CECs in California’s Ambient Aquatic Ecosystems:

Occurrence and Risk Screening of Key Classes

Final Report; March 2022

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ASC Contribution #1066

Suggested citation:

Sutton, Rebecca; Miller, Ezra; Wong, Adam; Mendez, Miguel; Lin, Diana. 2022. CECs in California’s Ambient Aquatic Ecosystems: Occurrence and Risk Screening of Key Classes. ASC Technical Report: Contribution #1066. Aquatic Science Center, Richmond, CA.

Acknowledgements

Funding for this study was provided by the California State Water Resources Control Board (Agreement No. 19-015-120). The authors gratefully acknowledge the reviews provided by project expert advisors Derek Muir (Environment and Climate Change Canada), David Sedlak (University of California at Berkeley), Dan Villeneuve (US Environmental Protection Agency), and Inge Werner (Swiss Centre for Applied Ecotoxicology). Many others contributed thoughtful perspectives to inform the scope of this effort, including staff and representatives from California Department of Toxic Substances Control, Safer Consumer Products Program; California Office of Environmental Health Hazard Assessment; California Department of Pesticide Regulation; Representatives from the State and Regional Water Boards; California Ocean Protection Council; Bay Area Clean Water Agencies; Orange County Water District; California Stormwater Quality Association; California Coastkeeper Alliance; San Francisco Baykeeper; Heal the Bay; Green Science Policy Institute; and the Southern California Coastal Water Research Project. Additional guidance was provided by the State Water Board's Aquatic Ecosystems CEC Advisory Panel II (EcoPanel). Aquatic Science Center staff Don Yee, Kelly Moran, Melissa Foley, and Jay Davis also provided valuable guidance and review. The authors are particularly grateful for the support and assistance from project managers and staff with the State Water Board (Erica Kalve and Sarabeth George) and the San Francisco Bay Regional Water Quality Control Board (Tom Mumley and Margaret Monahan).

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Executive Summary

Constituents of emerging concern (CECs) may be defined as contaminants that are largely un- or under-regulated in the US, or have limited regulation in environmental media (e.g., air, water, sediment, and biota) around the world. CECs encompass a vast number and wide variety of substances, including chemical compounds ranging from pharmaceuticals to flame retardants to current-use pesticides; physical materials and radioactive substances; and numerous biological agents (e.g., bacteria, viruses, antibiotic resistant genes). For purposes of this report, CECs to be examined will include synthetic organic chemicals that have the potential to enter the environment and cause adverse ecological or human health impacts.

For most CECs, major information gaps hinder the ability of scientists and regulators to accurately assess potential risks to inform effective monitoring and management priorities. The State and Regional Water Boards have established a statewide Constituents of Emerging Concern Program (CEC Program) that will coordinate CEC monitoring and management efforts in the state and provide a framework for statewide prioritization and management of CECs. With this report, scientists at the Aquatic Science Center (ASC) are supporting the CEC Program by synthesizing available CECs data for multiple classes of contaminants in California, and screening observed occurrences via a tiered risk-based framework.

CECs encompass thousands of chemicals that cannot be comprehensively reviewed in a single report, so we focused on key classes of CECs defined either by chemical properties or structure (alkylphenols and alkylphenol ethoxylates; bisphenols; organophosphate esters; phthalates; PFAS) or by function or use in society (current-use pesticides [and degradates] with urban applications; pharmaceuticals; personal care and cleaning product ingredients). An additional class, brominated flame retardants, including PBDEs, is defined by both chemical and use properties. By considering individual compounds as members of broader classes, we are able to more efficiently and systematically evaluate and address potential environmental impacts (Section 1). For example, the use of chemical classes can provide a means of evaluating data-poor compounds, through comparison with chemically similar, data-rich compounds. Likewise, use classes can provide a framework for reducing contamination from many compounds via a single management action, by addressing a specific function or use rather than a specific chemical.

The classes we selected were prioritized based on discussions with experts and stakeholders representing regulatory agencies, regulated stakeholders, non-governmental (advocacy) organizations, scientific institutes, and State and Regional Water Board staff (Section 2). We also included additional chemical and use classes relevant to individual compounds identified as monitoring priorities in a previous convening of the State Water Board’s Aquatic Ecosystems CEC Advisory Panel (EcoPanel), which examined contaminants from urban settings. We focused our evaluations on ambient surface waters in freshwater, marine, and estuarine ecosystems, and included evaluations for sediment and biota for a subset of chemical classes as an example of application across multiple matrices.

Our specific objective is to demonstrate a proof of concept for this risk screening approach, providing the Water Boards with a model and the ability to move forward with broader implementation as part of the statewide CEC Program. Our risk screening approach can be further refined as new information is gathered, including improved toxicity thresholds and updated monitoring trigger levels (MTLs), and recommendations being developed by the currently convened EcoPanel. Risk screening can be used to inform priorities for monitoring and management, but does not include specific monitoring designs for contaminants or classes (e.g., minimum sample numbers and monitoring frequency), which will depend on whether data are to be used for screening purposes or more sophisticated assessments of temporal trends.

Ecological risks are the primary driver of this screening level evaluation, with additional consideration of human health risks relating to consumption of sport fish and shellfish (Section 2). An important part of this exercise was to compile ambient aquatic ecosystem occurrence data for the state from public databases with California monitoring data (i.e., CEDEN, Water Quality Portal, SDWIS, CIWQS, Geotracker), supplemented with relevant published peer-reviewed literature. We also demonstrated a pre-screening step for a subset of the contaminants reviewed using previously established MTLs to establish the need for further risk screening using the tiered risk-based framework. Toxicological thresholds used for risk screening were selected based on best currently available ecological risk thresholds, which we identified using various criteria, such as thresholds that a) are derived from well established experimental data (rather than predicted values); b) incorporate data from a greater number and diversity of species (e.g., species sensitivity distributions); c) are based on chronic exposures (rather than acute); d) have been incorporated or adopted in state, federal, or international regulations; and e) are transparent regarding their derivation. Human health fish consumption thresholds were also compiled for evaluation, when available. Drinking water occurrence data and associated risk assessment was beyond the scope of this effort. It is important to note that our understanding of CECs is rapidly evolving, and future screening activities should rely on the most up-to-date and robust toxicity thresholds, where available.

Ecological risk quotients (RQs) were calculated for each compound in each matrix and setting (e.g., freshwater, estuarine water) as the ratio of each compound's 90th percentile concentration to its appropriate ecological risk threshold. Compounds or classes were assigned to risk tiers (Low, Moderate, High, Very High Concern) based on the highest RQs observed for the matrix and settings examined. Compounds or classes where there was insufficient understanding of occurrence levels or toxicity thresholds were assigned to the Possible Concern category. For chemical classes in which individual compounds are grouped primarily by chemical similarities, we indicated a class-wide risk tier based on the highest RQ and associated risk tier for an individual member of the class. In contrast, compounds in use classes received individual rather than class-wide tier assignments due to greater diversity in both chemistry and toxicity (i.e., modes of action).

We emphasize that the outcomes of this risk screening exercise, based on aggregated data spanning a broad geographic and temporal range, are not intended to indicate a current threat relating to any individual aquatic ecosystem in the state. Instead, we intend to provide high-level information to inform broader priorities and decisions regarding monitoring and management.

The highest risk tier, Very High Concern, contains per- and polyfluoroalkyl substances (PFAS), polybrominated diphenyl ethers (PBDEs), organophosphate esters, and fipronil and degradates (Section 3). PFAS and PBDEs have human health concerns because concentrations reported in fish are above human health advisory levels. Organophosphate esters and fipronil and degradates have been observed at concentrations in freshwater, marine, or estuarine waters at concentrations where there is a very high probability of an adverse impact on California wildlife (RQs greater than 100).

In the High Concern tier, we classified imidacloprid, pyrethroids, and some specific pharmaceuticals (ibuprofen, fluoxetine, and azithromycin). For these compounds, RQs between 10 and 100 indicate a high probability of an adverse impact on California wildlife. Next, we classified bisphenols and some additional pharmaceuticals (metoprolol, gemfibrozil, ciprofloxacin, clarithromycin, and erythromycin) as Moderate Concern. Concentrations of these compounds or classes in surface waters indicate a moderate probability of an adverse impact on California wildlife (RQs between one and ten). In the lowest risk tier category, Low Concern, we classified the antimicrobial triclosan and the synthetic musk galaxolide, as occurrence data suggest minimal impact on California wildlife (RQs less than one). Of note, the risk screening for both of these contaminants was for water only, and did not include an evaluation of risk based on occurrence in sediment.

A significant limitation of the compiled occurrence data is that a large portion of the data were generated using analytical methods that are insufficiently sensitive (detection limits above toxicological thresholds) to inform risk screening. We classified alkylphenols and alkylphenol ethoxylates, phthalates, and the hormones 17α-ethinylestradiol and 17β-estradiol as Possible Concern due to uncertainty with respect to occurrence because of inadequate method sensitivity. Alternative brominated flame retardants other than PBDEs were classified as Possible Concern due to limited occurrence data and toxicity thresholds for these compounds.

We also provided a brief review of broader global scientific efforts to identify and analyze new CECs and assess risks to wildlife and humans using new tools and approaches (Section 4). Our review suggested more expansive monitoring is possible for some of the classes reviewed in this report, such as PFAS and alkylphenol ethoxylates. Such monitoring may take advantage of strategic application of exploratory analytical methods. We further synthesized findings from novel suspect screening and non-targeted analysis of environmental samples from California, along with in silico screening of inventories of chemicals in commerce, to identify new classes that merit monitoring consideration. These new compounds and classes include additional brominated and chlorinated compounds; siloxanes; additional plastic ingredients; vehicle-related compounds such as 6PPDQ, a contaminant derived from vehicle tires; and quaternary ammonium compounds (QACs). New scientific findings can continue to inform the Water Boards CEC Program to anticipate new CECs and address data gaps limited by current traditional monitoring methods and approaches.

We developed recommendations for Water Boards monitoring and management actions for each class, which were informed by the risk screening; additional factors such as persistence in the environment, bioaccumulation, and changing use patterns; and current management actions (Table ES.1 and Section 5). A class-based approach for monitoring and management is an important tool to reduce regrettable substitution, a term that describes the replacement of individual compounds targeted for management by data-poor compounds that are later discovered to have ecological and/or human health risks.

We recommend screening level monitoring for most compounds and classes reviewed to increase the availability of current occurrence data across ecological settings (freshwater, estuarine, marine), using methods with sufficient sensitivity for risk screening. For a handful of compounds or classes subject to significant management actions (PFAS, PBDEs, organophosphate esters, triclosan, triclocarban), monitoring to assess temporal trends and evaluate the impacts of management may be useful. Monitoring PFAS and PBDEs in sport fish and seafood is important to address risks to human health. Finally, additional classes of contaminants not included in this risk screening exercise that the Water Boards may wish to consider for screening level monitoring include 6PPDQ and other vehicle and tire-related compounds, plastic ingredients, chlorinated paraffins, and siloxanes.

With respect to potential management actions, we recommend the Water Boards conduct assessments of the diverse sources or uses of chemical classes (alkylphenols and alkylphenol ethoxylates; bisphenols; organophosphate esters; phthalates; PFAS), to identify sources or uses that may contribute to significant levels of environmental contamination. Ingredient transparency initiatives relating to PFAS, as well as personal care and cleaning products, can guide source identification exercises. Identified sources can be addressed via pollution prevention activities; the Water Boards can use this information to support sister agencies with regulatory authority over formulation and use of consumer products and pesticides, and inform legislation relating to CECs.

Table ES.1: Summary of risk screening and monitoring and management recommendations for CEC classes evaluated in this report.

Class	Risk Screening Tier	Past & Current Management Actions	Additional Factors	Monitoring Recommendations	Management Recommendations
Alkylphenols & Alkylphenol Ethoxylates	Possible Concern	- Product-specific management actions	Persistence	- Screening level monitoring in surface waters, sediment. Apply methods with sufficient sensitivity.	- Identify and address major sources (e.g., through DTSC). Use a class-based approach to avoid regrettable substitution.
Bisphenols	Moderate Concern	- Product-specific management actions - Proposition 65 warnings	Regrettable substitution	- Screening level monitoring in surface waters, sediment for BPA, BPF, BPS. Apply methods with sufficient sensitivity.	- Identify and address major sources (e.g., through DTSC). Use a class-based approach to avoid regrettable substitution.
Organophosphate Esters	Very High Concern	- Changes to flammability standards - Product-specific management actions - USEPA regulation of individual compounds	Regrettable substitution Trends in use	- Screening level monitoring. Apply methods with sufficient sensitivity. - Evaluate temporal trends.	- Identify and address major sources (e.g., through DTSC). Use a class-based approach to avoid regrettable substitution. - Address other flammability standards.
Phthalates	Possible Concern	- Restriction in products for children - Proposition 65 warnings - USEPA risk evaluations	None	- Screening level monitoring. Apply methods with sufficient sensitivity.	- Identify and address major sources (e.g., through DTSC). Use a class-based approach to avoid regrettable substitution.
Brominated Flame Retardants, including PBDEs	**PBDEs:** Very High Concern **Others:** Possible Concern	- Bans and phase-out of PBDEs - Changes in flammability standards - Product-specific management actions	Persistence Bioaccumulation PBDE occurrence trends	- Evaluate temporal trends of PBDEs. - Monitor fish for PBDEs to protect people. - Screening level monitoring for other brominated flame retardants.	- Address other flammability standards.
Current-use Urban Pesticides	**Fipronil, degradates:** Very High Concern **Imidacloprid:** High Concern **Pyrethroids:** High Concern	- USEPA and CDPR registration and oversight	Trends in use	- Coordinate with CDPR.	- Provide support to CDPR.
Pharmaceuticals	**Ibuprofen, fluoxetine, azithromycin:** High Concern **Metoprolol, gemfibrozil:** Moderate Concern **Ciprofloxacin, clarithromycin, erythromycin:** Moderate Concern **17α-ethinylestradiol, 17β-estradiol:** Possible Concern	- Policies to reduce unnecessary use of antibiotics - Safer disposal programs	Trends in use	- Screening level monitoring. Apply methods with sufficient sensitivity.	- Support efforts to reduce unnecessary use of antibiotics. - Support expansion of existing drug take-back programs.
Personal Care & Cleaning Products	**Triclocarban:** Moderate Concern **Triclosan, galaxolide:** Low Concern **Tonalide:** Possible Concern	- Product-specific management actions - Fragrance ingredients disclosure	Persistence	- Screening level monitoring. Apply methods with sufficient sensitivity. - Evaluate temporal trends of triclosan, triclocarban.	- Support efforts to reduce unnecessary use of antimicrobials. Use a class-based approach to avoid regrettable substitution. - Support ingredient disclosure.
PFAS	Very High Concern	- Phase-out of individual compounds - Product-specific management actions - USEPA proposed PFAS reporting requirement - State Water Board PFAS Action Plan	Persistence Bioaccumulation Regrettable substitution	- Screening level monitoring in surface waters, sediment, fish. Apply methods with sufficient sensitivity. - Evaluate temporal trends. - Consider strategic use of exploratory PFAS screening methods.	- Support reporting requirements, inventories. - Identify and address major sources (e.g., through DTSC). Use a class-based approach to avoid regrettable substitution. - Support inter-agency collaboration to address ecological concerns. - Support elimination where non-essential.
New CECs	Not Screened			- Screening level monitoring of: 1) 6PPDQ and other vehicle and tire-related compounds 2) Plastic ingredients 3) Chlorinated paraffins 4) Siloxanes	

1. Background and Study Objectives

Constituents of emerging concern (CECs) may be defined as contaminants that are largely un- or under-regulated in the US, or have limited regulation in environmental media (e.g., air, water, sediment, and biota) around the world. CECs encompass a vast number and wide variety of substances, including chemical compounds ranging from pharmaceuticals to flame retardants to current-use pesticides; physical materials and radioactive substances; and numerous biological agents (e.g., bacteria, viruses, antibiotic resistant genes). For purposes of this report, CECs to be examined will include several classes of synthetic organic chemicals that have the potential to enter the environment and cause adverse ecological or human health impacts. For most CECs, major information gaps hinder the ability of scientists and regulators to accurately assess potential risks to inform effective monitoring and management priorities.

In 2021, the State and Regional Water Boards (Water Boards) established a new Constituents of Emerging Concern Program (CEC Program) in the Division of Water Quality. Prior to that, the Water Boards have been working collaboratively to coordinate ongoing CEC monitoring and management efforts in the state and provide a framework for statewide prioritization and management of CECs. The initial activities are broken into three phases: compile existing knowledge and resources; improve coordination between the Water Boards and other groups; and develop and implement a statewide CEC management strategy.

To support the CEC Program in compiling existing knowledge and resources, the Aquatic Science Center (ASC) was tasked with a data synthesis and risk screening exercise to inform recommendations for monitoring and management of CECs in ambient state waters. The broad goals of this report are to: 1) synthesize and evaluate the significance of available CECs water quality data in California through screening via a tiered risk-based framework; and 2) identify priorities for monitoring and management of CECs in freshwater, estuarine, and marine settings in California. Our specific objective is to demonstrate a proof of concept for this risk screening approach, providing the Water Boards with a model and the ability to move forward with broader implementation as part of the statewide CEC Program.

An integral element of this exercise is the use of a class-based approach when evaluating occurrence, risk, and monitoring and management priorities for CECs. By considering individual chemical contaminants as members of broader classes defined by similar chemical structures (e.g., bisphenols, a chemical class) and/or function or use in society (e.g., pharmaceuticals, a use class), we are able to more efficiently and systematically evaluate and address potential environmental impacts. For example, members of a chemical class may have similar properties with respect to persistence, bioaccumulation, and/or toxicity; a class-based approach can provide an early alert as to potential concerns of a poorly studied compound that is chemically similar to a well-established toxicant. Similarly, members of a use class might be addressed by a single management action that affects product requirements (e.g., flammability standards that impact use of flame retardants) or disposal practices (e.g., pharmaceutical take-back programs). Members of a chemical or use class may be substituted within formulations or other products in response to the phase-out or ban of a well-studied and clearly toxic compound in the same class; in this case, a class-based approach can indicate which chemicals may see increasing use as substitutes and, therefore, merit future monitoring. Class-based approaches are generally recognized as a useful element of many forward-looking frameworks for contaminant monitoring and management, and have been adopted by an increasing number of science and regulatory agencies in California (e.g., California Biomonitoring; California Department of Toxic Substances Control, Safer Consumer Products Program [Bălan et al., 2021]) and abroad (e.g., Kwiatkowski et al., 2020).

As a group, CECs encompass thousands of substances, and cannot be comprehensively reviewed in a single report. This data synthesis and risk screening exercise focuses on key classes of synthetic organic chemical contaminants that have the potential to enter the environment and cause adverse ecological or human health impacts. The classes we selected were prioritized based on discussions with experts and stakeholders representing regulatory agencies, regulated stakeholders, non-governmental (advocacy) organizations, scientific institutes, and Water Boards staff (Section 2). We also included additional chemical and use classes relevant to individual compounds identified as monitoring priorities in a previous convening of the State Water Board's EcoPanel. The overall scope of the report was further refined through discussions with the Water Boards.

The risk screening exercise we demonstrate in this report is informed by recommendations from the first EcoPanel, which was convened in 2009 on behalf of the State Water Board "to provide unbiased science-based recommendations for monitoring chemicals of emerging concern in oceanic, brackish and fresh waters across the state that receive discharge of treated municipal wastewater effluent and stormwater" (Anderson et al., 2012). The first EcoPanel developed a number of recommendations for monitoring strategies to address CECs in ambient aquatic ecosystems in California. They established a process for calculating monitoring trigger levels (MTLs) for individual CECs, which can be used to identify monitoring priorities in freshwater, estuarine, and marine settings (Anderson et al. 2012). Following initial, screening level CECs monitoring, the EcoPanel outlined a "Conceptual Tiered Risk and Action Based Monitoring Approach" to guide adaptive monitoring and management. Subsequently, the Regional Monitoring Program for Water Quality in San Francisco Bay adapted this tiered risk-based framework for CECs, and has used it to guide Bay monitoring activities for almost a decade (Sutton et al., 2017; Lin, Sutton, Shimabuku et al., 2018). The tiered risk-based framework described in this report is a further refinement of the original recommendations of the EcoPanel, and is intended to be a straightforward tool that can be readily applied to a wide range of CECs as an initial investigation of potential risks.

Risk screening approaches like the one described here are valuable in establishing an initial indication of potential risk, particularly when data are limited, as is generally the case for CECs. The tiered risk-based framework is especially useful in that it allows for multiple classifications of risk. Along with Very High, High, Moderate, and Low Concern tiers, we include a tier of Possible Concern, where uncertainties in occurrence and/or toxicity information prevent a more definite risk-based classification. It is not uncommon for data gaps to prevent a more accurate risk assessment for individual CECs or classes of CECs.

It is important to note that the outcomes of this risk screening exercise, based on aggregated data spanning a broad geographic and temporal range, are not intended to indicate a current threat relating to any individual aquatic ecosystem in the state. Instead, we intend to provide high-level information to inform broader priorities and decisions regarding monitoring and management.

Our understanding of CECs is rapidly evolving, requiring strong engagement with the latest science and an adaptive approach to assure that the Water Boards activities are effective in protecting California waters. We have included a review of some new tools and approaches that may be useful for informing forward-looking efforts to tackle newly identified CECs, including use of more exploratory targeted analytical methods to quantify additional compounds, as well as techniques to identify new or unexpected contaminants such as suspect screening and non-targeted analysis, and in silico screening of chemicals in commerce (Section 4). By tracking the science that emerges from these tools and approaches, the Water Boards and collaborators can identify additional CECs of interest beyond those reviewed here, informing monitoring and management priorities.

The risk screening information provided in this report is one of many factors to be considered in establishing priorities for monitoring and management. Additional factors we considered in developing recommendations for the Water Boards included: trends in use and environmental release; persistence in the environment; geographic distribution of data; anticipated exposure to wildlife and humans; availability of toxicity information; limitations of available analytical methods; and current and/or upcoming management actions (Section 5). For the Water Boards consideration, we have assembled a wide array of potential monitoring priorities intended to inform decision-making, as well as management actions designed to reduce environmental contamination. As the Water Boards identifies high priority actions as part of the CEC Program, selection criteria might include the anticipated water quality improvements and overall feasibility of each of these recommendations.

