaminants. Developing a bioactivity profile for Bay samples, in which multiple different molecular initiating events are screened and

compared, will help determine which AOPs are most relevant for CECs in the Bay. California is moving towards using toxicity for regulatory evaluation of chemicals such as pesticides in urban waters (California State Water Resources Control Board, 2020); although the focus will likely be on traditional whole-organism toxicity testing, *in vitro* screening is cheaper and requires a smaller sample volume, so can be used to evaluate more sites.

Quality evaluation of existing threshold information could also weigh into what kinds of ecotoxicity studies should be prioritized. In cases where there is a clear indication of need for further toxicity information on a specific CEC, the RMP can also use funds for more traditional toxicity evaluation. For example, past RMP-funded toxicity studies include investigating the toxic effects of polybrominated diphenyl ethers (PBDEs) on bird embryotoxicity (Rattner et al., 2013, 2011) and copper on salmonid olfactory response (Baldwin, 2015; Sommers et al., 2016).

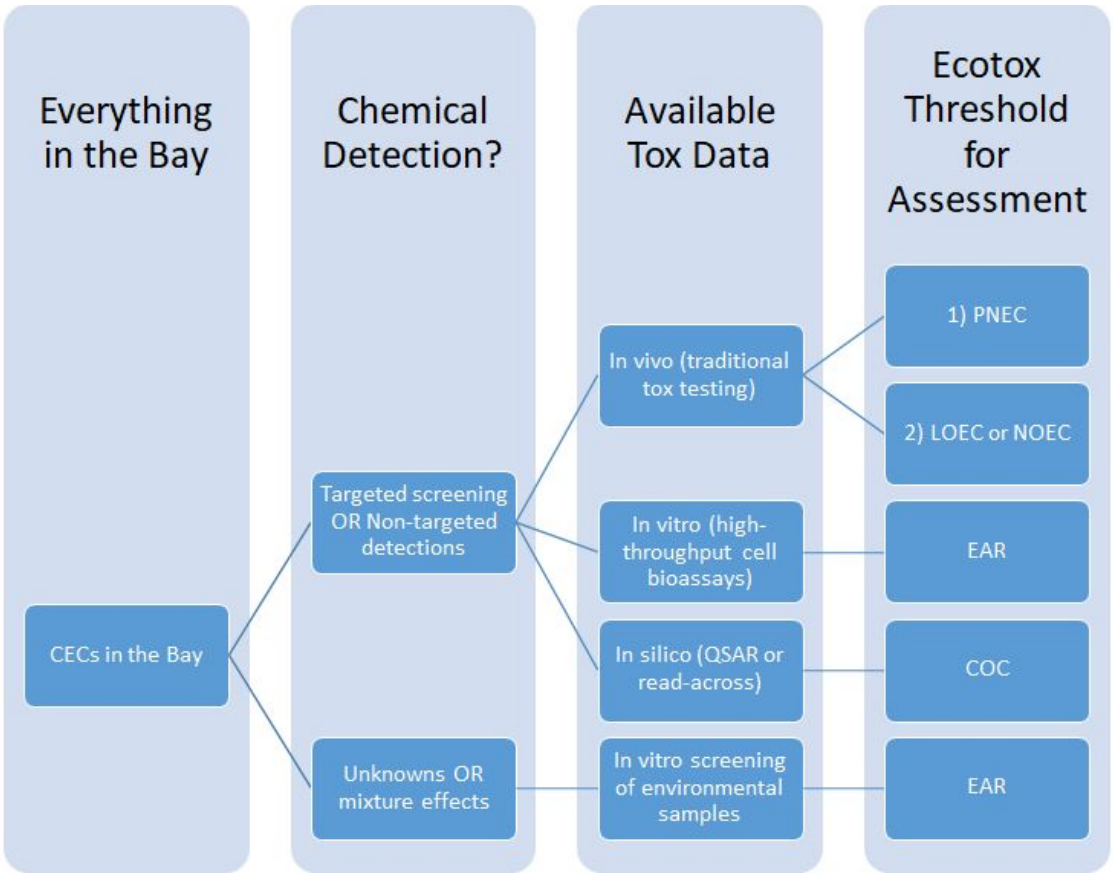

Figure 4. "Conceptual model" workflow for assessing potential impacts of CECs to San Francisco Bay biota based on available ecotoxicological data. Abbreviations: PNEC predicted no effect concentration for an ecosystem, LOEC or NOEC low or no observed effect concentration, EAR exposure:activity ratio, COC concentration of concern.