In parallel with this CEC synthesis and risk screening exercise, the EcoPanel was re-convened in 2020 to provide additional support to the CEC Program. These two activities are meant to inform and complement each other in service to the Water Boards.

2. Methods

In this section, we summarize the methods used to develop the synthesis of state occurrence data and the risk screening analysis presented in Section 3. Elements of our approach include solicitation of input from stakeholders and experts to shape the scope of this exercise; exploration of the use of monitoring trigger levels (MTLs) as a pre-screening step to identify contaminants of interest; compilation of occurrence data and toxicity thresholds; and development and application of the tiered risk-based framework.

2.1. Refining the Scope of the Project through Stakeholder and Expert Interviews

The specific scope of the project, and especially the list of CEC classes evaluated as part of this synthesis and risk screening exercise was developed through extensive discussion with stakeholders representing various regulatory agencies, regulated stakeholders (e.g., wastewater and stormwater agency associations), non-governmental and advocacy organizations, and scientific institutes. Entities represented included:

- California Department of Toxic Substances Control, Safer Consumer Products Program
- California Office of Environmental Health Hazard Assessment
- California Department of Pesticide Regulation
- Representatives from the State and Regional Water Boards
- California Ocean Protection Council
- Bay Area Clean Water Agencies
- Orange County Water District
- California Stormwater Quality Association
- California Coastkeeper Alliance
- San Francisco Baykeeper
- Heal the Bay
- Green Science Policy Institute
- Southern California Coastal Water Research Project

All stakeholders were asked the same set of questions designed to understand the stakeholder's perspective on priority CEC data needs (Appendix E). The goal was to learn from previous processes and ensure this effort would be helpful for the larger community. Additionally, we wanted to understand what priority data gaps stakeholders identified. Detailed description of comments received from the stakeholders are provided in an ASC memo to the Water Boards (Lin et al., 2020), and major themes are summarized in the project workplan (Aquatic Science Center, 2020) and Appendix E.

In addition, ASC has incorporated input from the EcoPanel and four independent scientific experts:

- Derek Muir, Environment and Climate Change Canada (also an EcoPanelist)
- David Sedlak, University of California at Berkeley
- Dan Villeneuve, US Environmental Protection Agency
- Inge Werner, Swiss Centre for Applied Ecotoxicology

Stakeholder and expert feedback indicated a preference for evaluating CECs using a class-based approach, as described in Section 1, and were integral to establishing which classes to include within this data synthesis and risk screening exercise. Multiple stakeholders suggested prioritizing work on classes including PBDEs, brominated flame retardants, organophosphate esters, phthalates, and PFAS. The scope of the pesticides evaluation, including its focus on urban applications, was established in coordination with the California Department of Pesticide Regulation (CDPR). We also included additional chemical and use classes relevant to individual compounds identified as monitoring priorities in the 2012 EcoPanel report (Anderson et al., 2012), which focused on contaminants from urban (as opposed to agricultural) settings, including alkylphenols and alkylphenol ethoxylates, bisphenols, pharmaceuticals, and personal care and cleaning product ingredients. Microplastics were excluded from the scope of this report due to the development of an independent statewide microplastics strategy (California Ocean Protection Council, 2022). Biological agents such as antibiotic-resistant genes and bacteria were also excluded, due to limitations in the scientific understanding of antibiotic resistance in the environment, which have prevented development of robust effects thresholds specific to these contaminants (Singer et al., 2016).

We focused on evaluation of ambient environmental monitoring data for samples collected within ~15 years of the project start date (i.e., 2005-present) using targeted (quantitative) analytical methods. The complete list of classes of contaminants for which we compiled available occurrence data for evaluation through the tiered risk-based framework are:

- Alkylphenols and alkylphenol ethoxylates
- Bisphenols
- Organophosphate esters
- Phthalates
- Brominated flame retardants, including polybrominated diphenyl ethers (PBDEs)
- Current-use pesticides (and degradates) with urban applications
- Pharmaceuticals
- Personal care and cleaning product ingredients
- Per- and polyfluoroalkyl substances (PFAS)

The risk screening focuses primarily on ambient water data from freshwater, estuarine, and marine settings. For a subset of classes, an additional risk screening of available sediment, fish, and bivalve data is provided as a model of a multi-matrix risk screening.

The subset of classes were selected to demonstrate application on contaminants with a range of available occurrence and toxicity data.

Some stakeholders indicated that establishing a process for risk screening and prioritization of which CECs to monitor is more valuable than providing a static list of monitoring and management priorities, as the latter can become outdated quickly. An important goal of this report is to provide the Water Boards and broader stakeholder community with a model risk screening approach that is clear and easy to implement more broadly as new CECs are identified and more monitoring and toxicity data become available. This work product is best considered a proof of concept for a process that can be adapted and implemented into the future.

To further aid the Water Boards in maintaining a forward-looking perspective on CECs, we included a discussion of new scientific tools and approaches (Section 4), which can inform monitoring and management priorities beyond the compounds included in the scope of this risk screening exercise. Specifically, we identified classes that may be of interest for future monitoring based on improvements to targeted analytical methods (Section 4.1), suspect screening and non-targeted analysis of environmental samples (Section 4.2), and in silico screening of chemicals in commerce (Section 4.2). Given the rapid pace of chemical and product development worldwide, these tools and approaches can be used to identify and analyze new CECs and assess risks to wildlife and humans. The Water Boards are well-positioned to track the state of the science and address newly identified emerging contaminants more efficiently and effectively via the statewide CEC Program.

2.2. Monitoring Trigger Levels: A Tool for Identifying CECs of Potential Interest for Monitoring and Management

In 2012, the EcoPanel proposed a chemical-by-chemical risk-based approach for screening CECs to identify those with the greatest potential to pose a risk to either ecological receptors or human health (Anderson et al., 2012). This risk-based screening process includes development of monitoring trigger levels (MTLs), which can serve as a tool to conservatively identify individual CECs that could be considered for further monitoring in different types of ecosystems. In this section, we briefly describe the process the EcoPanel developed, then evaluate state occurrence data for the contaminants the EcoPanel identified as monitoring priorities. For contaminants that were evaluated previously by the EcoPanel, the resulting monitoring trigger quotient (MTQ; described below) can be used as a pre-screening step in our overall risk screening, to see whether further analysis is valuable.

The EcoPanel risk-based screening process includes four steps:

1. Develop monitoring trigger levels (MTLs) for CECs that pose the greatest potential risk to aquatic systems based on published effects concentrations.
2. Compile measured or predicted environmental concentrations (MECs or PECs) for CECs for which MTLs could be estimated.

3. Identify those CECs that have the greatest potential to pose a risk by comparing MECs (or PECs) to MTLs. CECs with a monitoring trigger quotient (MTQ = MEC (or PEC)/MTL) greater than “1” are then identified for monitoring. (Note that an MTQ of greater than 1.0 does not indicate a risk is present, only that sufficient potential for a risk exists such that the chemical should be considered for inclusion in a monitoring program.)
4. Apply the risk-based screening method (steps 1-3) to each of three representative scenarios that capture the key types of exposure (sources and fate) to CECs in the state’s inland, coastal, and marine receiving water systems.

The development of MTLs is a critical first step in the process, as it provides non-regulatory and conservative screening values for identifying CECs that warrant further evaluation. The process includes review of toxicological literature to identify lowest observed effect concentrations (LOECs) and no observed effect concentrations (NOECs) from studies of reproduction and growth or survival of fish and invertebrates to screen individual CECs. For non-microbial effects, NOECs for reproduction and chronic growth/survival were adjusted using safety factors. The safety factors account for 1) extrapolation of freshwater effects data to saltwater species; 2) CECs having specific modes of action on developmental, neuroendocrine or immunological targets in eukaryotic organisms in the literature; and 3) CECs with an unknown mode of action.

At the time the aquatic ecosystem MTLs were developed, the risk-based screening focused on CECs for which occurrence and toxicity information could be obtained, giving priority to those data representing California sources and receiving waters. The EcoPanel also refined the focused universe of chemicals to those with NOECs <0.1 mg/L; if NOECs were not available, acute LC50s (lethal concentration for 50% of the population) were utilized. The EcoPanel recognized the discrepancies associated with using deterministic metrics of toxicity (i.e., NOECs) and recommended that MTLs be regularly revisited as additional data become available so that studies that use a more probabilistic assessment of toxicity (i.e., species sensitivity distributions) may be used.

Sections 4, 6, and Appendix D of the 2012 EcoPanel Report provide detailed information on the development of the MTLs recommended for CEC monitoring (Anderson et al., 2012). Based on the limited set of compounds where toxicity and occurrence data were available, the EcoPanel published MTLs for 16 CECs that were recommended for further monitoring. An additional CEC (fipronil) was added to this list, as documented in related pilot study guidance (Dodder et al., 2015). Table 2.1 provides a summary of currently available MTLs.

Table 2.1. Table of Monitoring Trigger Levels (Anderson et al., 2012, Dodder et al., 2015)

Compound	CEC Class	Freshwater (µg/L)	Estuarine Water (µg/L)	Freshwater Sediment (µg/kg)	Estuarine Sediment (µg/kg)	Marine Sediment (µg/kg)	Biota (ug/kg dw)
Bis(2-ethylhexyl) phthalate	phthalates	--	--		--	130	--
Bisphenol A	bisphenols	0.06	0.006		--	--	--
Bifenthrin	pesticides	0.0004	0.00004		0.052	--	--
Butylbenzyl phthalate	phthalates	--	--		--	6.3	--
Permethrin	pesticides	0.001	0.0001		0.73	--	--
Chlorpyrifos	pesticides	0.005	0.001		--	--	--
Fipronil	pesticides	0.042	0.005	0.09	6.5		
Estrone	pharmaceutical	0.006	0.0006		--	--	--
Ibuprofen	pharmaceutical	0.1	--		--	--	--
17-beta estradiol	pharmaceutical	0.002	0.0002		--	--	--
Diclofenac	pharmaceutical	0.1	--		--	--	--
Galaxolide	PCCP	0.7	0.07		--	--	--
Triclosan	PCCP	0.25	--		--	--	--
Nonylphenol (NP)	AP/APEs	--	--		--	14	--
PBDE-47 and PBDE-99	PBDE	--	--		0.03	0.3	28.9
PFOS	PFAS	--	--		--	--	1000

Where MTLs are available, they are expected to represent a cautious and conservative approach for assessing potential risks; we recommend using them as a pre-screening step in the present risk screening. If maximum concentrations in ambient ecosystems of California are below MTLs (MTQ < 1), then monitoring may not be a high priority for an individual CEC. We therefore compared available maximum concentrations in freshwater, estuarine, and marine surface waters of the state compiled for this synthesis (see Appendix A for concentrations in freshwater, estuarine, and marine systems) to the EcoPanel's published MTLs to assess the need for additional risk screening. We also compared maximum contaminant concentrations for sediment and fish and bivalve tissue to appropriate MTLs for nonylphenol (NP), PBDE 047, PBDE 099, and perfluorooctane sulfonate (PFOS) (see Appendix B and C), relevant to the scope of this report.

In freshwater systems, we calculated the highest MTQs (>1,000) for the pesticides bifenthrin and permethrin, based on maximum water concentrations of 19 and 10 µg/L, respectively; chlorpyrifos was not reviewed due to limited remaining registered uses in the state. We calculated MTQs between one and ten for the compounds bisphenol A (BPA; maximum concentration 0.14 µg/L), fipronil (maximum concentration 0.38 µg/L), ibuprofen (maximum concentration 0.71 µg/L), and triclosan (maximum concentration 0.27 µg/L).

Available California data on galaxolide and 17β-estradiol in freshwater settings include maximum concentrations of 0.1 and 0.0012 µg/L, respectively, below available MTLs. This pre-screening step suggests limited need to address these compounds via additional risk screening and/or monitoring and management. For purposes of this report, we proceeded with risk screening for these two compounds to demonstrate the overall utility of the tiered risk-based framework. The current EcoPanel is in the process of revising existing MTLs; updated values could change the outcome of this pre-screening.

In estuarine systems, we calculated high MTQs (>100) for bifenthrin and permethrin based on maximum concentrations in water of 0.16 and 0.015 µg/L, and in sediment of 440 and 130 µg/kg dw, respectively. We also calculated high MTQs (>100) for PBDE 047 and PBDE 099 based on maximum concentrations in sediment of 88 and 34 µg/kg dw. We calculated MTQs between one and ten for BPA and fipronil based on concentrations in estuarine water of 0.05 and 0.012 µg/L, respectively. The maximum concentration of fipronil in estuarine sediment, 0.28 µg/kg dw, is not greater than the MTL. As with freshwater, available California data on galaxolide in estuarine settings is below the MTL, with a maximum concentration of 0.03 µg/L. Data for 17β-estradiol are not available in estuarine water.

In marine systems, MTQs for PBDE 047 and 099 for sediment exceed one, based on maximum concentrations of 2.6 and 3.5, respectively. Marine sediment data are not available for nonylphenol (NP).

Tissue MTQs are difficult to calculate using the present occurrence database because MTLs are provided on a dry weight basis, while relevant fish tissue data are generally reported on a wet weight basis. Nevertheless, maximum wet weight concentrations exceed the MTLs for PBDE 047 and PFOS.

As indicated above, a need for the tiered risk screening described in this report is indicated by MTQs greater than one for most of the CECs originally identified by the EcoPanel. However, for the majority of the contaminants reviewed in this report, MTLs are not yet available. Risk screening using the tiered risk-based framework we describe can proceed without the pre-screening step that requires contaminant MTLs. New and/or updated MTLs for many of the CECs for which occurrence data are available are forthcoming as part of the activities of the currently convened EcoPanel, via a grant that is co-funded by the State Water Board and the California Ocean Protection Council. Additionally, the State Water Board is taking steps to internalize the process of deriving and managing MTLs and plans to use MTLs to inform monitoring and management actions for the CEC Program.

In the future, MTLs should be derived using the best available toxicity thresholds, such as those compiled within this report, based on further evaluation and application of safety factors as follows:

- Compile available thresholds and metadata for each toxicity value with additional supporting information about the reported value including:
    - Indicate if the threshold was generated using multiple species and a species distribution curve, or if it is based on a no observed effect concentration (NOEC) for a single species, or other type of measurement.
    - Document what effect was evaluated such as reproduction, growth, survival, or development.
    - Determine if the study used acute or chronic test methods and if the study included measuring the chemical of interest in the water that the toxicity testing was conducted or if it was predicted based on the study approach.
    - Document the testing species, and if the study included marine species.
    - Document if a safety factor was applied and why; use that information to back calculate the original toxicity study value to remove the safety factor and reapply safety factors in a standardized way as described in the following steps.
- Identify most sensitive species (e.g., algae, invertebrate, fish)
- Apply safety factors as necessary
    - Acute to chronic conversion (10x)
    - Freshwater to saltwater conversion (10x)
    - Endocrine disrupting compounds (EDC) or unknown mode of action (10x)
    - No safety factor used if already incorporated in PNEC
- Freshwater flow chart
    - Acute L(E)C50 ÷ 10 (acute to chronic UF) ÷ 10 (EDC UF)
    - Chronic NOEC (growth, survival, reproduction) ÷ 10 (EDC UF)

    - PNEC (SSD -based on growth/survival/reproduction) ÷ 10 (EDC UF)
- Saltwater flow chart
    - Acute L(E)C50 (SW) ÷ 10 (acute to chronic UF) ÷ 10 (EDC UF)
    - Acute L(E)C50 (FW) ÷ 10 (acute to chronic UF) ÷ 10 (FW-SW UF) ÷ 10 (EDC UF)
    - Chronic NOEC (SW) ÷ 10 (EDC UF)
    - Chronic NOEC (FW) (growth, survival, reproduction) ÷ 10 (FW-SW UF) ÷ (EDC UF)
    - PNEC (SW) ÷ 10 (EDC UF)
    - PNEC (FW) ÷ 10 (FW-SW UF) ÷ 10 (EDC UF)

This additional analysis is not yet complete, and is anticipated to be completed in a report from the EcoPanel in 2022.

The goals of this report complement the current work of the EcoPanel by demonstrating the next step in risk screening for prioritized classes of contaminants. Important refinements that we illustrate include the use of a tiered risk-based framework that provides greater ability to sort and prioritize CECs by Low, Moderate, High, and Very High Concern, as well as the use of a class-based approach that begins to address concerns associated with chemical-by-chemical monitoring and management. Further description of our goals and objectives are provided above, and our methods and the tiered risk-based framework are described below.

2.3. Compilation of Datasets

We gathered California occurrence data recorded in public databases and published peer-reviewed literature. We then compiled available toxicity thresholds and selected our preferred toxicity threshold to utilize in our risk evaluation. Each of these steps are further described in the following sections.

2.3.1. Occurrence data from databases

To compile chemical specific data from various sources, we first compiled a more comprehensive class-based CEC list that could be used to identify and classify individual compounds within each of the prioritized classes. This “compiled CEC list” was assembled using many resources, including 1) public databases of environmental data (Table 2.2); 2) analyte lists from standardized or commercially available analytical methods; 3) presence within class-based lists compiled by other agencies, such as the California Biomonitoring list of designated chemicals (https://biomonitoring.ca.gov/chemicals/designated-chemicals); and 4) recent reviews of CEC classes published in scientific journals.

We included various chemical names and identifiers (e.g., compound names and synonyms, acronyms, Chemical Abstracts Services [CAS] numbers, PubChem ID numbers) that could be used to identify chemical data present in various databases. For some CECs with multiple isomers (i.e., the same chemical formula but a somewhat different arrangement of atoms and bonds), we included individual isomers as well as summed isomers. Compounds were cross-listed in multiple classes where appropriate. For some compounds, particularly members of classes including bisphenols, organophosphate esters, and phthalates, we included readily identifiable metabolites as well. At the request of the State Water Board, cyanotoxins were included in the list, but were not evaluated in this report.

In the case of pesticides, we distinguished urban-use pesticides from those solely used in agricultural applications by first using the California Department of Pesticide Regulation (CDPR) Pesticide Use Reporting (PUR) database to filter the overall active ingredient list based on applications in urban settings. Additional pesticides identified by Water Boards staff were scrutinized by reviewing currently registered products in the CDPR's California Product/Label Database Application (CDPR, 2022) to ascertain whether use in urban regions could be anticipated. We identified pesticide degradates using web searches of the parent compounds, leading to either articles or databases listing likely degradates for the given parent compound.

We used the compiled CEC list to mine monitoring records of these chemicals in the databases containing California data listed in Table 2.2. The occurrence data we assembled form the core of the risk screening for California's ambient aquatic ecosystems. We refer to this compilation of occurrence data from databases as the occurrence database for this project.

Table 2.2. Sources of occurrence data compiled in the occurrence database.

Database Name	Organization
CEDEN (http://www.ceden.org/)	SWRCB
Water Quality Portal (https://www.waterqualitydata.us/)	NWQMC (EPA WQX, USGS NWIS)
SDWIS, California EDT (https://www.waterboards.ca.gov/drinking_water/certlic/drinkingwater/publicwatersystems.html)	SWRCB
CIWQS (https://www.waterboards.ca.gov/water_issues/programs/ciwqs/)	SWRCB
GeoTracker - PFAS Only (https://geotracker.waterboards.ca.gov/map/pfas_map)	SWRCB

Information that was mined from each database was defined in the approved Workplan (Aquatic Science Center, 2020), listed below:

- Project
- Sample Location
- Sample Date
- Sample Type (field, blank, duplicate, etc)
- Parameter (chemical name)
- Result (or non-detect)
- Units (original units are stored in database and a separate field was added to convert Results to a standard unit)
- MDL/RL (method detection limit / reporting limit)
- Sample Matrix (may be inferred from data source)

State Water Board staff are developing a data management plan that should fully describe the methods used to compile the occurrence database. We provide a brief description of the methods here.

We developed code to download data from the databases in Table 2.2. CEDEN data were mined using a SQL server datamart and filtered on SQL server based on the compiled CEC list, then pulled into R using the RODBC library. CIWQS data were provided in .CSV format by the State Water Board based on the compiled CEC list that ASC provided. EDT data were assembled from the dbf files on the EDT library website. STORET/NWIS was pulled from the NWQMC Water Quality Portal using the dataRetrieval package in R, filtered using the compound list and a statecode value of "CA". GeoTracker data was downloaded from the State Water Board's PFAS Map site in CSV format. Additional metadata were compiled including quality assurance data (blanks, spikes, reference materials).

Upon compilation, the database was provided to the State Water Board as a draft working product to inform the parallel activities of the EcoPanel. State Water Board staff conducted additional coding and data processing, provided additional geospatial metadata, and removed records not suitable for use (e.g., units not reported in concentrations, passive sampler results, quality assurance samples). The State Water Board data cleanup was completed to support the EcoPanel's separate evaluation process in accordance with EcoPanel data criteria. In order to ensure data used in this report were consistent with data used by the EcoPanel, the State Water Board provided the revised occurrence database to ASC.

An important element of the State Water Board revision was the addition of information about sample collection sites that can be used to classify records into freshwater, estuarine, and marine ecosystems so that risk evaluation could be conducted separately for each of these types of ecosystems. While records for surface water, sediment, tissue, effluent, and stormwater were compiled in the occurrence database, we focused our report on the surface water records in freshwater, estuarine, and marine settings. We also incorporated evaluation of sediment, fish, and bivalve tissue records for evaluation for a subset of the chemical classes as a model for multi-matrix risk screening.

Data from the occurrence database were summarized for each class and compound as shown in tables in Appendix A (surface water), B (sediment), and C (biota). For each contaminant, reporting is summarized by matrix (water, sediment, fish or bivalve tissue collected from freshwater, estuarine, and marine environments), fraction (for water, dissolved phase only [filtered] versus total water [unfiltered]), and basis (dry weight [dw] versus wet weight [ww]). The 90th percentile concentrations are provided for risk screening purposes.

Because a significant portion of the monitoring data for some CECs are non-detects, and method detection or reporting limits for a given compound can span several orders of magnitude if reported at all, we substituted non-detects with a single median detection limit specific to each compound, matrix, fraction, and basis to calculate 90th percentiles. Median detection limit is the inclusive 50th percentile value from all reported detection limits for that combination of compound, fraction, matrix, and basis that were present in the occurrence database. The 90th percentile value is inclusive of existing values, rather than an interpolated number. This method for handling non-detects was developed in consultation with the EcoPanel.

2.3.2. Occurrence data from peer-reviewed literature

We also conducted a literature review of relevant California occurrence data in published peer-reviewed journals to supplement data from the occurrence database for the risk screening. We compiled relevant CEC data for surface water, sediment, and biota that are relevant to human consumption. Only data from samples collected within the last 15 years were included. Relevant studies were identified by querying a variety of scientific literature databases and search engines such as Web of Science, ScienceDirect, Scopus, PubMed, and Google Scholar. A variety of search terms were used to locate relevant literature including, but not limited to: contaminants of emerging concern or CECs, specific contaminant classes, California (state, counties, and cities), water bodies (e.g., rivers, estuaries, ocean), and specific matrices (e.g., water, sediment). Within our summaries of occurrence data for each class (Section 3), we included maximum concentrations for studies in the literature that were higher than those reported in the occurrence database, or took place in settings not covered by the database, and therefore could be important for risk screening. For risk screening purposes, we compared these notable maximum concentrations to preferred thresholds, where they were significantly different from the 90th percentile concentrations from the occurrence database. Such comparisons are documented in the risk screening subsections of Section 3.

2.3.3. Summary of relevant risk thresholds

Ecological risks are the primary driver of this screening level evaluation, with additional consideration of human health risks relating to consumption of sport fish and shellfish. Therefore, we compiled the best available published ecological risk thresholds to evaluate occurrence data in water and, for some classes, sediment as described further below. We also compiled human health fish consumption thresholds when available to evaluate occurrence data in sportfish and shellfish. Human health risks related to drinking water were not a part of this evaluation; treated drinking water data were not compiled or screened for risks.

Interpretation of ecotoxicity data often requires understanding additional study context (species tested, toxic endpoint, exposure route, exposure length, etc.). There are a variety of different types of ecotoxicity thresholds, from benchmarks established by government bodies to predicted no effect concentrations (PNECs) for all aquatic organisms in a specific ecosystem, to concentrations derived from a study of a specific effect to a single test species (e.g., ECx, NOEC, LOEC). Some of these ecotoxicity thresholds are more protective than others. The number of toxicity tests that have been performed, and therefore what data exist, varies from compound to compound, especially for CECs.

We used only ecosystem-level thresholds derived using either a probabilistic approach and species sensitivity distributions or a deterministic approach applying assessment factors to single species thresholds to account for interspecies differences; single endpoint thresholds from single species toxicity tests (e.g., ECx, NOEC, LOEC) without assessment factors were not used for the risk screening evaluation. Thresholds were compiled from a variety of sources, including ECHA REACH registration documentation (see Appendix D), the NORMAN Ecotoxicology Database (https://www.norman-network.com/nds/ecotox/), Environment and Climate Change Canada's water quality guidelines, the US Environmental Protection Agency (USEPA) Office of Pesticide Programs' aquatic life benchmarks for pesticides, Water Boards' water quality criteria for pesticides, and primary literature review. In cases where multiple thresholds were available for a compound, the level of uncertainty in the threshold, transparency in how the threshold was derived, and best professional judgment were used to choose the threshold for risk screening. We gave preference to thresholds derived from data from multiple species (and representing multiple trophic levels), chronic exposures (rather than acute), and predominantly experimental data (rather than predicted). We also preferentially used thresholds that have transparent derivation and have been incorporated or adopted in state, federal, or international regulation. All else being equal, we chose the lower threshold to be more protective. While the thresholds we selected are the most robust currently available, future implementation of this type of risk screening process should use newer thresholds if they are found to be more appropriate based on toxicological criteria. Calculations of new thresholds (e.g., deriving a new PNEC from available single species data) were outside the project scope.

The ecological risk thresholds and human consumption thresholds were used to evaluate occurrence data in the occurrence database and further supplemented with data from published scientific journals, as described in the next section.

2.4. Description of Tiered Risk-based Framework

Each class of CECs was evaluated using a tiered risk-based framework. As noted previously, the EcoPanel recommended use of a tiered risk-based monitoring approach as part of a broader strategy for monitoring CECs in California waters (Anderson et al., 2012). The Regional Monitoring Program for Water Quality in San Francisco Bay (Bay RMP) adapted this framework and has used it successfully for almost a decade (Sutton et al., 2017; Lin, Sutton, Shimabuku et al., 2018; Miller et al., 2020).

The framework we used for this prioritization differed slightly from the Bay RMP framework in that placement within each tier was based solely on comparison of available occurrence data and toxicological thresholds. Specifically, classification within the present framework does not account for additional factors such as persistence in the environment, cumulative impacts, or changing use or occurrence patterns, which are incorporated into the Bay RMP framework. In addition, for the present framework, placement in a specific risk tier is not linked to a specific set of recommended monitoring or management responses. Instead, the risk tier and additional factors are considered in establishing monitoring and management recommendations for each class of CECs (Section 5).

We used a class-based approach, in which individual chemicals are grouped as members of broader classes defined by chemical similarities (e.g., phthalates, a chemical class), or function or use in society (e.g., pharmaceuticals, a use class). In some cases, a class might be defined by both chemistry and use (e.g., brominated flame retardants). Members of a chemical class often have similar properties with respect to persistence, bioaccumulation, or toxicity. Evaluating chemical classes within the tiered risk-based framework, rather than individual compounds, provides a means of evaluating and addressing data-poor compounds. As noted previously, members of a use class might be addressed by a single management action that affects product requirements (e.g., flammability standards that impact use of flame retardants) or disposal practices (e.g., pharmaceutical take-back programs). Substances within a specific chemical or use class may be substituted for one another within products or formulations; in this case, a class-based approach can indicate which compounds may see increasing use as substitutes and, therefore, merit future monitoring.

Classes of CECs were placed in one of five tiers (see below) based on a comparison of occurrence data to available toxicity information, where sufficient occurrence data were available (discussed in the definition of the Possible Concern tier). Ecological risks are the primary driver of this screening level evaluation, with additional consideration of human health risks relating to consumption of sport fish and shellfish. For risk screening to be protective while limiting the influence of outliers, we used the 90th percentile concentration as calculated using the median detection limit for non-detections (rather than assuming non-detection meant zero concentration; Section 2.3.1).

Ecological risk quotients (RQs) were calculated for each compound in each matrix (e.g., freshwater, estuarine water) as the ratio of each compound's 90th percentile concentration to its appropriate ecological risk threshold. For the estuarine matrix, we used both freshwater and marine thresholds (where available) because estuarine-specific thresholds are generally not available and salinity varies throughout estuarine systems. Human health risk quotients were calculated for biota occurrence data, for which we used the lowest available human health fish tissue consumption threshold.

RQs were used to place individual compounds in Low, Moderate, High, and Very High risk tiers (described below). In cases where there were insufficient data to calculate a suitable risk quotient (e.g., insufficient monitoring data; 90th percentile concentrations based on median detection limits, indicating insufficient method sensitivity; or toxicity thresholds not available), we placed compounds in the Possible Concern tier.

For chemical classes (alkylphenols and alkylphenol ethoxylates; bisphenols; organophosphate esters; phthalates; PFAS), we assigned class-wide tiers based on the highest individual compound risk within the class. For brominated flame retardants, we established separate risk tiers for PBDEs and alternative brominated compounds. For use classes (current-use pesticides [and degradates] with urban applications; pharmaceuticals; personal care and cleaning product ingredients), a class-wide tier is not recommended due to the chemical and toxicological diversity of individual compounds.

It is important to recognize there is uncertainty in the high-level risk screening approach we applied here; there may be uncertainty in our understanding of the level of occurrence of these compound classes in California ecosystems (e.g., available data may not adequately represent statewide conditions) as well as uncertainty in toxicity thresholds (including lack of thresholds). Nevertheless, it is important to have a relatively high level of confidence particularly when placing compounds in the High or Very High Concern risk tiers for the purposes of this report, to inform the Water Boards priorities for managing and monitoring CECs. Therefore, when there was uncertainty in placing a chemical class in a risk tier, we placed these chemical classes in Possible Concern. Alkylphenol and alkyphenol ethoxylates are an example of a compound class where there was uncertainty in the level of occurrence in California waters, so we placed this class in the Possible Concern risk tier (See Section 3.1.3).

Possible Concern – Lack of robust toxicity thresholds, limited occurrence data, and/or insufficient analytical method sensitivity suggests uncertainty in the level of effect on California wildlife. Occurrence data for a compound in a matrix were considered limited when fewer than 10 samples were available in any ecosystem type (freshwater, estuarine, marine), or where data were available in only a single geographic location in the state (e.g., San Francisco Bay). Insufficient method sensitivity is indicated when detection limits of analytical methods for a contaminant are higher than toxicity thresholds, and result in low detection frequencies, as well as 90th percentile concentrations equivalent to median detection limits.

Very High Concern – Occurrence data suggest a very high probability of an adverse impact on California wildlife, with sufficient data indicating occurrence concentrations two or more orders of magnitude above ecological risk thresholds (RQ of 100 or greater) or above available human health thresholds for fish consumption (RQ of one or greater).

High Concern – Occurrence data suggest a high probability of an adverse impact on California wildlife, with sufficient data indicating occurrence concentrations between one and two orders of magnitude above ecological risk thresholds (RQ between 10 and 100) and below available human health thresholds for fish consumption (RQ less than one).

Moderate Concern – Occurrence data suggest a moderate probability of an adverse impact on California wildlife, with sufficient data indicating occurrence concentrations above ecological risk thresholds, but less than one order of magnitude greater (RQ between one and 10) and below available human health thresholds for fish consumption (RQ less than one).

Low Concern – Occurrence data suggest minimal impact on California wildlife, with sufficient data indicating occurrence concentrations below ecological and human health risk thresholds (RQs less than one).

For data-poor compounds that lack suitable toxicity thresholds but are structurally similar to a compound with well understood toxicity, read-across methods can be used. Read-across is an approach where available data of a data-rich substance are used to predict behavior or toxicity of a chemically similar data-poor substance. Read-across methods are already commonly used in risk assessment. We elected to use read-across methods for our risk screening of bisphenols (Section 3.2.3). Calculating toxicity thresholds for individual CECs without available literature thresholds is outside the scope of this project.

3. Occurrence and Risk Screening of Emerging Contaminants in Aquatic Ecosystems of California

With this section, we demonstrate an approach for evaluating risks of CECs using a tiered risk-based framework and occurrence data compiled from around the state. This risk screening approach can directly inform monitoring and management priorities. For each contaminant class, we highlighted compounds of interest and provided a brief description of properties relevant to sources (i.e., the original products or uses from which contaminants are released, such as consumer products) and environmental fate; summarized the available occurrence data in ambient aquatic matrices within California; and conducted a risk screening evaluation that resulted in classifications of Very High, High, Moderate, Low, or Possible Concern, based on a comparison of observed concentrations relative to the best available toxicity thresholds.

For chemical classes (alkylphenols and alkylphenol ethoxylates; bisphenols; organophosphate esters; phthalates; brominated flame retardants, including PBDEs; PFAS), we applied the class-based approach to guide our overall risk screening classification for all members of the class, typically driven by the highest level of concern identified for an individual compound. In the case of bisphenols (Section 3.2.3), our risk screening involved the use of read-across methods, where available data of a data-rich substance can be used to predict behavior or toxicity of a chemically similar data-poor substance. By taking this protective approach to risk-based screening, we attempt to avoid underestimating the risks associated with data-poor compounds, a factor that has led to regrettable substitutions by manufacturers in the past.

For use classes (current-use pesticides [and degradates] with urban applications; pharmaceuticals; personal care and cleaning product ingredients), we did not establish a class-wide risk classification, due to the chemical and toxicological diversity of individual compounds within these classes.

This risk screening demonstration is focused primarily on observations in water. However, we extended the approach to the matrices of sediment, as well as fish and bivalve tissue (potentially consumed by humans), for a subset of the classes examined (alkylphenols and alkylphenol ethoxylates; bisphenols; brominated flame retardants, including PBDEs; and PFAS). Future multi-matrix applications of this approach may be indicated for compounds and classes based on an understanding of their sources, pathways, and fate in the environment.

A summary of the California occurrence data compiled from public databases, as well as comparisons to our preferred ecotoxicological thresholds, is provided in tables in Appendix A (surface waters), B (sediment), and C (fish and bivalves).

3.1. Alkylphenols and alkylphenol ethoxylates

3.1.1. Definition, chemical properties, and use

Alkylphenols (APs) and alkylphenol ethoxylates (APEOs) are a broad chemical class. APs consist of linear or branched alkyl chains bonded to a phenol functional group. APs are reacted with ethylene oxide to produce complex mixtures of APEOs, with hydrophobic alkyl chains and hydrophilic ethylene oxide chains. APEOs are a subclass within a larger ethoxylated surfactant class that includes other nonionic surfactants.

There are various nomenclature conventions and acronyms used to describe APEO isomers and homologues, such as NP**n**EO, where NP (nonylphenol) refers to the alkyl chain, and **n** refers to the length of the ethylene oxide side chain. CAS numbers and other naming schemes used to describe APEOs are often inadequate or even inaccurate for describing the chemistry of APEOs, which are used and present in the environment as complex mixtures (USEPA, 2010a).

Structural differences in AP and APEO isomers and homologues affect the partitioning, toxicity, and persistence of these compounds (Guenther et al., 2005). APEOs with a greater number of ethoxylate units tend to be more hydrophilic, while APs and short-chain APEOs (defined as one to two ethoxylate units) tend to be more hydrophobic, persistent, and toxic, with increased partitioning to suspended solids and the sediment.

APEOs are used in a wide range of cleaning products, pesticide formulations, textile and paper processing, paints and coatings, plastics, tire rubber, among many other industrial and consumer applications. They are consumed on the order of a hundred million pounds per year (EPA, 2010). Nonylphenol ethoxylates (NPEOs) are by far the most widely used APEOs, representing approximately 80% of the total production of APEOs. Technical mixtures of NPEOs used commercially represent complex mixtures of over 100 isomers and homologues due to variation in the branching and lengths of the alkylphenol chain, the length of the ethoxylate side chain, and the substitution position on the phenol ring (Guenther et al., 2005; Lu & Gan, 2014). Octylphenol ethoxylates (OPEOs) are the next most common APEOs in use.

Treated wastewater has been recognized as an important pathway for these compounds to enter the environment (Ahel, 1994; Bergé et al., 2012). Stormwater runoff is another pathway that has previously not been well studied, but is receiving increased attention (Tian et al., 2020). Most investigations of APEO occurrence have focused on NPEOs, particularly short-chain NPEOs, which have been reported in wastewater effluent, wastewater sludge and biosolids, landfill leachate, air, and stormwater; in the water and sediment of ambient freshwater, estuarine, and ocean settings; and in aquatic biota (Bergé et al., 2012; S. L. Klosterhaus et al., 2013; Soares et al., 2008; Tian et al., 2020). NPEOs can biodegrade under aerobic and anaerobic conditions in wastewater treatment plants, ambient waters, and in sludge to nonylphenol carboxylates and short-chain NPEOs, and ultimately to NP (Chiu et al., 2010; Soares et al., 2008; Ying et al., 2002).

A major challenge for environmental monitoring is the availability of analytical methods sensitive enough to detect these compounds in water at ecologically relevant levels. Additionally, commercial methods are not available to quantify other APEOs besides NP, OP, and short-chain NPEOs and OPEOs. Methods are similarly limited for related compounds such as alkylphenol carboxylates (important degradation products of APEOs) and alcohol ethoxylates (important replacement chemicals). Therefore, current monitoring methods are insufficient to fully assess the occurrence and risk from this broad chemical class.

Increasing environmental concern and regulatory review has led to voluntary phase-out of NPEOs in consumer laundry detergents, and DTSC has proposed listing NPEOs in all laundry detergents as a Priority Product in California due to concern about impacts to aquatic organisms (DTSC, 2018). If listed, detergent manufacturers would be required to identify and evaluate alternatives to use of NPEOs that would reduce adverse impacts. The California Air Resources Board also banned APEOs in a variety of cleaning products in 2012. The Minnesota Pollution Control Agency has drafted water quality standards for NP and short-chain NPEOs (MPCA, 2010). The USEPA previously released a Chemical Action Plan for NP/NPEOs that included identification of safer alternatives (EPA, 2010). In Europe, NP/NPEOs are listed as Substances of Very High Concern (ECHA, 2013), and restrictions are in place for most products and uses.

3.1.2. Occurrence summary in California water, sediment, fish, and bivalves

The occurrence database for California contains occurrence data on NP, short-chain nonylphenol ethoxylates (i.e., NP1EO and NP2EO), octylphenol, and short-chain octylphenol ethoxylates in water (Appendix Table A.1). The highest number of records are associated with freshwater monitoring efforts focused on NP and short-chain NPEOs, followed by estuarine monitoring of NPs. The lowest number of records are associated with marine monitoring efforts. Nonylphenol is moderately hydrophobic (log K_{ow} = 4.48) (Soares et al., 2008) and, therefore, tends to adsorb to suspended particles and sediment. Most of the water data was reported in the total water fraction, while fewer data points were available for the dissolved water fraction.

Unfortunately, summarized data in water generally indicate that methods used for most studies are not sufficiently sensitive, resulting in low to zero detection frequencies and estimated 90th percentile concentrations driven by median detection limits. The 90th percentile concentrations for all reported analytes in freshwater, estuarine, and marine systems are the median detection limits, and in the low µg/L range. A maximum concentration of 8.7 µg/L NP and 3.8 µg/L of NPnEO total water fraction have been reported in freshwater.

Key studies in the literature provide important additional occurrence data for California. Specifically, monitoring of various rivers in the Central Valley revealed concentrations of NP, NP1EO, NP2EO, and OP of up to 0.19 µg/L, 0.04 µg/L, 0.24 µg/L, and 0.048 µg/L, respectively (Lavado et al., 2009). Nonylphenol was detected at up to 0.42 µg/L in Morro Bay, an estuary in Central California (Diehl et al., 2012).

Sediment concentrations for NP/NPEOs and OP/OPEOs also indicate methods used for most studies are not sufficiently sensitive for these compounds in sediment (Appendix Table B.1). The estimated 90th percentile concentration for NP in estuarine sediment is based on the median detention limit of 750 µg/kg dw (maximum 200 µg/kg dw), with a detection frequency of 31%. Similarly, the estimated 90th percentile concentrations for NP1EO, NP2EO, OP, and OPnEO are based on high detection limits between 100-1,000 µg/kg dw. There are only four data points available for these compounds in freshwater sediment. The estimated 90th percentile concentrations are driven by the median detection limits, ranging from 35-700 µg/kg dw. Marine sediment data are not present in the database.

Published literature values provide supplemental occurrence data on presence of NP and short-chain NPEOs in marine sediment. Nonylphenol was detected in all five surface sediment samples (maximum 380 µg/kg dw) collected in 2006 near marine wastewater outfalls in southern California marine coastal embayments (K. A. Maruya et al., 2012). Sediment samples from Morro Bay, an estuary with more limited urban influence, contained NP at up to 53 µg/kg dw (Diehl et al., 2012). Another study that collected marine sediment in southern California coastal embayments in 2013 illustrates the challenge of analyzing APEOs in the environment due to possible blank contamination. In this study, blank contamination resulted in elevated detection limits (60 µg/kg dw), and NP were not detected above reporting limits (K. A. Maruya et al., 2016).

Tissue data summarized in the database are limited to two data points from bivalves from an estuary (Appendix Table C.1). Nonylphenol, NP1EO, and NP2EO were detected at up to 94, 41, and 190 µg/kg ww. A time series analysis of archived sediment and bivalve tissue samples from four locations along the California coast found NP and short-chain NPEOs to be ubiquitous and abundant, and also found decreasing concentration trends since the 1970s (K. A. Maruya et al., 2015).

3.1.3. Risk screening

Alkylphenols and APEOs have been shown to impair growth, reproduction, and survival in aquatic organisms, with toxicity increasing with decreasing ethoxylate unit chain length. These compounds have various modes of toxic action, including narcosis, endocrine-mediated pathways (mainly estrogen agonist activity, although there is also some evidence for additional androgen-mediated pathway effects), and when at high enough concentrations for micelles to form, physical surfactant effects (DTSC, 2018). The compounds are additive or synergistic when present in mixtures, and toxicity is therefore sometimes assessed cumulatively (Teneyck & Markee, 2007). For example, Canada uses a toxic equivalency (TEQ) approach to assess nonylphenol and its ethoxylates' combined effects, with their potency expressed relative to nonylphenol because it is the most toxic (CCME, 2002). Canada also recommends including nonylphenol ethoxycarboxylates (NPECs), OP, OPEOs, and octylphenol ethoxycarboxylates (OPECs) in the calculation of total nonylphenol equivalents, as these compounds are very similar in structure and behavior to nonylphenol and its ethoxylates and likely act through a similar mode of action. Studies have also shown the potential for synergistic effect of APs and APEOs in combination with other contaminants such as natural and synthetic estrogens and other endocrine-disrupting compounds such as pesticides that could elicit estrogenic effects on fish at environmentally relevant concentrations (Schlenk et al., 2012).

Several thresholds exist for NP and are in close agreement. The European Chemicals Agency (ECHA) has developed predicted no effect concentrations (PNECs) using species sensitivity analyses for NP (mixed isomers) of 0.61 µg/L for freshwater and 0.57 µg/L in marine water, which we used for this risk screening (see Appendix D). Wang et al. (2018) calculated a similar PNEC of 0.72 µg/L using a species sensitivity-weighted distribution with traditional and non-traditional (e.g., responses at the cellular biomarker and genome levels) endpoints (Y. Wang et al., 2018). The ECHA PNECs developed for 4-n-octylphenol (European Communities, 2002) and 4-tert-octylphenol (see Appendix D) in surface water are similar to those of NP: 0.1 µg/L and 0.632 µg/L, respectively. Canada's water quality guidelines for the weighted sum of NP, NPEs, OP, OPEOs, and OPECs are 1 µg/L for freshwater and 0.7 µg/L for marine water.