Suggested Next Steps

To date, the RMP has identified ecotoxicity thresholds for comparison with measured concentrations of CECs on a study-by-study basis, giving preference to the lowest available thresholds under the assumption that these will be most protective. However, this approach is inefficient and may not always result in using the most appropriate threshold value. For example, a threshold with higher uncertainty may be lower than one based on more experimental data, but the latter would be a more appropriate value to evaluate risk. A proposed method for improving and streamlining comparison of detected CEC concentrations with toxicity thresholds is to compile and evaluate the quality of available thresholds, and to use the methods discussed above to calculate thresholds for data-poor compounds.

To assess possible effects from unknowns (i.e., chemicals for which the RMP has not monitored) and contaminant mixtures, a possible screening study would be to assess biological activity of environmental samples using a battery of high-throughput bioassays that represent different modes of action. A focus on mode of action also allows observation of the sum of toxicity from environmental mixtures, rather than attempting to predict effects from single compound toxicity assessments. Environmental samples from pathways to the Bay would make sense as a starting place because contaminants would be more concentrated (and therefore easier to detect) and observed toxicity could be more readily linked back to management actions. Sample bioactivity profiles could then be compared with a) known adverse outcome pathways to predict possible effects on Bay wildlife and endpoints for targeted monitoring, b) known contaminants to predict which contaminants may be responsible for observed activity, and c) expected bioactivity based on known chemical concentrations to determine whether measured compounds, unknowns, or a combination are responsible for observed activity. Locations with higher sample bioactivity or bioactivity due to unknowns could then be prioritized for future monitoring and special studies. This type of approach could also be combined with effect-directed chemical analysis via targeted, suspect, and non-targeted screening approaches to identify new contaminants of concern (Brack et al., 2016).

Another approach the RMP could take in predictive toxicology is to develop predictive toxicology tools specific to important Bay species. For example, a joint collaboration between the Canadian government and academics is currently developing quantitative PCR arrays and corresponding data evaluation tools for both standard laboratory testing species and native species of commercial, recreational, and Aboriginal concern (http://www.ecotoxchip.ca/). Quantitative PCR arrays for Bay species could be used to characterize the possible toxicological effects of environmental chemicals and complex mixtures and prioritize contaminants or sites.

RMP CEC Multi-Year Plan

Assembled below are recommended studies that have grown out of the RMP CEC Strategy, structured as a multi-year plan (Table 4).The multi-year plan focuses on RMP Special Studies, but also provides information on Status and Trends and other RMP monitoring efforts relevant to CECs, along with external, pro bono collaborations.

Special Studies are primarily designed in response to the RMP priority questions for emerging contaminants:

- MQ1: Which CECs have the potential to adversely impact beneficial uses in San Francisco Bay?
- MQ2: What are the sources, pathways and loadings leading to the presence of individual CECs or groups of CECs in the Bay?
- MQ3: What are the physical, chemical, and biological processes that may affect the transport and fate of individual CECs or groups of CECs in the Bay?
- MQ4: Have the concentrations of individual CECs or groups of CECs increased or decreased in the Bay?
- MQ5: Are the concentrations of individual CECs or groups of CECs predicted to increase or decrease in the future?
- MQ6: What are the effects of management actions?

The purpose of this multi-year plan is to guide program management. These recommendations will be revisited and revised each year as part of the RMP budget planning process. The plan will be adapted to reflect advances in science and changes in policy needs.

MULTI-YEAR PLAN FOR EMERGING CONTAMINANTS

Special studies and monitoring in the RMP from 2015 to 2024. Numbers indicate budget allocations in $1000s. Budgets in parentheses represent funding or in-kind services from external sources. Items highlighted in blue are additions and changes from the multi-year plan approved in 2020.