Toxicity data for APs and APEOs in sediment are more limited, but PNECs based on chronic invertebrate tests exist for several compounds. The ECHA PNECs for NP are 4620 µg/kg dw for freshwater sediment and 1230 µg/kg dw for marine sediment (see Appendix D). These values are also used by ECHA for 4-tert-octylphenol, which has determined that read-across from NP to OP for toxicity to sediment organisms can be used because the two substances are considered analogue substances with similar toxicities. Similarly, data for biota and human health are minimal. However, a threshold of 8,700 µg/kg ww (8.7 mg/kg ww) exists for NP in seafood (European Commission, 2005).

Comparison of occurrence data with ecotoxicity thresholds indicates that when methods are sufficiently sensitive, NP by itself is generally detected at concentrations near or below its PNECs, with a maximum ecological risk quotient (RQ) for NP in freshwater of 0.6. However, the majority of data are below detection limits, which are near or above ecotoxicity thresholds. Using the TEQ approach consistent with Canada's water quality guidelines, it could be possible for the concentration of the weighted sum of NP, its ethoxylates, and OP and its ethoxylates to exceed guidelines for marine and freshwater. The lack of data on other important ethoxylates and high method detection limits make it difficult to estimate accurate occurrence sum concentrations. Using the maximum values reported in the Central Valley (Lavado et al., 2009) and Canada's water quality guidelines and TEQ approach, the total equivalent concentration RQ including NP, NP1EO, NP2EO, and OP would be 0.38.

Estuarine sediment NP occurrence data were also generated with insufficiently sensitive methods; calculating an ecological RQ using the maximum detected concentration rather than the 90th percentile (which is higher due to a high method detection limit) indicate a maximum RQ of 0.04 or 0.16, depending on whether the freshwater or marine sediment toxicity threshold is used for comparison. The few data points available for NP in bivalves indicate this compound is not a concern for human consumption, with a RQ of 0.01.

Most monitoring results are below detection limits and there are limited data on longer chain ethoxylates, making it difficult to assess risk for this chemical class using the TEQ approach to include all contributions from this broad chemical class. While calculated RQs from quantified concentrations indicate Low Concern, maximum values in freshwater were above ecotoxicity thresholds. Consistent with our approach to indicate a higher level of concern than Low Concern due to uncertainty, as well as limited occurrence data, we classified alkylphenols and alkylphenol ethoxylates as Possible Concern.

3.2. Bisphenols

3.2.1. Definition, chemical properties, and use

Bisphenols are a broad chemical class consisting of compounds containing two phenols connected by a functional group. Bisphenol A (BPA), the most well known in the class, has an alkyl group connecting the two phenols. BPA is also the most well studied in the class, with analogues such as bisphenol S (BPS) and bisphenol F (BPF) receiving more recent attention from the scientific community.

BPA and other bisphenols are consistently dischaged to the environment and have been detected in aquatic matrices around the world (Chen et al., 2016; Corrales et al., 2015; L.-H. Wu et al., 2018). As a class, bisphenols are expected to mimic many of the properties of BPA, including relatively high mobility and moderate water solubility. Some bisphenols are more hydrophobic and have greater likelihood to partition to particles, leading to moderate potential for persistence and bioaccumulation in the environment – though significant amounts can be metabolized (Catenza et al., 2021; Q. Wang et al., 2017). More study is needed to more fully understand persistence and fate of bisphenol analogues beyond BPA (Chen et al., 2016).

Bisphenols are widely produced and used around the globe. Applications are widespread across industrial and consumer goods, due to properties including thermal stability, non-reactivity, and durability. Bisphenols are a primary component in polycarbonate plastics, which are used in a diverse array of products including water pipes, water bottles, toys, medical devices, and electronics. Bisphenols are also used in the production of flame retardants as well as epoxy resin linings that are found in metal-based food and beverage cans. Further notable uses include as thermal reactants in paper products, antioxidants in tire production, and additives in textiles. The vast applications of bisphenols have led to high production volumes worldwide.

Toxicological concerns led to bans of BPA in key products, with several states, including California, and the federal government implementing targeted restrictions since 2009. These bans have predominantly applied to plastic products with food contact including baby bottles, sippy cups, and sports bottles (NCSL, 2017). BPA is currently listed on California’s Proposition 65 List for developmental toxicity and female reproductive toxicity (OEHHA, 2021), including a warning at the point of sale for food contact exposure.

However, at the same time as the first bans were coming into place in the US, manufacturers and industry overall began to substitute BPA with other bisphenols. Production of alternatives such as BPF and BPS has grown significantly and is expected to continue growing (Rochester & Bolden, 2015). BPF and BPS are the most common alternatives used in consumer products, and have been found in personal care and food contact products (Rochester and Bolden, 2015).

3.2.2. Occurrence summary in California water, sediment, fish, and bivalves

In the occurrence database, more than 100 records describe the presence of BPA in surface waters (fresh, estuarine, and marine waters), with a maximum concentration of 0.14 µg/L in freshwater (Appendix Table A.2). Method detection limits vary from 0.0007 to 100 µg/L. BPA has also been monitored in estuarine sediment in California, with a single detection of 71 µg/kg dw reported (Appendix Table B.2). It was not detected in the single freshwater sediment record within the occurrence database (MDL = 100 µg/kg dw). One estuarine study with five samples reported no detected bivalve tissue concentrations (MDL range 460-1450 µg/kg dw; Appendix Table C.2).

Bisphenols other than BPA have limited occurrence data in the database, with just one study of 23 samples in estuarine water (Appendix Table A.2). This study reported no detections of BPAB, BPB, BPC, BPE, BPG, BPM, BPP, BPPH, BPTMC, or BPZ (MDL range 0.0007-0.0014 µg/L). Thirty-two percent of samples had detectable BPS (MDL = 0.001 µg/L; maximum detected concentration of 0.12 µg/L in the dissolved phase). While 73% of these samples also had detectable BPF (MDL = 0.0008 µg/L; maximum detected concentration of 0.15 µg/L), a publication accompanying these data documented field blank sample contamination and, therefore, indicated reservations about the quantification (Shimabuku et al., 2022).

In the literature, several studies included BPA, but were often constrained by high method detection limits, and most reported no detections. Maruya et al. (2016) did not detect BPA in water or sediment samples from the Santa Clara River watershed. Similarly, Sengupta et al. (2014) did not detect BPA in water samples from the Los Angeles and San Gabriel Rivers (MDL = 25 ng/L). Vidal-Dorsch et al. (2012) reported no detections in samples of marine receiving waters.

Some bisphenols have a tendency to adsorb to suspended particles, such that dissolved phase concentrations alone would not fully capture occurrence in surface waters (i.e., total water concentrations). For example, the study of bisphenols in San Francisco Bay indicated the majority of BPA was found in the particulate phase (Shimabuku et al., 2022). While dissolved phase BPA is readily bioavailable, BPA associated with suspended particles is less so, with potential exposure primarily limited to filter feeding organisms and benthic infauna. In contrast to BPA, BPS was concentrated in the dissolved phase, and would be more generally bioavailable to aquatic organisms.

3.2.3. Risk screening

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). As noted above, it is listed on California's Proposition 65 List for developmental toxicity and female reproductive toxicity (OEHHA, 2021).

Though numerous toxicity studies have been conducted for BPA, conflicting data and non-monotonic dose-response curves have led to the generation of multiple predicted no effect concentrations (PNECs). For example, the European Chemicals Agency (Bakker et al., 2016) was criticized for questionable study selection in determination of a BPA marine PNEC of 0.15 µg/L because it excluded a toxicity evaluation of *Marisa cornuarietis*, possibly the most sensitive species (Oehlmann et al., 2008). In contrast, Wright-Walters et al. (2011) used a weight of evidence approach to evaluate 61 BPA toxicity studies involving 24 marine and freshwater organisms to derive an overall aquatic PNEC of 0.06 µg/L. More recent derivation of PNECs using similar 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 (Y. Wang et al., 2018). Consistent with the 2012 EcoPanel recommendations (Anderson et al., 2012), we use the more protective Wright-Walters et al. (2011) BPA PNEC of 0.06 µg/L.

In contrast to BPA, most other bisphenols are poorly understood with respect to potential toxicity (Pelch et al., 2019). 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 & Bolden, 2015; Rosenmai et al., 2014; L. Wang et al., 2018). Thresholds of 26.4 and 54 µg/L have been developed for BPAF and BPF, respectively, using the species-sensitivity distribution approach (L. Wang et al., 2018). Preliminary marine PNECs have also been derived for BPAF (0.52 µg/L), BPC (0.24 µg/L), BPS (27 µg/L), and BP-TMC (0.5 µg/L) as part of REACH registrations (see Appendix D). Comparison of these thresholds with that of BPA might seem to indicate that these compounds are presently a lower concern. However, these PNECs are based on limited freshwater data and, therefore, have inherently high uncertainty, especially when applied to marine and estuarine waters. It is likely that the higher values for these thresholds reflect limited data availability, as opposed to reduced toxicity.

Considering their structural similarities, bisphenols are expected to have overlapping modes of action and likely exhibit additive toxicity (Pelch et al., 2019). As a result, we recommend applying a read-across approach, and using the robust BPA threshold for each of the bisphenols reviewed individually.

For all bisphenols, including BPA, available sediment toxicity testing data is limited. Given the paucity of data, robust sediment thresholds based on sediment toxicity testing have not been developed. Instead, available thresholds are based on an equilibrium-partitioning approach, in which a value to protect organisms exposed to sediment pore water is estimated based on a water threshold (Di Toro et al., 1991), which is more uncertain than using data from toxicity testing of benthic species. Using the K_{oc} for BPA of 708 L/kg and normalizing the value to 1% organic carbon in sediment, Environment and Climate Change Canada developed a Federal Environmental Quality Guideline of 25 µg/kg dw (Environment and Climate Change Canada, 2018). Preliminary PNECs for sediment have been derived for several other bisphenols using the same equilibrium-partitioning approach as part of REACH registrations. As with water thresholds, applying a read-across approach and using the BPA threshold for all bisphenols in sediment is warranted.

Human health thresholds for bisphenols specific to seafood have not been established by authoritative bodies.

Comparison of California occurrence data and ecotoxicity thresholds indicate BPA is of Moderate Concern, with a maximum ecological risk quotient (RQ) of 2.3 for BPA in freshwater (using the maximum detected concentration rather than the 90th percentile, which may be artificially high due to high detection limits) and 4.0 in estuarine sediment. While several studies have looked for BPA in California surface waters, available data suitable for risk screening are limited due to method detection limits that exceed the ecotoxicity threshold of 0.06 µg/L. Water risk may actually be higher for filter feeding organisms, as the available occurrence data indicates some bisphenols are also associated with the particulate phase, and bioavailability may therefore be increased for filter feeding organisms.

The limited occurrence data for other bisphenols in California surface waters result in RQs less than one, but as these data are limited to a single effort studying estuarine water, risk for state waters outside this specific estuary cannot be assessed.

Overall, bisphenols are considered Moderate Concern*,* a classification driven primarily by observations of BPA in freshwater, as well as in estuarine sediment in California.

3.3. Organophosphate esters

3.3.1. Definition, chemical properties, and use

Organophosphate esters may be defined primarily as a chemical class and secondarily via function or use in society. These compounds are organic esters of phosphoric acid, and contain either alkyl chains or aryl groups. They may be halogenated (chlorinated or brominated) or non-halogenated. For purposes of this synthesis and risk screening, organophosphate esters are further defined as synthetic compounds used primarily as flame retardants and plastic additives; naturally derived compounds and synthetic organophosphate esters primarily used as pesticides are excluded.

More commonly monitored organophosphate esters include chlorinated compounds tris(1,3-dichloro-2-propyl) phosphate (TDCIPP), tris(2-chloroethyl) phosphate (TCEP), and tris(2-chloroisopropyl) phosphate (TCPP). Notable non-halogenated organophosphate esters include triphenyl phosphate (TPhP), 2-ethylhexyl diphenyl phosphate (EHDPP), and tris(2-butoxyethyl) phosphate (TBOEP).

There is considerable variation in the chemical properties of individual members of the organophosphate ester class (van der Veen & de Boer, 2012). Several of these compounds are moderately or highly water soluble, and many are semi-volatile (van der Veen and de Boer, 2012). Occurrence in remote settings suggests long-range transport is taking place, despite early assumptions that organophosphate esters might not be sufficiently persistent to build up in the environment (Blum et al., 2019). Studies of biota indicate organisms can readily metabolize some organophosphate esters; nevertheless, bioaccumulation has been observed in a variety of species (Greaves & Letcher, 2017; Pantelaki & Voutsa, 2020).

Organophosphate esters are used as flame retardants and plasticizers in consumer and industrial products including textiles, furniture, electronics, plastics, lubricants, varnishes, and construction materials (Wei et al., 2015). They may also be derived from oxidation of phosphites, commonly used as antioxidants in plastic products (Liu & Mabury, 2018; Venier et al., 2018).

Use of organophosphate esters as flame retardants in foam furniture increased dramatically following a California ban and nationwide phase-out of PBDEs (Cooper et al., 2016; Rodgers et al., 2018; Stapleton et al., 2012). More recently, California banned the use of all flame retardants in foam furniture, mattresses, and children’s products, which is expected to lead to declines in this particular use of the compounds.

3.3.2. Occurrence summary in California waters

The database contains occurrence data on 14 organophosphate esters based on reporting from over a dozen agencies (Appendix Table A.3). Freshwater monitoring efforts have focused on six compounds: TDCIPP, TCEP, TCPP, TPhP, TBOEP, and TnBP (tri-n-butyl phosphate); measurements are reported for the dissolved phase and for total water, depending on the analytical method used by each program. Unfortunately, summarized data indicate the methods used for many studies are not sufficiently sensitive, resulting in estimated 90th percentile concentrations strongly influenced by high median detection limits. For example, the estimated 90th percentile total water concentration for four organophosphate esters (TDCIPP, TCEP, TPhP, TBOEP) in freshwater is 50 µg/L, reflecting the median detection limits for these compounds. Where sufficiently sensitive methods were used, 90th percentile concentrations in both the dissolved phase and total water are generally below 0.2 µg/L, and maximum concentrations are below 0.8 µg/L.

In contrast, estuarine water measurements covered all 14 compounds, and generally employed methods with better sensitivity. Comparison of 90th percentile concentrations for the more plentiful dissolved phase data indicates the top seven compounds present in estuarine waters of California were TCPP (2.9 µg/L) > TBOEP (2.0 µg/L) > TCEP (0.35 µg/L) > TDCIPP (0.25 µg/L) > TPhP (0.15 µg/L) > TnBP (0.11 µg/L) > EHDPP (0.035 µg/L). Marine data in the database are extremely limited, with only a single detection of TBOEP (0.08 µg/L; total water) among analyses conducted by USGS of five organophosphate esters at six sites.

Key studies in the literature provide important additional occurrence data for California. Specifically, monitoring of the Los Angeles, San Gabriel, and Santa Clara Rivers, three effluent-dominated rivers in southern California, during dry weather (low flow) conditions revealed total water concentrations of TCPP, TDCIPP, and TCEP up to 3.3 µg/L, 1.4 µg/L, and 0.81 µg/L, respectively (K. A. Maruya et al., 2016; Sengupta et al., 2014). Vidal-Dorsch et al. (2012) detected a maximum total water concentration of 0.056 µg/L TCPP in marine samples (n=20; detection frequency 10%), and did not detect TCEP (reporting limit 0.05 µg/L).

As noted above, organophosphate esters may be reported in the dissolved phase, or as total water concentrations that can include contributions from suspended particles. Most of the organophosphate esters observed at higher levels in California aquatic ecosystems are found primarily in the dissolved phase, with the exception of EHDPP (Shimabuku et al., 2022; Sutton, Chen, et al., 2019). This suggests that dissolved phase and total water concentrations are likely to be similar for the majority of organophosphate esters described above. Contaminants associated with suspended particles are likely to be less bioavailable overall, with potential exposure primarily limited to filter feeding organisms and benthic infauna.

3.3.3. Risk screening

The toxicological concerns of organophosphate esters have been summarized in recent reviews (Blum et al., 2019; Greaves & Letcher, 2017; Patisaul et al., 2021; van der Veen & de Boer, 2012; X. Wang et al., 2020; Yang et al., 2019). Structurally similar to organophosphate pesticides designed to act on the nervous system, the organophosphate esters used as flame retardants and plastic ingredients are also linked to neurological harm, including developmental neurotoxicity. Many of these compounds are recognized as endocrine-disrupting, especially with respect to thyroid function, and can produce a range of other reproductive and developmental effects. Chlorinated organophosphate esters are considered known or suspected carcinogens. Some organophosphate esters demonstrate toxic impacts to aquatic life, often at environmentally relevant levels (e.g., (C. Wang et al., 2020; Y. Zhu et al., 2015) [TDCIPP]).

Comparison of occurrence data and ecotoxicity thresholds indicates TDCIPP is of Very High Concern in California’s surface waters. The preferred ecotoxicity threshold, 0.00046 µg/L, is based on a species sensitivity distribution developed from toxicity data available from more than eight studies (Xing et al., 2019), and is used in this risk screening for comparison to fresh, estuarine, and marine waters. Focusing specifically on monitoring conducted with methods of sufficient sensitivity to detect TDCIPP either dissolved and total water fraction (i.e., excluding the freshwater total water estimated 90th percentile concentration of 50 µg/L), comparison to both freshwater and estuarine 90th percentile concentrations reported in the database reveals ecological RQs ranging from 370 to 700. The freshwater RQ would be even higher if the maximum concentrations reported in effluent-dominated rivers of southern California were used instead (Sengupta et al., 2014; Maruya et al., 2016). A similar RQ may be calculated for marine settings, but is based on insufficient sample size and method sensitivity, indicating a need for additional monitoring data.

Two other chlorinated organophosphates, TCEP and TCPP, would be of Low Concern if considered individually using this risk screening method. Available ecotoxicity thresholds based on species sensitivity distributions are 14 µg/L for TCEP and 59 µg/L for TCPP (Xing et al., 2019). For monitoring efforts with methods of sufficient sensitivity (detection limits below thresholds), RQs calculated using the occurrence database are less than 0.05. An evaluation of the additional literature studies does not result in a higher level of concern. However, overall data for these contaminants are not as robust, due to the frequent use of methods with insufficient sensitivity, and the limited monitoring of marine waters.

Two non-halogenated, aromatic organophosphate esters, TPhP and EHDPP, would be classified as Moderate Concern individually, particularly for estuarine waters. TPhP has a freshwater aquatic toxicity threshold of 0.74 µg/L, and a marine threshold of 0.074 µg/L, both of which are environmental quality standards in the European Union (Ministry of Environment and Food of Denmark, 2017). Comparison to 90th percentile concentrations results in an estuarine RQ of 2.0 based on the marine threshold. Most freshwater monitoring efforts for this contaminant are of insufficient sensitivity; the few available data of sufficient quality suggest Low Concern. Marine monitoring of TPhP is insufficient in terms of quantity and sensitivity, with uncertainty consistent with a classification of Possible Concern in this setting. Both marine and freshwater data are lacking for EHDPP. Available estuarine monitoring relative to an aquatic toxicity threshold of 0.018 µg/L for freshwater and marine settings (derived by the NORMAN Ecotoxicology Database using the *Daphnia magna* acute LC50) results in the RQ of 1.9, consistent with a Moderate level of concern. It is important to note that this threshold has high uncertainty and these data are for the dissolved phase, while the majority of EHDPP may be associated with suspended particles (Shimabuku et al., 2022; Sutton, Chen, et al., 2019), resulting in a potential additional exposure pathway for filter-feeding organisms.

Occurrence data for an additional organophosphate ester, TBOEP, indicate that this contaminant would approach levels of Moderate Concern in estuarine environments. TBOEP has a freshwater aquatic toxicity threshold of 24 µg/L, and a marine threshold of 2.4 µg/L, derived from the Rainbow trout acute LC50 using assessment factors of 1000 and 10,000, respectively (see Appendix D). The 90th percentile concentration calculated using the occurrence database indicates an RQ of 0.83 relative to the highly uncertain marine toxicity threshold. The majority of freshwater data are of insufficient sensitivity, while marine data are of insufficient quantity, resulting in a setting-specific indication of Possible Concern for these environments.

Most other organophosphate esters would be classified individually as Low Concern, where sufficient occurrence data are available. Levels of TnBP in both freshwater and estuarine environments are well below the available ecotoxicity threshold of 62 µg/L (Xing et al., 2019); limited marine data are consistent with Possible Concern in this setting. For five additional compounds (triethyl phosphate, tripropyl phosphate, tricresyl phosphate, tris(2,3-dibromopropyl) phosphate, bisphenol A bis(diphenyl phosphate)), estuarine occurrence data indicate Low Concern, although for two of these compounds (tripropyl phosphate, tris(2,3-dibromopropyl) phosphate), only freshwater toxicity thresholds are available. No freshwater or marine occurrence data are available for these five compounds. A final organophosphate ester, tris(2-isopropylphenyl) phosphate, has insufficient toxicity information, resulting in a classification of Possible Concern in all settings.

Toxicological information to date supports the use of a class-based risk screening approach (Blum et al., 2019; Greaves & Letcher, 2017; Patisaul et al., 2021; van der Veen & de Boer, 2012; X. Wang et al., 2020; Yang et al., 2019). Co-occurrence of multiple members of this class is generally expected in ambient aquatic environments of California; however, methods for assessing cumulative impacts have not been developed, and significant variation may be anticipated for different specific compounds and modes of action. As a result, we recommend a class-based risk screening approach that focuses on the highest category of concern observed for individual members of the class.

Organophosphate esters are considered Very High Concern, a classification driven primarily by observations of TDCIPP in multiple freshwater and estuarine ecosystems in California.

3.4. Phthalates

3.4.1. Definition, chemical properties, and use

Phthalates are a chemical class consisting of ester derivatives of phthalic acid, with ester groups (and their additional side alkyl chains) restricted to the *ortho* position. Among the class, di-(2-ethylhexyl) phthalate (DEHP) is the most well-studied and regulated due to its vast production, use in a broad range of products, and ubiquity in the environment.