Element	Study	Funder	Questions addressed	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Strategy	CEC Strategy[1] (not a Special Study after 2020)	RMP	1-6	20	48	50	65	70	75	*60*	*80*	*60*	*60*
	Stormwater Monitoring Strategy	RMP	1,2								50	50	
MODERATE CONCERN CECs													
PFAS	CECs in Municipal Wastewater[2]	RMP	1,2,4	27.5									
	Effluent TOP Analysis	DTSC	1,2,4,6	(50)									
	PFAS: Synthesis and Strategy	RMP	1-6			56							
	Margin Sediment Archiving	RMP	1				2.5						
	PFOS/PFOA Bay Model	Interwaste	1,2,3,5				(7)						
	Stormwater PFAS[3]	RMP	1,2					33	40	29.6			
	North Bay Margin Sediment PFAS ($40-$125k)	SEP proposal	1,2,4,6										
	PFAS in Ambient Bay Water	RMP	1,4,6							50			
	PFAS in Influent, Effluent, Biosolids; Study TBD, est. value	BACWA	1,2,4,6							(365)			
	Harbor Seal (PFAS and Nonpolar NTA; SEP proposal, ~$100k) [4]	SEP proposal	1,4,6										
	PFAS in Sewershed and Effluent	RMP	1,2,4								100		
	Air Deposition PFAS	RMP	1,2										100
	RMP Status and Trends[5]	RMP S&T	1,4		E		E	F		E			E, F
AP/APEs	Margin Sediment Archiving, Analysis	RMP	1,4				2.5						
	Stormwater Ethoxylated Surfactants[3]	RMP	1,2					33	40	29.6			
	Ethoxylated Surfactants in Water, Margin Sediment, and Wastewater	RMP	1,2,4					123					
	Archived Tissue	RMP	1,4								100		

Element	Study	Funder	Questions addressed	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Bisphenols	Bisphenols in Bay Water	RMP SIU	1	(25)		50						50	
	Bisphenols in Stormwater	RMP	1,2						21	29.6			
	Bisphenols in Wastewater, Sediment	RMP	1,2						72				
	Bisphenols in Sport Fish, Bivalves	RMP	1										80
Organo-phosphate Esters	Phosphate Flame Retardants in Ambient Bay Water	RMP ECCC	1,4			47						60	
	Stormwater Phosphate Flame Retardants[3]	RMP	1,2					33	40	29.6			
	Conceptual, Steady-State Model[7]	RMP	1,2,3,6								94		
Fipronil	CECs in Municipal Wastewater[2]	RMP	1,2,3	27.5									
	Fipronil, Fipronil Degradates, and Imidacloprid in Wastewater	RMP	1,2,3		30								
	Fipronil, Fipronil Degradates, and Imidacloprid in Biosolids	ASU	1,2,3		(8)								
	Conceptual and Quantitative Model	RMP	1,2,3,6										70
	RMP Status and Trends[5,6]	RMP	1,3,4				S	F			S		
Imida-cloprid	Imidacloprid, Imidacloprid Degradates and other Neonicotinoids in Bay Water	RMP	1			40							
	Wet Season Monitoring of Current-Use Pesticides and Imidacloprid[7]	RMP	1,2										10
LOW or POSSIBLE CONCERN CECs													
PBDEs	RMP Status and Trends[5]	RMP S&T	1,3,4		B, E		S, E	F		E	S		E, F
Alt. Flame Retardants	Brominated Flame Retardants in Bay Matrices	RMP	1,4									80	
Pharma-ceuticals	Pharmaceuticals in Wastewater	RMP POTWs	1,2,4		(68)		30						
	Antibiotics and QACs in Surface Sediment and Cores	U Minn	1,3,4				(8)						
	Pharmaceuticals in Water & (Archived) Sediment	RMP	1,2,4									180	

Element	Study	Funder	Questions addressed	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
Plastic Additives	Phthalates in Bay Matrices	RMP	1,4										70
Personal Care/ Cleaning	Triclosan in Small Fish	RMP	1			41							
	Musks in Water & Sediment[8]	RMP	1				64.5						
	Siloxanes in Sediment and Effluent	SWEAM DTSC	1,2				(15)						
	Sunscreens in Wastewater	MMP	1,2						(36.5)				
	New Concerns in Bay Water, Wastewater	RMP	1,2									80	
	QACs in Wastewater	MMP	1,2,4							(58.2)			
Pesticides	DPR Priorities in Water & Sediment[8]	RMP USGS	1,2,3				64.5 (6.8)						
	Ag Pesticides in Water & Sediment of North Bay Margins (~$100k)	SEP proposal	1,2										
	Wet Season Monitoring of Current-Use Pesticides and Imidacloprid[7]	RMP	1,2										90
PHCZs	Sediment, Tissue	SIU	1	(15)	(20)	(40)							
Brominated Azo Dyes	Archived Sediment (~$60k)	SEP proposal	1										
Construc-tion Related	Isothiazolinone Biocides and Other Contaminants in Stormwater (~$50k)	U Iowa SEP Proposal	1,2					(2)					
	New concerns	RMP	1										50
Chlorinated Paraffins	Chlorinated Paraffins in Sediment (~$60k, 2022)	SEP proposal	1										
NON-TARGETED & OTHER STUDIES													
Non-targeted	Non-targeted Analysis of Water-soluble CECs	RMP / Duke / AXYS	1,2		52 (10) (6)								
	Non-targeted Analysis of Sediment	RMP	1,2				101						
	Non-targeted Analysis of Runoff from North Bay Wildfires	RMP DTSC Water Brd Duke	1,2				36 (20) (27) (3)						