Phthalates share a variety of physical and chemical properties, though some variations occur due to differences in the side alkyl chains present. Phthalates generally have low water solubility and are considered semivolatile. DEHP, which has longer alkyl side chains and thus higher molecular weight, is one of the least soluble, with higher proportional partitioning to particles in the aquatic environment (Gupta, 2011; Roark, 2020).

Phthalates have been observed in aquatic ecosystems and organisms globally (Net et al., 2015). Phthalates are not expected to bioaccumulate or biomagnify, and readily metabolize in biota, especially those at higher trophic levels (Net et al., 2015; J. Zhang et al., 2021).

Phthalates are used in a wide range of applications, predominantly due to their utility as plasticizers to increase the flexibility and longevity of manufactured plastic. High molecular weight phthalates are known for use in construction materials and polyvinyl chloride (PVC) products such as clothing, flooring, food packaging, children's toys, upholstery, and medical devices. Since they are not chemically attached to the plastic polymer, phthalates can readily leach and enter the environment (Roark, 2020). Compounds with low molecular weight are more likely to be found in formulations including personal care products, cleaning supplies, insecticides, paints, and pharmaceuticals (Gao & Wen, 2016; Net et al., 2015).

DEHP and other phthalates, particularly butyl benzyl phthalate (BBzP), dibutyl phthalate (DBP), di-isodecyl phthalate (DIDP), diisononyl phthalate (DINP), and di-n-hexyl phthalate (DnHP), have been a focus for state and federal regulators in the US due to their presence in a variety of consumer products and potential human health concerns (Eichler et al., 2019; OEHHA, 2021). Most notably, the use of several phthalates is restricted in children's toys to levels no greater than 0.1% (CPSC, 2017). Dimethyl phthalate (DMP), diethyl phthalate (DEP), and di-n-octyl phthalate (DnOP) are others that have been prioritized for management actions by the EU and China (Eichler et al., 2019; Gani et al., 2017). Of note, DEHP is listed as a Clean Water Act Priority Pollutant.

3.4.2. Occurrence summary in California waters

The database contains water monitoring data for six phthalates: DMP, DEP, DBP, BBzP, DEHP, and DnOP (Appendix Table A.4). In freshwater, studies with sufficient sensitivity provide 90th percentile concentrations in the order: DEHP (0.57 µg/L) > BBzP (0.18 µg/L) > DBP (0.16 µg/L) > DEP (0.11 µg/L) > DMP (0.05 µg/L) > DnOP (0.01 µg/L).

Far more limited data in estuarine water include detections of a single phthalate, DEP, at a maximum concentration of 0.05 µg/L. More data are available for marine settings, including maximum concentrations of DEHP (20 µg/L) > DBP (0.39 µg/L) > DEP (0.23 µg/L) > BBzP (0.051 µg/L). Vidal-Dorsch et al. (2012) reported an additional maximum concentration of 0.085 µg/L DnOP in marine water.

The scope of this review included water samples only; sediment data were not reviewed. While water data are sufficient to demonstrate the risk screening concept, a more comprehensive assessment that includes sediment for less soluble and more hydrophobic compounds like phthalates should be conducted in the future.

3.4.3. Risk screening

Phthalates are reproductive and developmental toxicants. Exposure to phthalates can lead to activation of cytochromes P450 (CYP450) detoxification enzymes and endocrine system receptors of aquatic organisms, which can lead to oxidative stress, metabolic disorders, endocrine disruption, and immunosuppression (Y. Zhang et al., 2021). The CYP450 detoxification pathway and the endocrine systems can also activate and/or inhibit each other, causing genotoxicity and cell apoptosis, which can block organism-level growth and development. Phthalates are metabolized into phthalate monoesters, which follow the same modes of toxic action in aquatic organisms, although these metabolites tend to exhibit weaker toxicity than their parent compounds.

Exposure to a mixture of phthalates does not consistently result in additive toxicity, because phthalates can act as both agonists and/or antagonists on CYP450 and endocrine systems (Y. Zhang et al., 2021). However, a cumulative approach has been used for assessing phthalate exposure and risk assessment in human health (Christensen et al., 2014).

Occurrence data suggest multiple phthalates are present in California waters at concentrations below individual ecotoxicity thresholds, although method detection limits are not always sufficiently sensitive. Risk quotients for DEHP range from 0.44 to 0.77 in fresh and marine water, based on the fresh and marine water threshold of 1.3 µg/L, which is an environmental quality standard in the European Union (European Union, 2013). Risk quotients for DBP (RQ of 1.5) and DEP (RQ of 1.2) in marine water do indicate exceedance of marine thresholds derived as part of ECHA REACH registrations of 1.0 µg/L and 1.2 µg/L, respectively (see Appendix D); however, the detection frequencies for both contaminants were under 10%, and the 90th percentile concentrations estimated were based on median method detection limits that exceeded thresholds. Two other phthalates (BBzP and DMP) have risk quotients that indicate concentrations do not exceed marine toxicity thresholds (both RQs 0.068). DnOP does not have an available toxicity threshold.

Toxicological information to date supports the use of a class-based risk screening approach (Y. Zhang et al., 2021). Co-occurrence of multiple members of this class is generally expected in ambient aquatic environments of California; however, due to overlapping agonistic and antagonistic effects, an additive approach is not recommended for assessing risk. As a result, we recommend a class-based risk screening approach that focuses on the highest category of concern observed for individual members of the class.

<u>Phthalates are classified as Possible Concern.</u> This classification reflects our screening evaluations that show that while detected water concentrations were below toxicity thresholds, our understanding of occurrence levels in California ecosystems is limited by available monitoring data, particularly in estuarine and marine ecosystems. These additional uncertainties elevate our concern for phthalates from Low Concern indicated by the risk quotients to Possible Concern.

3.5. Brominated flame retardants, including polybrominated diphenyl ethers

3.5.1. Definition, chemical properties, and use

As the name suggests, the brominated flame retardants class consists of organic compounds that contain bromine and are added to products as flame retardants. Thus, the class is defined by a combination of chemical and use properties. The most well-known members of this class are the polybrominated diphenyl ethers (PBDEs), which came into widespread use in the US in the 1970s in response to flammability standards for a variety of consumer products (USEPA, 2010b). Another relatively well-studied brominated flame retardant is hexabromocyclododecane (HBCD). A number of other brominated compounds that are newer to the market or only more recently observed in the environment are included within this class of contaminants, such as bis(2-ethylhexyl) tetrabromophthalate (BEHTBP) and 2-ethylhexyl-2,3,4,5-tetrabromobenzoate (EH-TBB), ingredients in flame retardants such as Firemaster 550 (Covaci et al., 2011).

Members of this broad class often share key chemical properties relevant to fate in the environment. Available studies indicate that many brominated flame retardants are persistent and bioaccumulative (Covaci et al., 2011; Hou et al., 2021; USEPA, 2010b). These compounds are hydrophobic, with a tendency to partition to sediment and lipids, and semi-volatile, which can lead to long-range atmospheric transport (Covaci et al., 2011; Hou et al., 2021; USEPA, 2010b).

A major source of these compounds to the environment is migration from consumer goods in use or in the waste stream (USEPA, 2010b; Zuiderveen et al., 2020). Flame retardants have been incorporated into a broad array of consumer products, primarily due to regulatory or voluntary industry flammability standards. For several decades, three commercial mixtures of PBDEs were among the most widely used flame retardants (USEPA, 2010b). While there are 209 possible PBDE congeners, the composition of these commercial formulations and their relative use has led to a particular focus on congeners such as PBDE 047, 099, and 209 with respect to both environmental monitoring and toxicity investigations. HBCD has 16 possible diastereoisomers, but commercial mixtures are primarily composed of α-HBCD, β-HBCD, and γ-HBCD, which have been the focus of scientific research (Covaci et al., 2006).

State bans and nationwide phase-outs of PBDEs have led manufacturers to shift to alternatives, including other brominated flame retardants, as well organophosphate esters (Section 3.3) and other compounds. Countries that are signatories to the Stockholm Convention on Persistent Organic Pollutants are expected to phase out PBDEs, polybrominated biphenyls (PBBs), and many uses of HBCD (Sharkey et al., 2020); the US has not ratified the Stockholm Convention. In December 2021, the USEPA released a draft revised risk determination for HBCD, which stated that HBCD presents an unreasonable risk of injury to health and the environment via current uses (https://www.epa.gov/system/files/documents/2021-12/9823-01_risk-determination.pdf).

When this risk determination is finalized, the agency can then address any unreasonable risks posed by HBCD, which could include regulations to prohibit or limit the manufacture, processing, distribution in the marketplace, use, or disposal of this substance.

State bans on broader numbers of flame retardants, particularly in categories like furniture, children's products, and bedding, are expected to lead to reduced use (Sharkey et al., 2020). Likewise, review of specific flammability standards has led to revisions that remove the incentive to add these chemicals to products (Cooper et al., 2016). However, many regulatory or voluntary flammability standards that essentially require use of flame retardants remain in place for product categories including electronics, building and construction materials, motor vehicles, and children's car seats (Babrauskas et al., 2012; Charbonnet et al., 2020; Reddam et al., 2020; Y. Wu et al., 2019).

3.5.2. Occurrence summary in California water, sediment, fish, and bivalves

Significant quantities of occurrence data are available for PBDEs in California aquatic ecosystems, including measurements of over 50 congeners in water, sediment, and fish and bivalve tissues. We reviewed occurrence data and screened associated risks for these three matrices. We focused primarily on PBDE 047, PBDE 099, and PBDE 209, which are dominant congeners found in commercial mixtures, and are often observed in the environment. While PBDE 047 and 099 are relatively well-characterized with respect to occurrence, PBDE 209 presents analytical challenges (Kierkegaard et al. 2009) and, therefore, is not always included in monitoring studies.

In freshwater settings, maximum reported PBDE concentrations in water in the database range from PBDE 209 (1.1 µg/L) > PBDE 099 (0.2 µg/L) > PBDE 047 (0.13 µg/L); 90th percentile concentrations derived from the database were PBDE 209 (0.059 µg/L) > PBDE 099 (0.0094 µg/L) > PBDE 047 (0.0070 µg/L) (Appendix Table A.5a). These are total water concentrations, including PBDEs present in the dissolved phase as well as adsorbed to suspended particles. This is important to note, as PBDEs have low water solubilities, with significant proportions adsorbed to particles.

For estuarine settings, the database contains water monitoring data for both total water and the dissolved phase only, depending on the methods employed by different monitoring efforts (Appendix Table A.5a). Maximum dissolved phase concentrations ranged from PBDE 099 (0.022 µg/L) > PBDE 047 (0.0048 µg/L) > PBDE 209 (0.0009 µg/L), with estimated 90th percentile concentrations of PBDE 099 (0.0039 µg/L) > PBDE 047 (0.0030 µg/L) > PBDE 209 (0.0008 µg/L). Maximum and 90th percentile concentrations reported for total water were lower in the database. While we might expect total water concentrations of PBDEs to be significantly greater than dissolved phase concentrations for individual water samples, the occurrence database compiles records from multiple monitoring efforts and a range of locations and dates, leading to this counterintuitive finding. Of note, hydroxylated PBDEs were also observed at low levels (<0.0001 µg/L) in surface water at two sites in the San Francisco Bay (Kerrigan et al. 2015).

Interestingly, available marine concentrations include higher 90th percentile values in total water relative to estuarine water: 0.050 µg/L PBDE 209 > 0.0048 µg/L PBDE 099 > 0.0036 µg/L PBDE 047 (Appendix Table A.5a). Maximum reported PBDE concentrations in marine water in the database range from PBDE 209 (0.069 µg/L) > PBDE 099 (0.012 µg/L) > PBDE 047 (0.01 gµ/L).

In sediment, PBDE 209 was typically the most abundant congener (Appendix Table B.5a). In freshwater sediment, the maximum concentration of PBDE 209 in the database was 540 µg/kg dw, followed by PBDE 099 (150 µg/kg dw) and PBDE 047 (100 µg/kg dw). The 90th percentile concentration of PBDE 209 was 120 µg/kg dw, at least an order of magnitude higher than the 90th percentile concentrations for other congeners. California freshwater sediment concentrations reported in the literature are lower than the 90th percentiles derived from the database (Maruya et al., 2016).

In estuarine sediment, the maximum concentrations reported in the database were 380 µg/kg dw PBDE 209 > 88 µg/kg dw PBDE 047 > 34 µg/kg dw PBDE 099; 90th percentile concentrations were an order of magnitude lower for each of these congeners (Appendix Table B.5a). A study of estuarine embayments in southern California revealed maximum sediment concentrations of 11 and 19 µg/kg dw for PBDE 047 and 099, respectively; PBDE 209 was not evaluated (Maruya et al., 2016). A study of San Francisco Bay sediment samples not included in the database indicated PBDE 209 was the dominant congener, with a maximum concentration of 5.5 µg/kg dw; maximum concentrations for PBDE 047 and 099 were 1.0 and 0.57 µg/kg dw, respectively (S. L. Klosterhaus et al., 2012). A hydroxylated form of PBDE 047 (6-OH-BDE 47) and its brominated dioxin (1,3,7-tribromodibenzo-p-dioxin) photoproduct were observed in surface sediments from San Francisco Bay; lower concentrations were found in sediment from the Point Reyes National Seashore, a nearby marine setting with limited urban influence (Kerrigan et al., 2015).

In marine sediment, the maximum concentrations in the database were 3.5 µg/kg dw PBDE 099 > 2.6 µg/kg dw PBDE 047 > 1.4 µg/kg dw PBDE 209; the 90th percentile concentrations were less than a quarter of these maximum values (Appendix Table B.5a). In the literature, sediment samples collected near ocean outfalls discharging treated wastewater effluent had higher maximum concentrations of 28 and 16 µg/kg ww for PBDE 047 and 099, respectively; PBDE 209 was not included (K. A. Maruya et al., 2012). A study of sediment cores collected near ocean outfalls reported maximum concentrations at depth of 31 µg/kg dw for PBDE 209, 2.0 µg/kg dw for PBDE 047, and 0.39 µg/kg dw for PBDE 099 (J. Li et al., 2019).

In aquatic species, PBDE 047 is typically the dominant congener in tissues (Appendix Table C.5a). Fish tissue data in the database include maximum and 90th percentile concentrations of 96 and 8 µg/kg ww for freshwater; 220 and 20 µg/kg ww for estuarine water; and 14 and 6.6 µg/kg ww for marine water. Additional studies in the literature provide maximum concentrations of PBDE 047 of 230 µg/kg ww for small prey fish in effluent dominated streams in southern California (Maruya et al., 2016); 74 µg/kg ww for sport fish in San Francisco Bay (Klosterhaus et al., 2012); and 11 and 3.8 µg/kg ww for fish collected from the Santa Clara Estuary and Los Angeles Harbor, respectively (Maruya et al., 2016). Maximum concentrations of summed PBDEs in these studies are 370, 124, 13, and 5.3 µg/kg ww, respectively. The most recent study of PBDEs in sport fish, collected from San Francisco Bay in 2019, focused on shiner surfperch, a species with the highest reported average concentrations in this location (Buzby et al., 2021). An examination of temporal trends in this species indicates PBDEs have declined following bans and phase-outs, although the rate of decline has leveled off in recent years. The highest concentrations of PBDE 047 (13 µg/kg ww) and summed PBDEs (23 µg/kg ww) observed were in a largemouth bass collected from Artesian Slough, a location strongly influenced by wastewater discharges.

Bivalve monitoring data in the database are generally reported on a dry weight rather than wet weight basis; maximum and 90th percentile concentrations were 54 and 20 µg/kg dw for freshwater; 54 and 18 µg/kg dw for estuarine water; and 68 and 18 µg/kg dw for marine water (Appendix Table C.5a). In the literature, a study of resident coastal bivalves reported a maximum concentration of PBDE 047 of 14 µg/kg ww (Dodder et al., 2014), while a study of transplanted bivalves deployed for 90 days at different depths in Santa Monica Bay and the Los Angeles Harbor, including sites near wastewater outfalls, observed a maximum concentration of 60 µg/kg dw (Joyce et al., 2015).

The database includes information for several other brominated flame retardants in estuarine water and sediment derived from a study of San Francisco Bay (Sutton, Chen et al., 2019); no data are available in freshwater or marine settings (Appendix Table A.5b). In estuarine water, monitoring for 20 brominated compounds resulted in detections of just three flame retardants in the dissolved phase, BEHTBP, EH-TBB, and tris(2,3-dibromopropyl) phosphate (also listed as an organophosphate ester). Limited detection in water is not surprising, as many of these compounds are hydrophobic and likely to adsorb to particles and settle to sediment. As a result, 14 of these 20 compounds were observed in estuarine sediment, with maximum concentrations for the top five in the order β-1,2-dibromo-4-(1,2-dibromoethyl)cyclohexane (β-DBE-DBCH; formerly abbreviated as TBECH) (1.6 µg/kg dw) > γ-DBE-DBCH (1.1 µg/kg dw) > tris(2,3-dibromopropyl) phosphate (1 µg/kg dw) > hexabromobenzene (HBB) (0.79 µg/kg dw) > BEHTBP (0.48 µg/kg dw) (Appendix Table B.5b). Two of three diastereoisomers of HBCD, α- and γ-HBCD, were commonly detected at lower concentrations in estuarine sediment, while β-HBCD was less frequently observed. An earlier study of sediment in San Francisco Bay provided detections of all three diastereoisomers of HBCD (maximum summed concentration 1.7 µg/kg dw), as well as pentabromoethylbenzene (PBEB) (maximum 0.1 µg/kg dw) and 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE) (maximum 0.06 µg/kg dw); decabromodiphenyl ethane (DBDPE), HBB, EH-TBB, and BEHTBP were not detected in this earlier study (Klosterhaus et al., 2012).

The database contains data on alternative brominated flame retardants in estuarine bivalves derived from a study of transplanted bivalves in San Francisco Bay (Sutton, Chen, et al., 2019); among the nine detected compounds, maximum wet weight concentrations for the top three contaminants were calculated to be α-HBCD (0.16 µg/kg ww) > 2,4,6-tribromophenyl allyl ether (TBP-AE) (0.12 µg/kg ww) > β-DBE-DBCH (0.060 µg/kg ww) (Appendix Table C.5b). The database also indicates no detections of polybrominated biphenyls (PBBs) in marine bivalves analyzed using a method with higher detection limits. Additional resident bivalve data from coastal marine sites include maximum detections of 0.37 µg/kg ww γ-HBCD, 0.22 µg/kg ww α-HBCD, and 0.081 µg/kg ww β-HBCD, along with 0.3 µg/kg ww BTBPE (Dodder et al. 2014). Finally, a study of sport fish in San Francisco Bay reported maximum concentrations of 0.25 µg/kg ww α-HBCD, 0.066 µg/kg ww β-HBCD, 0.065 µg/kg ww γ-HBCD, and 0.23 µg/kg ww HBB (Klosterhaus et al., 2012).

3.5.3. Risk screening

PBDEs are endocrine disruptors that target thyroid hormones and associated metabolic processes (USEPA, 2008a, 2008b). They are reproductive and developmental toxicants (USEPA, 2008a, 2008b; Yu et al., 2015), cause neurodevelopmental impacts (Costa et al., 2014), and may affect immune function in aquatic organisms (Arkoosh et al., 2010; Neale et al., 2005). Significant human health concerns have led to development of fish consumption guidance for the State of California.

On an individual congener basis, a review of risk quotients suggests Moderate Concern for several PBDEs with at least five bromines in surface waters. Ninetieth percentile concentrations of PBDEs with RQs greater than one for marine water include PBDE 085, 099, and 100; in estuarine water, the 90th percentile dissolved phase concentration of PBDE 099 reveals an RQ of one. In freshwater settings, RQs greater than one are calculated for PBDEs 085, 099, and 100, based on Canada's federal environmental quality guideline for fresh and marine waters (Environment and Climate Change Canada, 2013). For some congeners, such as PBDE 003, lack of analytical sensitivity (detection limits greater than toxicity thresholds) results in high estimates of 90th percentile concentrations. Due to uncertainty with respect to occurrence, available data for some individual analytes like PBDE 003 are consistent with the classification of Possible Concern. As a class, and across ecosystems, PBDEs are considered a Moderate Concern in water.

A review of RQs calculated using 90th percentile concentrations for estuarine and marine sediment indicates Moderate Concern is warranted for a greater number of individual congeners with five or more bromines. We used Canada's federal environmental quality guidelines for fresh and marine waters as our thresholds of choice except for PBDEs with nine bromines, for which guidelines have not been derived. These guidelines are benchmarks for aquatic ecosystems that are intended to protect all forms of aquatic life, but are derived using primarily invertebrate data because very few fish data exist (Environment and Climate Change Canada, 2013). For PBDEs with nine bromines, we used the lowest available threshold in the NORMAN ecotoxicological database for total PBDEs. At least a dozen congeners, ranging from PBDE 085 to PBDE 208, have RQs greater than one in estuarine and/or marine sediment. In freshwater sediment, RQs exceeding one were calculated for PBDE 085, 099, 100, and 209. In some cases, maximum concentrations reported in the literature would result in higher RQs, though values do not exceed 100. As with water, insufficient method sensitivity for a handful of congeners results in occurrence data consistent with Possible Concern rather than Moderate Concern.

Risk quotients calculated for individual PBDE congeners are expected to underestimate overall risk, due to occurrence of multiple congeners in environmental matrices, and resulting cumulative exposure and risk to biota. The advisory tissue levels provided by CalEPA's Office of Environmental Health Hazard Assessment (OEHHA) to provide guidance around fish consumption to protect human health are calculated relative to total PBDEs, and specifically mention sums of PBDE 028, 047, 099, 100, 153, 154, 183, 190, and 209 (Klasing & Brodberg, 2011). The lowest advisory tissue level, suitable for consumption of three servings of fish per week, is less than 100 µg/kg ww. Fish tissues with 100-210 µg/kg ww may be safely consumed for two meals per week, while those with 210-630 µg/kg ww may be consumed once per week. Fish with over 630 µg/kg ww total PBDEs should not be consumed.

In the database, 90th percentile concentrations of the dominant congener in bivalves and fish, PBDE 047, were 38 and 20 µg/kg ww, respectively (for estuarine settings; excluding bivalve concentrations reported on a dry weight basis), while in the literature, maximum concentrations of PBDE 047 in California sport fish were 74 µg/kg ww (S. L. Klosterhaus et al., 2012). Although the database was not constructed to provide summed PBDE concentrations for each tissue sample, a protective assumption that PBDE 047 can make up around half the total PBDEs in fish tissue may be appropriate (Hites, 2004). Meanwhile, the California-based literature indicates a maximum concentration of total PBDEs in sport fish of 124 µg/kg ww (calculated from Klosterhaus et al., 2012). A study of small prey fish (convict cichlid, *Amatitlania nigrofasciata*) that are not generally considered for human consumption reported higher maximum values of 300 µg/kg ww total PBDEs (K. A. Maruya et al., 2016).

Comparison of these measurements to advisory tissue levels suggests that some fish may exceed the maximum level of total PBDEs recommended to protect people eating high fish diets. As a result, PBDEs are classified as Very High Concern overall in California aquatic environments.

Among other brominated flame retardants, few toxicity thresholds are available, limiting the conclusions that can be drawn from this risk screening exercise.Thresholds that are available are often derived from limited toxicological testing and predictions based on chemical structure, and therefore have high uncertainty. In estuarine water, the 90th percentile concentration of BEHTBP is approaching the freshwater and marine toxicity threshold of 0.00095 µg/L, based on the predicted acute *Pimephales promelas* LC50 with an assessment factor of 1000 (NORMAN ecotoxicological database). While the RQ of 0.58 suggests a classification of Low Concern, occurrence data are limited in geographic scope, as noted above, and the threshold based on acute predicted data has a high degree of uncertainty, such that an overall classification of Possible Concern for this contaminant is warranted. An RQ for pentabromobenzyl bromide/pentabromotoluene (coeluants) of 1.0 in estuarine water is solely driven by lack of detection using a method with a detection limit equal to the freshwater threshold of 0.0002 µg/L, which is a Dutch environmental quality standard for surface freshwater; as a result, a classification of Possible Concern is appropriate. In sediment, HBCD levels were significantly lower than the available toxicity threshold, indicating Low Concern for San Francisco Bay, the only setting for which California occurrence data are available. Thresholds relating to human consumption of seafood are not available for these alternative brominated flame retardants.

Overall, these alternative brominated flame retardants are classified as Possible Concerns in California, primarily due to limited occurrence data and toxicity thresholds.

3.6. Select current-use pesticides with urban applications, and pesticide degradates

3.6.1. Definition, chemical properties, and use

Used significantly across the globe, pesticides are a diverse set of synthetic, organic, and inorganic compounds used to control pests. Pesticides are a use class; members may be divided into subclasses defined by chemical similarities. All insecticide, herbicide, fungicide, rodenticide, and antimicrobial active ingredients technically fall under the class; for purposes of this report, key antimicrobials have been reviewed as personal care and cleaning product ingredients (see Section 3.8). This report focuses on three groups of pesticides known to be currently used in urban settings of California: imidacloprid, fipronil, and pyrethroids. All are well-known insecticides highlighted for their widespread urban uses and ubiquitous detection in the urban aquatic environment. In the synthesis of occurrence data and risk screening presented below, we have not differentiated urban vs. agricultural uses or settings, as this is beyond the scope of this report.

Current-use pesticides are unlike many of the other classes described in this report because they are subject to registration. Pesticides are regulated at the federal level by the USEPA, and at the state level by the CDPR. These regulatory agencies must approve registration of pesticide products for specific uses prior to first sale and use. The USEPA must approve product labels, which include application instructions and various precautions. In addition, CDPR’s Surface Water Protection Program conducts pesticides monitoring and modeling activities to inform regulation, including risk mitigation. Other California state, regional, and municipal agencies also monitor various pesticides to inform future management strategies and mitigation actions.

Imidacloprid is a member of the neonicotinoids class of insecticides. Neonicotinoids were developed as alternatives to organophosphate and carbamate insecticides, and are currently among the most widely used classes of insecticides globally (Bass et al., 2015; Simon-Delso et al., 2015). Of the many neonicotinoids produced today, imidacloprid was the first of the group to be registered in 1994, is the most heavily used in California (CDPR, 2020), and has a wide range of urban uses including indoor and outdoor pest control, landscape maintenance, and as pet treatments for fleas and ticks (CDPR, 2022). Manufacturers also incorporate imidacloprid into construction materials, such as polystyrene insulation, vinyl siding, adhesives, sealants, and pressure-treated wood (Friends of the Earth (FOE) & Responsible Purchasing Network (RPN), 2017; Sadaria et al., 2017).

With a similar structure to nicotine, neonicotinoids are highly water soluble, non-volatile, and mobile, with a low affinity for particles. While susceptible to photolysis, field studies suggest these pesticides can linger in aquatic ecosystems (Bonmatin et al., 2015). Transformation products like imidacloprid-urea can form in the environment (Todey et al., 2018).

Imidacloprid is ubiquitously detected in the aquatic environment including in surface water, wastewater effluent, urban runoff, and groundwater (Buzby et al., 2020; CDPR, 2016; Hladik & Kolpin, 2016; Sadaria et al., 2017). This has raised concerns as neonicotinoids are known to be highly toxic to invertebrates, targeting the central nervous system, and could pose a risk to non-target species, especially pollinators and aquatic invertebrates.

Fipronil is a phenylpyrazole insecticide, another more recently developed class of insecticides that also emerged as a replacement to organophosphates and other phased out pesticides. Fipronil is used in urban areas for outdoor structural application around buildings, pet treatments for fleas and ticks, as well as insect control bait and underground injections for termites (Sadaria et al., 2017). Unlike other states, California does not permit the agricultural use of fipronil (CDPR, 2017).

In aquatic environments, fipronil is generally present in both water and sediment due to its moderate water solubility and partitioning to particles. Fipronil degradation occurs through multiple pathways, including through exposure to sunlight and microbial activity. However, the resulting degradates (including fipronil sulfide, fipronil sulfone, and fipronil desulfinyl) show similar physical, chemical, and toxicological characteristics to fipronil, and appear to be more stable in the environment.

Fipronil and its degradates have been regularly detected in a variety of environmental matrices including surface water, wastewater, urban runoff, and sediment (Budd et al., 2015; B. Du, Wong, et al., 2020; Heberger et al., 2020; Nowell et al., 2021; Sadaria et al., 2017; Weston & Lydy, 2014). Like imidacloprid, there are concerns surrounding the effects of fipronil on pollinators and aquatic life with particular effects to the nervous system. Fipronil and its degradates have been linked to toxicity in benthic invertebrates, though further toxicity and environmental fate data are needed. Based on California monitoring data showing fipronil and degradates present at levels greater than toxicity thresholds, in the mid 2010s, CDPR examined mitigation options and worked with pesticide manufacturers to implement new label language expected to limit use by professional applicators while continuing to control pests. These changes, implemented in 2017 (with use of previously manufactured products permitted until 2019), are designed to reduce levels of this pesticide in urban runoff (Moran et al., 2020). CDPR continues to monitor fipronil in the environment to determine the effectiveness of these restrictions.

Pyrethroids are a group of synthetic organic insecticides similar to naturally produced pyrethrins. Type I pyrethroids generally lack an α-cyano functional group; commonly used examples include bifenthrin and permethrin. Those with this functional group are called Type II pyrethroids; an example of a Type II pyrethroid is cyfluthrin. Bifenthrin, cyfluthrin, and permethrin are the most commonly detected pyrethroids in urban settings (Budd et al., 2020) and are the focus of the pyrethroids synthesis and risk screening in this report.

Pyrethroids have many urban uses, with applications in indoor and outdoor pest control as well as products for landscape maintenance, pet shampoos to control fleas and ticks, and mosquito-protection clothing (CDPR, 2022). Additionally, permethrin-containing shampoos and lotions have been approved by the US Food and Drug Administration (FDA) to control human head lice and scabies. Generally, pyrethroids are poorly water soluble, with a tendency to partition to organic matter and sediment.

Several pyrethroid compounds, such as bifenthrin, cyfluthrin, and permethrin, are detected consistently in sediment, soil, surface water, wastewater effluent, and urban runoff (Budd et al., 2020; Holmes et al., 2008; Kuivila et al., 2012; Markle et al., 2014; Tang et al., 2018; Weston et al., 2013; Weston & Lydy, 2012). These compounds are highly toxic to aquatic organisms. In 2012, California placed restrictions on pyrethroid application methods used for structural pest control to limit the amount of pesticides entering urban waterways (Moran et al., 2020). CDPR continues to monitor pyrethroids in the environment to assess the effectiveness of these restrictions and to determine if supplemental management actions are needed. Additionally, the Central Valley Regional Water Quality Control Board has amended the Water Quality Control Plan for the Sacramento River and San Joaquin River Basins that requires pyrethroid monitoring and establishes total maximum daily loads (TMDLs) for discharge (CRWQCB-CVR, 2017).

3.6.2. Occurrence summary of select urban pesticides in California waters

We reviewed statewide occurrence data and screened associated risks for imidacloprid and its degradates, fipronil and its degradates, and three major pyrethroids—bifenthrin, cyfluthrin, and permethrin. These compounds were prioritized based on several criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, 90th percentile concentrations near or exceeding toxicity thresholds, and selection of compounds that represent diverse insecticide classes.

The scope of this review included water samples only; sediment data were not reviewed. While water data are sufficient to provide a proof of concept, a more comprehensive assessment that includes sediment for more hydrophobic compounds like fipronil and pyrethroids should be conducted in the future. The availability and quality of reported pesticide concentrations in freshwater, estuarine, and marine waters vary significantly, as we discuss below. In particular, we found more limited monitoring data available for these pesticides in marine waters.

Imidacloprid, with its relatively high water solubility, is an ideal pesticide to examine in ambient water. A few thousand measurements are available in freshwater ecosystems of California (Appendix Table A.6). While a significant number of these measurements were made using methods with insufficient sensitivity (detection limits greater than the selected threshold), those made with more sensitive methods detected imidacloprid in the majority of samples, and revealed maximum concentrations of up to 9.9 µg/L, and 90th percentile concentrations of up to 1.1 µg/L. Many of the higher concentrations observed in California are likely associated with agricultural rather than urban uses. Limited available monitoring data for imidacloprid degradates include detections of imidacloprid-urea at a maximum concentration of 0.058 µg/L; imidacloprid-desnitro-olefine and imidacloprid-olefine were not detected.

In estuarine settings, maximum imidalcoprid concentrations of up to 0.48 µg/L, and 90th percentile concentrations of 0.0045 µg/L, were reported in the database via over 100 measurements using a relatively sensitive method (Appendix Table A.6). As with freshwater settings, only imidacloprid-urea was observed in estuarine water, at a maximum concentration of 0.0095 µg/L. A few dozen measurements are available for imidacloprid in California marine environments; the maximum concentration in the database is 0.039 µg/L (Appendix Table A.6). No data are available for imidacloprid degradates in marine settings along the California coast.

Fipronil and its degradates, with more intermediate water solubility, are also widely observed in the surface waters of California (Appendix Table A.6). Since fipronil only has urban uses in California, all data can be linked to urban uses. Because a portion of these compounds partitions to suspended particles, in general dissolved phase concentrations in the database do not fully represent presence in the water column. The majority of freshwater fipronil data in the database were collected using methods with insufficient sensitivity (detection limits greater than the selected threshold), resulting in non-detects (Appendix Table A.6). Maximum dissolved phase concentrations of 0.13 µg/L, and total water concentrations (dissolved phase and particulate phase combined) of 0.38 µg/L, were reported. Data are also available for five fipronil degradates, with maximum freshwater dissolved phase and total water concentrations in the order fipronil sulfone (0.14 µg/L [dissolved phase], 0.098 µg/L [total water]) > fipronil desulfinyl (0.087 µg/L [dissolved phase], 0.08 µg/L [total water]) > fipronil sulfide (0.041 µg/L [dissolved phase], 0.016 µg/L [total water]) > fipronil amide (0.025 µg/L [dissolved phase], 0.039 µg/L [total water]) > desulfinylfipronil amide (0.013 µg/L [dissolved phase], 0.023 µg/L [total water]).

In the database, a smaller number of dissolved phase concentrations are reported for estuarine waters of California compared to freshwater (Appendix Table A.6). These include maximum concentrations of fipronil (0.012 µg/L), fipronil sulfone (0.0082 µg/L), fipronil desulfinyl (0.006 µg/L), and fipronil sulfide (0.0054 µg/L). The methods used for estuarine monitoring are generally not sufficiently sensitive for risk screening. In marine settings, fipronil has been observed at maximum dissolved phase concentrations of 0.019 µg/L, and maximum total water concentrations of 0.06 µg/L (Appendix Table A.6). The 90th percentile concentrations are 0.016 (dissolved phase) and 0.028 µg/L (total water). Maximum marine dissolved phase concentrations of fipronil degradates include fipronil sulfone (0.026 µg/L), fipronil desulfinyl (0.01 µg/L), and fipronil sulfide (0.006 µg/L). Fipronil amide and desulfinylfipronil amide have not been detected in estuarine or marine sites in California.

Pyrethroids have low water solubility and a high tendency to adsorb to suspended particles and bed sediment. Data for multiple pyrethroids are abundant in California freshwater environments; CDPR scientists recently reviewed a significant quantity of long term monitoring data (Budd et al. 2020). In the occurrence database, maximum dissolved phase and total water concentrations, respectively, of key pyrethroids in freshwater ecosystems include bifenthrin (0.14 µg/L [dissolved phase], 19 µg/L [total water]) > permethrin (0.015 µg/L [dissolved phase], 10 µg/L [total water]) > cyfluthrin (non-detect [dissolved phase], 3.4 µg/L [total water]) (Appendix Table A.6).

In contrast, estuarine water monitoring is more limited in California compared to freshwater (Appendix Table A.6). In the database, maximum bifenthrin concentrations for the dissolved phase are 0.012 µg/L, while for total water the maximum is 0.16 µg/L. The maximum total water cyfluthrin concentration reported for estuarine sites in California is 0.024 µg/L. In the database, estuarine permethrin monitoring data are available only for the dissolved phase, with a maximum concentration of 0.015 µg/L.

More data are available for pyrethroids in marine waters of California compared to estuarine waters (Appendix Table A.6). In the database, maximum total water concentrations for key pyrethroids in marine settings include bifenthrin (0.42 µg/L) > permethrin (0.22 µg/L) > cyfluthrin (0.025 µg/L).

3.6.3. Risk screening

For imidacloprid, fipronil and its sulfide and sulfone degradates, and pyrethroids, the pesticide ecotoxicity thresholds we used for this risk screening are the California water quality criteria equivalents to Federal Clean Water Act 304(a) criteria (“water quality criteria”) based on a methodology developed at the University of California at Davis for developing water quality criteria equivalents for chemicals with more limited aquatic organism toxicity data sets. This “UC Davis method” procedure (TenBrook et al., 2009) has been used to derive aquatic life criteria equivalents for more than a dozen chemicals (all of which are for pesticides in freshwater) using data for multiple species. While other ecotoxicity thresholds exist, some indicating risk at somewhat lower concentrations, these UC Davis method thresholds are preferred for informing Water Boards monitoring and management activities concerning pesticides. Because the UC

Davis method is not watershed-specific, these criteria equivalents are appropriate for any freshwater ecosystem in North America.

There are not currently enough toxicological data to support derivation of thresholds for the fipronil degradates fipronil desulfinyl, fipronil amide, and desulfinylfipronil amide, or the imidacloprid degradates imidacloprid-urea, imidacloprid-olefine, and imidacloprid-desnitro-olefine using the UC Davis method. Therefore, we used the lowest available USEPA Office of Pesticide Programs' aquatic life benchmarks for freshwater species for fipronil desulfinyl and imidacloprid-urea. These aquatic life benchmarks are derived using data for a single species. Thresholds are currently unavailable for fipronil amide, desulfinylfipronil amide, and imidacloprid-desnitro-olefine. Likewise, thresholds specific to marine or estuarine environments are not available for these pesticides.

A limitation of the present risk screening is that it is based on individual compounds, and does not account for the cumulative impacts of exposure to mixtures of related chemicals with the same modes of action. A robust assessment of each of these classes of insecticides could be accomplished using an approach such as concentration addition. This approach assumes each compound within a mixture differs only in its potency, such that all the compounds behave as concentrations or dilutions of one another, and they do not interact. Unfortunately, summing risk quotients developed using summary level concentrations (90th percentile concentrations for individual pesticides based on all California occurrence data since 2005) would require the assumption that the observed detections happened concurrently in time and space, and may overestimate the hazard if that is not true. Therefore, a database structure that contains time- and site-specific contaminant concentrations and ancillary data (e.g., total suspended solids, dissolved organic carbon) is needed to assess additive risk for each sample; risk screening could then be conducted based on cumulative exposure. A more complex and comprehensive risk assessment using concentration addition is beyond the scope of the present report, but could be integrated into future evaluations.

Imidacloprid is designed to act on post-synaptic nicotinic acetylcholine receptors in the nervous system, which are found only in insects (Matsuda et al., 2001). Its mode of action is similar on non-target insects and related invertebrates. In vertebrate organisms, imidacloprid's binding affinity is lower, but nicotinic receptors are present at neuromuscular junctions as well as in the central nervous system. Sub-lethal effects of exposure include genotoxic and cytotoxic effects, impaired immune function, and reduced growth and reproductive success (Gibbons et al., 2015).

The UC Davis method chronic criterion equivalent for imidacloprid effects on aquatic life in the watersheds of the Central Coast Regional Water Quality Control Board is 0.016 µg/L (Bower & Tjeerdema, 2019). Comparison of occurrence data (90th percentile concentration of 1.1 µg/L) with this threshold indicates imidacloprid is present in California freshwater habitats at levels of High Concern, with an ecological risk quotient of 69. Maximum detected levels of imidacloprid in estuarine and marine waters also exceed the ecotoxicological threshold, although 90th percentile concentrations fall below the threshold, with RQs between 0.28 and 0.5 derived from measurements of sufficient sensitivity.

Imidacloprid degradates appear to be lower concern than the parent compound based on the lack of detections of imidacloprid-olefine and imidacloprid-desnitro-olefine and very low RQs for imidacloprid-urea. However, without toxicological thresholds for the olefine and desnitro-olefine degradates, it is difficult to assess whether methods are adequately sensitive for detecting these compounds. Additionally, some studies have indicated that imidacloprid-olefine may be more toxic than the parent compound in insects (A. Huang et al., 2021; Suchail et al., 2001). The only threshold available for imidacloprid-urea is the aquatic life benchmark for acute invertebrate exposure, so while RQs show occurrence concentrations are at least five orders of magnitude below this threshold, this acute-only comparison is likely to underestimate the risk for chronic exposure to both invertebrates and other aquatic species.

We classify imidacloprid as High Concern, driven by the parent compound's levels in California freshwater. It is important to note that higher detections in California are often associated with agricultural rather than urban uses.

Fipronil is designed to disrupt the normal influx of chloride ions by the GABA-gated chloride channel and, at sufficient doses, causes excessive neural excitation, severe paralysis, and death in insects (Gunasekara et al., 2007). Its mode of action is similar in non-target invertebrates and vertebrates, which also have GABA receptors (although with lower fipronil binding affinity). Several of this pesticide's degradates have been indicated to be equal or more potent than the parent compound (Schlenk et al., 2001; Weston & Lydy, 2014).

The UC Davis method chronic criterion equivalent for fipronil effects on aquatic life is 0.0032 µg/L (Bower & Tjeerdema, 2017). For purposes of risk screening, we focused on freshwater measurements in the occurrence database with detection limits similar to or below this threshold, for which the 90th percentile concentration is 0.049 µg/L (total water). Comparison of this concentration with the threshold results in the RQ of 15, consistent with a classification of High Concern. Estuarine monitoring data in the database was produced by methods that were not sufficiently sensitive for risk screening (detection limits greater than the threshold). Marine risk quotients fall in the Moderate Concern range, with adequate method sensitivity and RQs between 5 and 9, depending on whether measurements made for the dissolved phase or total water are used.

Risk screening indicates a higher level of concern associated with a key fipronil degradate, fipronil sulfone. For freshwater monitoring data using methods with greater sensitivity (detection limits in the range of toxicity thresholds) in the occurrence database, the total water 90th percentile concentration of fipronil sulfone is 0.044 µg/L. The freshwater RQ calculated using the UC Davis method chronic criterion of 0.00017 µg/L (Bower & Tjeerdema, 2017) is 260, consistent with a classification of Very High Concern. While estuarine data of sufficient sensitivity are not available, comparison of the marine 90th percentile concentration (0.014 µg/L) to the threshold results in the RQ of 82.

The degradate fipronil sulfide also poses concerns based on available occurrence data. Comparison of the total water 90th percentile concentration of fipronil sulfide, 0.003 µg/L, to the UC Davis method chronic criterion, 0.00014 µg/L (Bower & Tjeerdema, 2017) reveals an RQ of 21, consistent with a classification of High Concern, similar to the parent compound. For marine settings, a 90th percentile concentration of 0.0005 µg/L results in an RQ of 3.6, suggesting Moderate Concern.

The fipronil degradate fipronil desulfinyl appears to be of lower concern than the parent compound. The maximum 90th percentile concentration derived using methods of sufficient sensitivity is 0.017 µg/L for freshwater, well below the USEPA aquatic life benchmark for chronic exposure to fish, 0.53 µg/L. However, this threshold may have more uncertainty. Aquatic life benchmarks are typically determined by an internal USEPA scientific process conducted in collaboration with CDPR (ROCKS committee process; https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/guidance-residues-concern-ecological-risk) to the extent the agency has toxicity data. Benchmarks are the lowest value among the aquatic toxicity data in USEPA's possession; these data must pass USEPA quality assurance review prior to use and may include both industry-generated values and values from the scientific literature. Chronic benchmarks, which can be based on NOAECs, NOECs, or short-term non-lethal endpoints (e.g., growth, reproduction), do not include a safety factor.