Element	Study	Funder	Questions addressed	2015	2016	2017	2018	2019	2020	2021	2022	2023	2024
	Follow-up Targeted Study, Stormwater[3]	RMP	1,2					33	40	29.6			
	Harbor Seal (PFAS and Nonpolar NTA; SEP proposal, ~$100k)[4]	SEP proposal	1,4,6										
	Follow-up Targeted Study (2018 sediment results)	RMP	1								100		
	Trash Hot Spots and Plastic Additives NTA Study	RMP	1									120	
	Follow-up Targeted Study (biota)	RMP	1										100
Other	Toxicology	RMP	1					15		60	60	60	60
RELEVANT STUDIES IN OTHER WORKGROUPS													
Bioassay (EEWG)	Linkage of In Vitro Estrogenic Assays with In Vivo End Points	RMP SCCWRP UF	1,2			45							
Modeling (SPLWG)	Integrated Monitoring and Modeling Strategy - CEC Conceptual Model	RMP	1,2,4							50			
RMP-funded Special Studies Subtotal - ECWG				**75**	**130**	**284**	**366**	**325**	**328**	**258**	***504****	***680****	***630****
RMP-funded CEC Strategy (not a Special Study after 2020)										**60**	***80***	***60***	***60***
RMP-funded Special Studies Subtotal – Other Workgroups				**0**	**0**	**45**	**0**	**0**	**0**	**50**			
RMP MMP & Supplemental Environmental Project Subtotal				**0**	**0**	**0**	**0**	**0**	**36.5**	**58.2**			
Pro-Bono & Externally Funded Studies Subtotal				**90**	**112**	**90**	**37**	**2**	**0**	**365**			
OVERALL TOTAL				**165**	**242**	**419**	**403**	**327**	**514.5**	**791.2**	***584***	***740***	***690***

*Several proposed studies for future years will be developed as SEP proposals rather than RMP proposals.

1 – The CEC Strategy funds preparation of RMP CEC Strategy Revisions, Updates, and Memos; it also funds literature review, scientific conference attendance, and responses to information requests from RMP stakeholders. A Revision to the CEC Strategy is planned for 2021, resulting in a higher funding request than in the prior years. While previously considered a Special Study, the CEC Strategy is now considered part of program management.
2 – The 2015 CECs in Municipal Wastewater study ($55k) included analyses of PFAS and fipronil; the budget has been split between these two groups.
3 – The multi-year (2019-2021) stormwater study includes five groups of analytes: PFAS, ethoxylated surfactants, organophosphate esters, bisphenols (added year 2), and targeted stormwater analytes identified via non-targeted analysis. The total cost ($461k) is spread across five groups and three years.
4 – The proposed non-targeted analysis of harbor seal tissues includes investigations of PFAS (2022; $100k) and nonpolar compounds (2021; $100k).
5 – When a CEC is proposed for inclusion in the the RMP Status and Trends monitoring, there is a letter in the cell denoting the matrix for which monitoring is proposed: W = water; S = sediment; B = bivalve; E = eggs; F = fish.
6 – Analysis of fipronil and fipronil degradates in sediment has been added to the RMP Status and Trends monitoring effort for 2018.
7 – The proposed study includes imidacloprid and other current-use pesticides; the projected budget has been split (10%, 90%) between these groups.
8 – The 2018 Musks and DPR Priorities in Water and Sediment study ($129k) included analyses of pesticides and fragrance ingredients; the budget has been split between these two groups.

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