Fipronil amide and desulfinylfipronil amide were detected in California freshwater, but without toxicological thresholds, it is difficult to assess whether methods are adequately sensitive for detecting these compounds.

For somewhat hydrophobic and lipophilic contaminants like fipronil and its degradates, a screening that focuses only on occurrence and toxicity in water may not be sufficient to assess risks. A recent CDPR review recommends examining concerns for both water and sediment to fully evaluate the risks of these compounds to aquatic ecosystems (Budd et al., 2015). Bower and Tjeerdema (2017) suggested that the cumulative impacts resulting from co-exposure with fipronil and its degradates should be evaluated using an additive mixture model, as they are assumed to act with the same mode of action, though mixture studies are lacking. Future work to assess the risks posed by fipronil and degradates across multiple matrices would require site-specific contaminant concentrations and ancillary data (e.g., organic carbon) in water and sediment.

We classify fipronil and degradates as Very High Concern, driven by the degradate fipronil sulfone in fresh and marine waters. All uses of fipronil in California are urban rather than agricultural.

Pyrethroids are broad spectrum pesticides that bind to and prevent the closure of voltage-gated ion channels in axonal membranes. Bifenthrin and permethrin are Type I pyrethroids and cyfluthrin is a Type II pyrethroid (containing an α-cyano moiety); the two types have slightly different toxicological profiles (Breckenridge et al., 2009). Because they have the same general mode of action, mixtures of pyrethroids should be considered in an additive manner when considering ecotoxicological risk by using the concentration addition model and following either the toxic unit or relative potency factor approach (Fojut et al., 2012). Fojut et al. (2012) also recommended that risk assessors use the dissolved phase pyrethroid concentration when assessing occurrence data because pyrethroids are hydrophobic and primarily partition to sediments and solid materials in the water column, and the dissolved concentration appears to be the best predictor of the bioavailable fraction.

The UC Davis method individual chronic criteria for bifenthrin, permethrin, and cyfluthrin effects on aquatic life are 0.0006 µg/L, 0.002 µg/L, and 0.00005 µg/L, respectively (Fojut et al., 2012). Looking at each compound individually, bifenthrin appears to be High Concern, with comparison of occurrence data using sufficiently sensitive methods in fresh and estuarine waters with its ecotoxicological threshold resulting in RQs generally between 10 and 100. Similarly, comparison of occurrence data of permethrin using sufficiently sensitive methods indicates Moderate Concern in freshwater and marine waters, based on RQs of 2.5 and 9.5, respectively. Cyfluthrin occurrence data indicate High Concern, with RQs of 22, 36, and 22 in fresh, estuarine, and marine waters, respectively. However, almost all the occurrence data with sufficient method detection sensitivity are total concentrations rather than the dissolved fraction; bioavailability may actually be lower if samples had a high amount of suspended sediment. A limitation of the occurrence database is that this information is not readily available.

A robust assessment of pyrethroids in aquatic ecosystems requires more comprehensive and site-specific information to account for the risks of exposure to mixtures in water (e.g., concentration addition as recommended by Fojut et al., 2012) as well as sediment (which is outside of the scope of this review). For hydrophobic and lipophilic contaminants like pyrethroids, water may not be the most important matrix for risk screening. In fact, studies indicate more significant toxicity concerns associated with pyrethroids in the sediment of freshwater ecosystems in California (Holmes et al., 2008; Anderson et al., 2012; Budd et al., 2020). Therefore, a database structure that contains site-specific contaminant concentrations and ancillary data is needed to fully assess the risk posed by pyrethroids across multiple matrices.

We classify pyrethroids as High Concern (considering water only), driven by observations of two of the three compounds selected for review (bifenthrin, cyfluthrin). This occurrence data synthesis and risk screening does not distinguish between ambient aquatic ecosystems influenced by urban vs. agricultural land use, and both types of use may contribute to environmental contamination.

3.7. Select pharmaceuticals

3.7.1. Definition, chemical properties, and use

Pharmaceuticals are an incredibly varied use class of thousands of compounds defined by their designed biological activity for therapeutic use in humans and animals. The class includes a wide range of prescription, over-the-counter, and veterinary drugs and their metabolites as well as diagnostics agents, vitamins, and other supplements. Generally, pharmaceuticals are small, polar compounds with a complex chemical structure and are pharmacologically active.

Pharmaceuticals can enter the environment primarily through waste streams from human uses with potential secondary pathways from manufacturing losses, sewering (e.g., flushing down the toilet), as well as animal uses in veterinary clinics and industrial animal farming operations. As analytical methods have advanced, methods have developed to find pharmaceuticals at low concentrations with widespread detections in the environment including surface water, wastewater influents and effluents, sediment, and wildlife (aus der Beek et al., 2016; Lin, Sutton, Sun, et al., 2018; K. A. Maruya et al., 2016; O'Flynn et al., 2021; Patel et al., 2019). This report will focus on groups of pharmaceuticals widely noted to be detected in these environmental matrices including antibiotics, antidepressants, beta-blockers, lipid-lowering drugs, hormones, and painkillers.

Hormones are a heterogeneous group of compounds with varying chemical properties, forms of synthesis, and functions. Here, we will focus on hormones used in pharmaceuticals, highlighting 17α-ethinylestradiol and 17β-estradiol. The former is a synthetic estrogen commonly found in oral contraceptives while the latter is a steroid hormone commonly used in hormone treatments for particular medical conditions. 17α-ethinylestradiol is moderately water soluble and lipophilic. Comparably, 17β-estradiol exhibits a low water solubility. Both are among the most widely used and —due to their continuous discharges—detected estrogenic hormones in the environment. Their presence can lead to adverse effects in reproductive and endocrine systems of both humans and animals.

Another widely used set of pharmaceuticals, painkillers are important to relieve pain or suppress inflammation in living organisms. Nonsteroidal anti-inflammatory drugs (NSAIDs) are common painkillers used to prevent development and progression of cardiovascular diseases, and several types of cancers through reduced inflammation (Davis et al., 2017; C. M. Ulrich et al., 2006). Ibuprofen is a well-known NSAID detected across environmental matrices due to its high water solubility, with potential to cause negative ecological health effects. (Diclofenac, another NSAID with ecotoxicity concerns that is widely used elsewhere, is not commonly used in the US.)

There are many pharmaceutical drugs prescribed to treat cardiovascular disease, a leading cause of mortality in the US. High blood pressure is a significant risk factor minimized by the use of drugs like beta blockers that change the rhythm of the heart or pressure in the blood vessels. Metoprolol is among the most prescribed beta blockers, and overall pharmaceuticals in the world, with increasing detections in the aquatic environment. Lipid-lowering drugs are another class used to treat high cholesterol, another major risk factor for cardiovascular disease. Nationally, more than a quarter of adults over the age of 40 use lipid lowering medications to treat cholesterol (Gu et al., 2014). Within the lipid-lowering drugs, gemfibrozil is widely administered and commonly detected in water matrices. Both beta blockers and lipid lowering drugs have been identified as widely observed chemical contaminants with potential to harm the environment.

Antidepressants are a pharmaceutical class that directly affects the nervous system to treat conditions such as depression, anxiety, and chronic pain. Antidepressants are among the most commonly prescribed drugs in the US with the CDC reporting that during 2011-2014, 12.7% of persons aged 12 and over took antidepressants medication in the past month, and one-fourth of persons who took antidepressant medication had done so for the past 10 years (Pratt et al., 2017). Fluoxetine is a widely prescribed antidepressant that is continuously detected in environmental matrices worldwide. These drugs can negatively impact ecological health; physiological effects have been observed in fish, mollusks, crustaceans, algae, and protozoans, and can affect fish behavior and survival at the low µg/L level (Weinberger & Klaper, 2014).

Antibiotics are a group of pharmaceuticals used to inhibit the growth and even destroy microbial pathogens present in humans and animals. Several groups of antibiotics that are widely prescribed are also widely detected in the environment, particularly macrolide antibiotics and quinolones. Azithromycin, clarithromycin, and erythromycin are the most common macrolides, which function by reducing the production of microbial proteins. Quinolones, especially ciprofloxacin, make it difficult for microbes to repair their DNA. Their essential function also makes antibiotics directly harmful to the environment and a particular concern for the potential evolution of antibiotic resistant genes and bacteria. Further studies are needed to understand the fate and ecotoxicological risk of antibiotics in the environment.

Currently, there is an increasing policy focus on proper pharmaceutical prescription, use, and disposal occurring at the federal, state, and local levels to consider management actions to decrease their presence in the environment. Recently, the USEPA prohibited the sewering of many hazardous waste pharmaceuticals to protect water quality. At the state level, California passed the first legislation in 2018 requiring a state-wide take-back program for pharmaceuticals and sharps used in households. California Senate Bill SB 212 (Jackson) was motivated by concerns about human health (e.g., antibiotic resistance in infectious bacteria, drug abuse, and accidental poisoning), rising drug expenditures, and environmental contamination (Wagoner, 2018). The California State Water Resources Control Board’s monitoring priorities for constituents of emerging concern (Tadesse, 2016) includes up to seven pharmaceutical compounds recommended for monitoring wastewater effluent, stormwater runoff, and receiving waters.

3.7.2. Occurrence summary of select pharmaceuticals in California waters

We reviewed occurrence data and screened associated risks for ten selected pharmaceuticals in water from freshwater, estuarine, and marine settings in California. These compounds were prioritized based on several criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, 90th percentile concentrations near or exceeding toxicity thresholds, selection of compounds that represent diverse use classes, and prioritization of use categories where there are available management actions. The ten compounds that we selected for discussion are the hormones 17α-ethinylestradiol and 17β-estradiol; a non-steroidal anti-inflammatory drug (NSAID), ibuprofen; the beta-blocker use to treat blood pressure, metoprolol; a lipid-lowering drug, gemfibrozil; an antidepressant, fluoxetine; and the antibiotics azithromycin, clarithromycin, erythromycin, and ciprofloxacin. The availability and quality of reported pharmaceutical concentrations in freshwater, estuarine, and marine waters vary significantly, as we discuss below. In particular, we found minimal to no monitoring data available for these pharmaceuticals in marine waters.

Monitoring data for the hormones 17α-ethinylestradiol and 17β-estradiol only include data for freshwater systems in California; data for marine and estuarine systems are lacking (Appendix Table A.7). The summarized data reveal that methods used for monitoring these compounds are not sufficiently sensitive, resulting in low detection frequencies (1-2%) and estimated 90th percentile concentrations driven by high detection limits. The estimated 90th percentile concentration for 17α-ethinylestradiol and 17β-estradiol are based on the median reported detection limits of 100 µg/L and 0.8 µg/L, respectively. A published study monitoring estrogenic compounds in the Sacramento-San Joaquin River system detected 17α-ethinylestradiol at up to 0.0005 µg/L (Lavado et al., 2009). Outside the database in published literature, monitoring of three-effluent dominated rivers in southern California, during dry weather (low flow) conditions, did not detect 17β-estradiol above detection limits (0.005 µg/L)

For a number of the selected pharmaceuticals, including ibuprofen (NSAID), metoprolol (beta blocker), gemfibrozil (lipid-lowering), and fluoxetine (antidepressant), there are a significant quantity of records in freshwater in the occurrence database (Appendix Table A.7). However, the summarized data indicate methods used for most of these studies are not sufficiently sensitive, resulting in low to zero detection frequencies. For a subset of studies using more sensitive methods, 90th percentile concentrations for ibuprofen and fluoxetine are based on the median detection limit of 0.02 and 0.005 µg/L, respectively. Metoprolol was also largely non-detect, and detection limits were either not reported or reported to be 10 µg/L. For a subset of studies using more sensitive methods, the 90th percentile concentration for gemfibrozil was 0.3 µg/L.

Key literature studies provided important additional occurrence data for California. Specifically, monitoring of the Los Angeles, San Gabriel, and Santa Clara Rivers, three effluent-dominated rivers in southern California, during dry weather (low flow) conditions revealed concentrations of ibuprofen, gemfibrozil, and fluoxetine of up to 0.04, 0.32, and 0.072 µg/L, respectively (K. A. Maruya et al., 2016; Sengupta et al., 2014).

There were more limited data on antibiotics in freshwater systems in the occurrence database, with the exception of erythromycin (Appendix Table A.7). There were hundreds of freshwater records for erythromycin in the database; however, most of the methods deployed were insufficiently sensitive, resulting in nearly all non-detects. Therefore, the 90th percentile concentration for this pharmaceutical was estimated based on the median detection limit of 10 µg/L. The 90th percentile concentrations of azithromycin and ciprofloxacin in freshwater were 0.005 µg/L and 0.007 µg/L (dissolved water fraction) based on seven data points. There are no freshwater data for clarithromycin.

In contrast, estuarine water measurements, while less numerous, show ubiquitous detection of several pharmaceuticals using more sensitive methods with low detection limits (Appendix Table A.7). Metoprolol, gemfibrozil, and fluoxetine were detected in 100% of 14 dissolved phase samples, with 90th percentile concentrations in estuarine waters of 0.74, 1.1, and 0.052 µg/L, respectively. Ibuprofen was detected in 29% of the same samples, with a 90th percentile concentration of 1.1 µg/L in the dissolved phase. While the 90th percentile concentrations estimated for five total water samples present in the database were lower for these compounds, these data were from different studies and represent differences in sampling locations rather than representative differences among the fractions.

The antibiotics azithromycin, clarithromycin, erythromycin, and ciprofloxacin have been frequently detected in estuarine waters; estimated 90th percentile concentrations in the database were 0.69, 0.19, 0.07, and 0.28 µg/L, respectively (Appendix Table A.7).

No marine data were present for any of the selected pharmaceuticals in the occurrence database (Appendix Table A.7). We did find limited marine occurrence data in published literature, including a study of various emerging contaminants in the Southern California Bight near effluent discharge points that found gemfibrozil to be frequently detected (90% detection frequency) at concentrations up to 0.013 µg/L, and also detected ibuprofen at concentration up to 0.03 µg/L (30% detection frequency). Another published study detected fluoxetine in coastal waters of California at concentrations up to 0.09 µg/L (Nödler et al., 2014).

3.7.3. Risk screening

Pharmaceuticals are designed to be bioactive chemicals and may have a wide variety of effects even at low concentrations, as this class of chemicals is grouped by use rather than chemical similarity. Even within use groups, such as antibiotics, compounds may have very different structures, and therefore different modes of action. We conducted a risk screening for each pharmaceutical that recognizes the significant differences among compounds as well as their applications.

Hormones are known endocrine disruptors and may have additive effects. We classified the hormones 17α-ethinylestradiol and 17β-estradiol as Possible Concern because methods used to monitor these compounds in estuarine, marine, and freshwater systems are insufficiently sensitive, with detection limits several orders of magnitude greater than ecotoxicity thresholds. The thresholds for 17α-ethinylestradiol (0.000035 µg/L) and 17β-estradiol (0.0004 µg/L) are environmental quality standards from the European Union and are both based on species sensitivity distributions (NORMAN ecotoxicological database).

Nonsteroidal anti-inflammatory drugs are known to cause sub-lethal effects in invertebrates with chronic exposure at low, environmentally relevant concentrations (Parolini, 2020). We classified ibuprofen as High Concern, based on an RQ value of 42, calculated from the 90th percentile concentration for estuarine waters, relative to the ecotoxicity threshold for ibuprofen, 0.026 µg/L, which is based on a species sensitivity distribution developed for reproduction (Q. Huang et al., 2018). While there is not sufficient freshwater and marine water occurrence data in the database to conduct a risk screening, maximum values reported in the literature suggest freshwater and marine waters are comparable to ecotoxicity thresholds.

We classified metoprolol and gemfibrozil as Moderate Concern and fluoxetine as High Concern based on risk quotients above one calculated from the 90th percentile concentration from estuarine water relative to the ecotoxicity threshold of each compound. The ecotoxicity thresholds for metoprolol (0.1 µg/L), gemfibrozil (0.3 µg/L), and fluoxetine (0.1 µg/L) were deterministically derived from experimental studies (Minguez et al., 2016; Orias & Perrodin, 2013). The 90th percentile concentrations in freshwater for metoprolol and gemfibrozil are all based on detection limits that are at or above the toxicity thresholds for these compounds, consistent with Moderate Concern. Risk quotients calculated for gemfibrozil from maximum values reported in the literature are below one, while RQ values calculated for fluoxetine with maximum values reported in the literature are above marine thresholds and below freshwater thresholds.

The highest RQ calculated for the selected antibiotics was for azithromycin (RQ of 36), which we classified as High Concern. The 90th percentile concentration in estuarine waters was compared to the ecotoxicity threshold of 0.019 µg/L, which was deterministically calculated from chronic *Microcystis aeruginosa* toxicity data using an assessment factor of 10 (NORMAN ecotoxicological database). The ecotoxicity thresholds for clarithromycin (0.12 µg/L; NORMAN ecotoxicological database), erythromycin (0.02 µg/L; Verlicchi et al., 2012), and ciprofloxacin (0.089 µg/L; NORMAN ecotoxicological database) have greater uncertainty compared to that of azithromycin, and comparison with occurrence data resulted in risk quotients between one and five.

In summary, we classified the pharmaceuticals ibuprofen, fluoxetine, and azithromycin as High Concern. We classified metoprolol and gemfibrozil as Moderate Concern. We also classified the antibiotics clarithromycin, erythromycin, and ciprofloxacin as Moderate Concern. Finally, we classified the hormones 17α-ethinylestradiol and 17β-estradiol as Possible Concern.

3.8. Select personal care and cleaning product ingredients

3.8.1. Definition, chemical properties, and use

Personal care and cleaning product ingredients (PCCPs) are a wide ranging use class of chemicals defined by their application in consumer products including any products applied on an external part of the body (e.g., skin, hair, nails), used for oral hygiene, or those used to clean and disinfect around homes and businesses. PCCPs are for external usage only, unlike pharmaceuticals (Section 3.7). There are no defining physical and chemical characteristics due to the great variety of compounds present within the class. Among PCCPs, antimicrobials and synthetic musk compounds (SMCs) are two chemical groups of particular interest, and have been frequently detected in the environment.

The term antimicrobials is broad and includes antibiotics (we have classified ingested antibiotics separately in pharmaceuticals) and registered pesticides that can kill or limit the growth of other microorganisms (e.g., parasites, viruses, fungi). Use of antimicrobials is registered by the USEPA or US Food and Drug Administration (FDA). Antimicrobials are used in a wide variety of products including PCCPs, such as skin-contact soaps and antiseptics, body washes, lotions, and cleaning products. One of the most well-studied substances in the class, triclosan, was used in soaps and cleaning products, and is also a material preservative in plastics, polymers, and textiles. Triclocarban is another well-known antimicrobial with similar chemical properties as triclosan, including low water solubility and high affinity for solids. Concerns about triclosan and triclocarban in the environment include persistence, bioaccumulation, endocrine disruption, toxicity to aquatic and other organisms, and antimicrobial resistance. Triclosan and triclocarban are among the emerging contaminants detected most ubiquitously in the environment including municipal wastewater, surface waters, sediments, and biota (Dhillon et al., 2015; Muir et al., 2017; Tran et al., 2018; Yun et al., 2020).

Growing concern about the human health and environmental impacts of triclosan and triclocarban have motivated restrictions on their use. This is especially true as many applications of these compounds include down the drain disposal, which can lead to significant loadings to the environment through wastewater effluent. Starting in 2016, the FDA banned the use of triclosan and triclocarban in over-the-counter soap and wash products, over-the-counter topical antiseptics in health-care settings, and over-the-counter hand sanitizers (FDA, 2016).

Synthetic musk compounds are a class of organic chemicals used extensively as fragrance ingredients including cosmetics, detergents, fabric softeners, perfumes, and soaps. Polycyclic musks, particularly galaxolide (or HHCB) and tonalide (or AHTN), have become the most important group of commercial SMCs. Galaxolide is a high production volume chemical in the US, produced or imported in quantities between one and ten million pounds in 2015 (USEPA, 2016). These compounds are only slightly soluble in water and moderately volatile, with a high lipophilicity that makes them more likely to partition from water into solids and remain there.

Galaxolide and tonalide are frequently detected across environmental compartments including surface water, wastewater effluents, sediment, air, and various biota (Heberger et al., 2020; K. Maruya et al., 2018; Wong et al., 2019; Yee et al., 2019). There are several potential environmental health concerns for both these compounds, including endocrine disruption, genetic damage, persistence, and bioaccumulation. Neither of these contaminants have been targeted for regulatory action in the US. California has listed galaxolide and tonalide as designated chemicals for biomonitoring; however, biomonitoring has not yet begun for either compound (Biomonitoring California, 2021).

3.8.2. Occurrence summary of select personal care and cleaning product ingredients in California waters

We reviewed occurrence data and screened associated risks for four selected personal care and cleaning product ingredients, triclosan, triclocarban, galaxolide, and tonalide, in water from freshwater, estuarine, and marine settings in California. These compounds were prioritized based on several criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, and 90th percentile concentrations near or exceeding toxicity thresholds. The availability and quality of data for these compounds varied, as we discuss below. In particular, we found minimal monitoring data available for these compounds in marine waters.

There are nearly three hundred records associated with triclosan and triclocarban in freshwater (Appendix Table A.8). However, methods used for most of these studies were not sufficiently sensitive, resulting in low to zero detection frequencies. In some cases, detection limits were not reported in public databases; detection limits varied significantly from 0.02–25 µg/L.

In contrast, estuarine water measurements, while less numerous, show detections of triclocarban and triclosan (Appendix Table A.8). The estimated 90th percentile concentrations of triclocarban and triclosan quantified in estuarine waters using methods with sufficient sensitivity were 0.047 µg/L and 0.16 µg/L, respectively. The database did not include information on these compounds in marine waters.

Galaxolide and tonalide have not been as widely monitored compared to triclosan and triclocarban. There are a greater number of records for galaxolide and tonalide in freshwater, compared to estuarine and marine waters (Appendix Table A.8). Once again, methods used for monitoring galaxolide in freshwater were insufficiently sensitive, resulting in 90th percentile values that are based on median detection limits (0.5 µg/L dissolved phase, 50 µg/L total water), which is higher than the maximum detected concentration of 0.0003 µg/L in the dissolved phase and 0.1 µg/L for total water. Tonalide has not been detected in freshwater.

In estuarine waters, the 90th percentile concentration for galaxolide was 0.02 µg/L measured in the dissolved phase (Appendix Table A.8). Tonalide has not been detected in estuarine waters above detection limits (0.02 µg/L). There are six records available for galaxolide and tonalide in marine waters, but methods used were insufficiently sensitive, resulting in no or low detection frequencies, and 90th percentile values based on the high detection limit of 0.2 µg/L (Appendix Table A.8).

Key literature studies provide important additional occurrence data for California. Specifically, monitoring of the Los Angeles, San Gabriel, and Santa Clara Rivers, three effluent-dominated rivers in southern California, during dry weather (low flow) conditions revealed concentrations of triclocarban (maximum: 0.102 µg/L), triclosan (maximum: 0.018 µg/L), galaxolide (maximum: 2.8 µg/L), and tonalide (maximum: 0.188 µg/L) (Sengupta et al., 2014; Maruya et al., 2016). Vidal-Dorsch et al. (2012) detected triclosan in the Southern California Bight coastal waters near effluent discharge locations at up to 0.0061 µg/L (40% detection frequency, n=20).

The scope of this review included water samples only; sediment data were not reviewed. While water data are sufficient to demonstrate the risk screening concept, a more comprehensive assessment that includes sediment for these compounds should be conducted in the future.

3.8.3. Risk screening

Since PCCPs are defined by use rather than chemical structure, individual PCCPs can be chemically diverse and have different modes of toxicity. Therefore, we conducted a risk screening for each PCCP compound.

The risk tier evaluations for triclosan, triclocarban, galaxolide, and tonalide are mainly based on a moderate number of data available for estuarine systems. There were insufficient data to evaluate the risks of these compounds in freshwater and marine systems, because of an insufficient number of measurements made using methods of sufficient sensitivity to provide suitable occurrence data relative to risk thresholds.

The ecotoxicity threshold for triclosan (0.47 µg/L for fresh and marine waters) is a Canadian Federal Water Quality Guideline based on robust chronic toxicity data using a species sensitivity distribution (Environment and Climate Change Canada, 2017). Comparison of the 90th percentile concentration of triclosan in estuarine waters results in an RQ of 0.34 for triclosan. This is consistent with Low Concern for triclosan in estuarine settings. While occurrence data in the database for triclosan in freshwater and marine systems are limited, monitoring studies of freshwater and marine water influenced by effluent discharges in Southern California are consistent with this risk categorization (K. A. Maruya et al., 2016; Sengupta et al., 2014; Vidal-Dorsch et al., 2012).

The European Chemicals Agency has developed a freshwater PNEC for triclocarban of 0.094 µg/L from chronic *Daphnia magna* data using an assessment factor of 50 (see Appendix D). The ecotoxicity threshold for triclocarban in marine waters (0.0094 µg/L) was estimated by applying an additional assessment factor to the freshwater threshold. Comparison of the 90th percentile concentration of triclocarban in estuarine waters results in an RQ of 0.5 using the freshwater threshold, and 5 using the marine threshold. These results are consistent with Moderate Concern for triclocarban in estuarine and marine environments. Data reported from studies of effluent-dominated rivers in southern California are consistent with this classification as well.

The ecotoxicity threshold for galaxolide in freshwater, 4.4 µg/L, is an ECHA PNEC deterministically calculated from the *Acartia tonsa* six day development EC10 using an assessment factor of 10 (see Appendix D). The threshold for marine waters of 0.44 µg/L was calculated by applying an additional assessment factor of 10 to the freshwater threshold. The 90th percentile concentration of galaxolide in estuarine waters is well below these thresholds and, therefore, we place galaxolide in the Low Concern category. This is consistent with the outcome of the pre-screening step using the EcoPanel monitoring trigger level for galaxolide, which indicated limited need for additional monitoring (Section 2.2). We are unable to evaluate the risk associated with tonalide because there are no ecotoxicity thresholds available. We would therefore classify tonalide as Possible Concern.

Our risk screening of occurrence data in water of select personal care and cleaning product ingredients in California waters indicates Moderate Concern for triclocarban*;* Low Concern for triclosan and galaxolide*;* and Possible Concern for tonalide*.* A more comprehensive risk screening for these specific contaminants would also include the occurrence and risk in sediment, which are not captured here.

3.9. Per- and polyfluoroalkyl substances

3.9.1. Definition, chemical properties, and use

Per- and polyfluoroalkyl substances (PFAS) are an extensive chemical class of fluorine-rich compounds. All PFAS are composed of carbon chains with attached fluorines. The Organisation for Economic Cooperation and Development (OECD) has recently established a broadly inclusive PFAS definition that includes all fluorinated substances that contain at least one fully fluorinated methyl or methylene carbon atom (Z. Wang et al., 2021). Within this broad class, perfluoroalkyl substances are fully fluorinated, meaning that no carbon-hydrogen bonds are present, and only fluorine atoms are bonded to the carbon backbone of the molecule. In contrast, polyfluoroalkyl substances may contain bonds to other atoms like hydrogen and oxygen. More than 4,700 PFAS are registered for use in consumer, commercial, and industrial applications, though fewer than 50 have been the subject of significant monitoring in environmental media (Y. Wang et al., 2019).

Much of the initial scientific focus on PFAS has been on perfluoroalkyl acids, including perfluoroalkyl sulfonates and perfluoroalkyl carboxylates, which are fully fluorinated chemicals with a sulfonate or carboxylate group at the end, respectively. Two of the most well studied PFAS are perfluorooctane sulfonate (PFOS) and perfluorooctanoic acid (PFOA), both molecules with eight-carbon (C8) chains. PFOS and PFOA are long-chain PFAS, a group that includes perfluoroalkyl sulfonates C6 and above, as well as perfluoroalkyl carboxylates C7 and above. As use of long-chain PFAS has been restricted, industries have substituted short-chain PFAS. These are structurally similar compounds with fewer fluorinated carbons, such as perfluorobutane sulfonate (PFBS), a C4 sulfonate, and perfluorohexanoic acid (PFHxA), a C6 carboxylate. In addition, perfluorinated alkyl ethers like GenX, and polyfluorinated substances have emerged as alternatives in the manufacture of fluoropolymers.

In the environment, some PFAS can transform to perfluoroalkyl acids such as PFOS and PFOA. These PFAS are often referred to as perfluoroalkyl acid precursors or "precursors" for short. For example, perfluorooctane sulfonamide (PFOSA) can convert to PFOS. Perfluoroalkyl acids do not undergo further degradation in the environment.

Compounds within this entire class generally share important physical and chemical properties relevant to fate in the environment. All PFAS are highly persistent or, in the case of precursors, degrade to substances that are highly persistent. Some PFAS, particularly the long-chain compounds, bioaccumulate in the food web, and have been observed in California fish and wildlife (Buzby et al., 2021; Kannan et al., 2006; M. D. Sedlak et al., 2017). However, unlike hydrophobic compounds such as PCBs and PAHs, which concentrate in the fatty tissues of animals, PFAS tend to bind to proteins and accumulate in blood, liver, eggs, and the fatty acid binding proteins in cells (Houde et al., 2011). Other physicochemical properties relevant to fate can vary by PFAS species, including volatility and relative partitioning between water and sediment in aquatic environments. Observations of PFAS in remote environments are consistent with long-range transport (Routti et al., 2019).

Unique, commercially relevant properties including thermal stability, non-reactivity, and repellency to water, dirt, and oil have led to numerous consumer and industrial applications of PFAS, and resulted in widespread environmental contamination. Today, some of the largest volumes of PFAS are used in electronics and machine manufacturing (Glüge et al., 2020). PFAS have also been extensively used in formulation of aqueous film-forming foams (AFFF) used to fight fuel fires, and as processing aids in the production of fluoropolymers (Glüge et al., 2020). Additionally, significant amounts of PFAS have been used as surface protectors of textiles, carpet, and paper products, with PFAS previously estimated to comprise 2 to 3% of textile fibers and 15% of the carpet by weight (Glüge et al., 2020; KEMI Swedish Chemicals Agency, 2014). Recent research has also documented the presence of PFAS in food packaging (Schaider et al., 2017), personal care products (Whitehead et al., 2021), and building materials (Fernández et al., 2021).

In the US, production of PFOS was phased out by 2002, and production of PFOA was phased out by 2015. PFOS was restricted under the global Stockholm Convention in 2009, while PFOA was similarly restricted in 2019, allowing some specific exceptions to a total ban for both compounds. PFOS, PFOA, and some other PFAS are on the European Candidate List of Very High Concern compounds due to persistence, bioaccumulation, and toxicity. 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 PFAS, means continuing use of and exposure to compounds that potentially pose similar risks.

Concerns relating to the persistence of PFAS, the high toxicity of well-studied members of this class, and the pattern of regrettable substitution observed in industry, have led scientific and regulatory bodies to recommend broad, class-based monitoring and management approaches. With respect to monitoring, analytical methods that provide a broader indication of PFAS occurrence include the total oxidizable precursors (TOP) assay, and total organic fluorine methods. The TOP assay can be used to quantify the overall presence of unknown precursors in a sample by oxidizing them to persistent end products, like PFOA, that are then analyzed via targeted analysis. The TOP assay has been applied to California stormwater and wastewater (Houtz et al., 2016; Houtz & Sedlak, 2012), and could be applied more generally to ambient environmental matrices in the state. The more general total organic fluorine methods quantify the collective presence of a wider range of PFAS, but use is currently constrained by relatively high detection limits. Non-targeted analysis and suspect screening methods (Section 4.2) can also supplement the information currently available via targeted analysis.

Management of PFAS as a class has been recommended by several countries within the European Union via a proposal in development to prohibit the production, marketing and use of the class throughout Europe, with exceptions for essential uses such as medical applications. In California, the Department of Toxic Substances Control’s Safer Consumer Products Program has established a clear rationale for management actions directed at the entire PFAS class (Bălan et al., 2021), and has begun to apply this approach to carpets and rugs made or sold in California. Similarly, state bans on PFAS in paper-based food packaging and products intended for infants and children, both of which take effect in 2023, rely on a class-wide approach, rather than bans of individual compounds. The state of Maine took a similar, class-based approach in recent legislation, enacting a law that bans the use of PFAS in all products by 2030, except in instances where the state makes a determination that a use is “currently unavoidable” (LD 1503/HP 1113).

3.9.2. Occurrence summary in California water, sediment, fish, and bivalves

The database contains occurrence data for 20 PFAS in surface waters (Appendix Table A.9), mostly focused on the perfluoroalkyl acids. Most PFAS data are reported as total water fraction, which include contributions from suspended particles. Despite the high level of concern associated with this class of contaminants, the number of records associated with each compound in ambient water is quite limited in California.

In freshwater, 17 PFAS have been monitored (Appendix Table A.9). Among the compounds reported, PFOS and PFOA were detected at the highest concentration, and the 90th percentile concentration reported for the total water fraction were 0.021 µg/L and 0.011 µg/L, respectively. Dissolved phase concentrations for PFOA and PFOS were below detection limits. The next most abundant compounds detected in freshwater were PFHxA (90th percentile concentration = 0.0077 µg/L) > PFPeA (perfluoropentanoic acid, 0.0068 µg/L) > PFBS (0.0056 µg/L) > PFHpA (perfluoroheptanoic acid, 0.0042 µg/L) > PFHxS (perfluorohexanesulfonate, 0.0037 µg/L) > PFDA (perfluorodecanoic acid (PFDA, 0.0032 µg/L) > PFNA (perfluorononanoic acid, 0.0028 µg/L) > PFOSA (0.0013 µg/L). Additional PFAS precursors including 6:2 FTS (6:2 fluorotelomer sulfonate), N-EtFOSAA and N-MeFOSAA (N-ethylperfluoro-1-octanesulfonamidoacetic acid and N-methylperfluoro-1-octanesulfonamidoacetic acid) have also been detected in freshwater systems, but generally at lower concentrations and lower detection frequencies compared to the perfluorocarboxylates and perfluorosulfonates. Literature studies revealed concentrations of PFOA up to 0.037 µg/L in effluent-dominated rivers in southern California (Sengupta et al., 2014).

In estuarine waters, 12 perfluoroalkyl acids have been detected, and PFOA (0.066 µg/L) was the most abundant (Appendix Table A.9). Other 90th percentile concentrations reported include PFHxA (0.024 µg/L) > PFHpA (0.022 µg/L) > PFBA (perfluorobutanoic acid, 0.017 µg/L) > PFNA (0.015 µg/L) > PFDA (0.012 µg/L) > PFHxS (0.0097 µg/L) > PFBS (0.0037 µg/L) > PFPeA (0.0033 µg/L).

Marine water data in the database is much more limited, with only three sample points (Appendix Table A.9). PFOA and PFOS were detected at the highest concentration with 90th percentile concentration of 0.004 and 0.0057 µg/L, respectively. These concentrations were followed by PFHxA (90th percentile concentration = 0.0033 µg/L) > PFPeA (0.002 µg/L) > PFDA (0.0017 µg/L) > PFHpA (0.0014 µg/L) > PFNA (0.0009 µg/L) > PFOSA (0.0007 µg/L) > and N-EtFOSAA (0.0003 µg/L).

Very limited sediment data are available in the database, which includes only records for estuarine systems, not freshwater and marine ecosystems (Appendix Table B.9). Sediment from estuarine systems have been monitored for 16 PFAS, with PFOS being the most frequently detected with a detection frequency of 69%, and a 90th percentile value of 2 µg/kg dw. And PFNA, PFDA, PFUnA (perfluoroundecanoic acid), PFDoA (perfluorododecanoic acid), PFOA, and PFOSA had comparable concentrations at between 0.01–0.2 µg/kg dw. Two short-chain PFAS, PFHxA and PFBS, had 90th percentile values that were based on the median detection limits of 0.099 and 0.2 µg/kg dw, respectively. PFAS precursors have also been detected in estuarine sediment, as referenced in the literature. Detected concentrations in San Francisco Bay sediment were highest for 6:6 PFPi (8.75 µg/kg dw) > N-EtFOSAA (2.57 µg/kg dw) > N-MeFOSAA (0.773 µg/kg dw) > FOSAA (0.254 µg/kg dw) (M. Sedlak et al., 2018). The phosphate ether class of precursors (e.g., di-PAPs) were largely not detected in Bay-wide samples collected in 2014. Di-PAPs were detected in 2012 (M. D. Sedlak et al., 2017); however, these precursors were measured using a method still in development, and as such it is likely that the findings from the finalized method used in 2014 reflect actual conditions in the Bay (Sedlak et al., 2018). The perfluoroalkyl phosphinates were also largely not detected.

Fish tissue data summarized in the databases shows detections of seven perfluoroalkyl acids in estuarine and freshwater fish (Appendix Table C.9). The database does not include data for marine fish. Fish tissue concentrations were greater for estuarine fish compared to freshwater fish. Comparison of the 90th percentile values in estuarine fish tissue indicate highest concentrations of PFOS in fish tissue of 29 µg/kg ww, followed by PFOSA (2.9 µg/kg ww) > PFDA (1.9 µg/kg ww) > PFNA (1.6 µg/kg ww) > PFDoA (1.3 µg/kg ww) > PFOA (0.77 µg/kg ww) > PFUnA (0.67 µg/kg ww). The database includes a small number of records for estuarine and marine bivalves; all measurements were below detection limits, which were generally in the low µg/kg level.

A more recent investigation of sport fish tissue samples collected from San Francisco Bay in 2019 examined 32 PFAS including a range of perfluoroalkyl carboxylates, perfluoroalkyl sulfonates, fluorotelomer sulfonates, fluorotelomer carboxylates, perfluorooctane sulfonamides, perfluorooctane sulfonamide ethanols, per- and polyfluoroether carboxylates, and ether sulfonates (Buzby et al., 2021). Analytical improvements associated with this newer method included detection limits as low as 0.2 ppb (µg/kg). As in the database, PFOS was the dominant PFAS (maximum concentration 10 µg/kg ww), followed by PFOSA (1.7 µg/kg ww) and long-chain perfluorocarboxylates like PFDA (1.7 µg/kg ww), PFDoA (1.5 µg/kg ww) and PFTeDA (perfluorotetradecanoic acid, 0.79 µg/kg ww). In general, short-chain perfluoroalkyl substances (e.g., PFBA and PFHxS), PFOA, and most precursors were rarely detected, and most reported concentrations were close to detection limits. The 2019 data appear to be some of the first fish tissue monitoring data in species consumed by humans for newer PFAS such as GenX and ADONA in the US, though neither were detected.

Another key literature study provides additional occurrence data for bivalves in marine waters. Bivalves from 68 stations from along California's coast were analyzed for bioaccumulating emerging contaminants as part of the National Oceanic and Atmospheric Administration's National Status and Trends Mussel Watch Program. This study found highest concentrations of PFDoA (maximum 29 µg/kg dw), followed by PFOS (5.5 µg/kg dw) > PFOSA (2.9 µg/kg dw) > PFUnA (2.8 µg/kg dw) > PFDA (1.2 ug/kw dw) (Dodder et al., 2014).

3.9.3. Risk screening

Exposure to perfluoroalkyl acids has been associated with a wide range of toxic effects, with most studies focused on PFOA and PFOS. Studies based on laboratory animals and human populations exposed to higher concentrations provide evidence that well-studied PFAS such as PFOS and PFOA can be classified as multi-system toxicants and developmental toxicants (DeWitt, 2015). Studies across species and chemicals suggest that PFOA and PFOS tend to cause liver damage, adverse developmental effects, and suppression of the immune system, and these effects may occur at exposure to environmentally relevant concentrations (Lau et al., 2007; Tahziz et al., 2021; Z. Wang et al., 2017).

The San Francisco Bay Regional Water Quality Control Board Interim Final Environmental Screening Levels (ESLs) for Aquatic Habitat Ecotoxicity in fresh and marine waters are 0.075 µg/L for PFOS and 4.4 µg/L for PFOA (SFBRWQCB, 2020). ESLs have not yet been derived for other PFAS. The occurrence database indicates that where data are available, state water concentrations of PFOS and PFOA are generally lower than their ESLs. The only other PFAS with water concentrations exceeding thresholds is PFOSA, for which one 90th percentile concentration each in freshwater, estuarine water, and marine water exceeded the European environmental quality standard of 0.00065 µg/L (EU Directive 2013/39/EU) with RQs of 2.0, 1.5, and 1.1, respectively.

Limited toxicity data exist for PFAS in sediment. Casado-Martinez et al. (2021) derived sediment quality guidelines for PFOS using the limited available toxicity testing data. Sardiña et al. (2019) used soil thresholds from Australia and New Zealand (Heads of EPAs Australia and New Zealand (HEPA), 2020) for sediment hazard screening and risk quotient calculations. Using their approach, comparing occurrence database concentrations of PFHxS with the HEPA threshold of 0.02 µg/kg dw, one study of marine sediment indicates High Concern, with an RQ of 10. In contrast, the thresholds for PFOS (13.5 µg/kg dw) and PFNA (700 µg/kg dw) were not exceeded (maximum RQs of 0.15 and 0.0002, respectively). Sediment toxicity thresholds have not been developed for detected PFAS precursors in sediment.

In humans, PFAS have been associated with a variety of toxic effects, including carcinogenicity and abnormal development. Infants, children, and pregnant/nursing people are considered to be at higher risk, as these compounds can cross the placental barrier and concentrate in breast milk. The USEPA has established a human health fish tissue benchmark for PFOS, the most commonly detected PFAS, of 68 µg/kg ww (USEPA, 2020b), but many state health departments have lower guidelines. No human health or regulatory thresholds have yet been established for PFAS in California fish. In other states where sampling for PFAS in fish has been more extensive, such as Minnesota and Michigan, concentrations have been high enough that the states have established their own thresholds for issuing consumption guidelines.

Currently, at least nine states have fish consumption guidelines for PFOS, and several of these states also have thresholds for other PFAS including PFOA, PFNA, and PFBS (Longsworth, 2021). Federal and state thresholds vary because they may include not only an analysis of risk from the contaminant, but often also a risk-benefit analysis balancing toxicity of the contaminant with the known benefits of consuming fish. Thresholds may also vary because they evaluate different studies and endpoints or use different factors and assumptions (e.g., body weight and consumption rate).

The lowest state thresholds for PFAS are currently in Massachusetts. To be protective, we have chosen to use these thresholds for this risk screening. The Massachusetts Department of Public Health recommends a candidate Fish Action Level (cFAL) of 0.22 µg/kg ww. This value is meant to be applied individually to fish tissue concentrations of PFBS, PFHxS, PFOA, PFOS, and PFNA. A waterbody-specific analysis to develop fish consumption advisories is required if any of these compounds individually exceed the cFAL. Concentrations below 0.22 µg/kg ww for any of these compounds is considered protective for unlimited consumption of fish for both the general population and sensitive populations. All individual compound 90th percentile concentrations for both freshwater and estuarine fish exceed this cFAL, with RQs ranging from 2.3 to 130, placing PFAS in the Very High Concern tier of our risk-based framework based on human health concerns.

Risk assessment of PFAS is increasingly being done as a sum of all PFAS instead of evaluating each compound individually. The high persistence, accumulation potential, and hazards (known and potential) of PFAS studied to date warrant treating all PFAS as a single class (Kwiatkowski et al., 2020). The most precautionary grouping approach would be to group all PFAS based on their high persistence alone, while a less precautionary grouping approach for risk assessment would be to group only the PFAS that have the same toxicological effects, mechanisms of action, and elimination kinetics (Cousins et al., 2020). However, toxicological effects, modes of action, and elimination kinetics are not well documented for the majority of compounds that make up the PFAS class. In California, the Department of Toxic Substances Control (DTSC) has decided to regulate PFAS as a class in certain consumer products because of their common hazard trait of persistence (Bălan et al., 2021). Thresholds for the sum of PFAS are not yet available for water and sediment.

The European Food Safety Authority (EFSA) has established a tolerable weekly intake (TWI) of 4.4 ng/kg body weight for the sum of PFAS (using data for PFOA, PFNA, PFHxS, and PFOS; EFSA (Panel on Contaminants in the Food Chain (CONTAM Panel), 2020)). This threshold includes all exposure from foods and drinking water, so it is difficult to calculate back to a value specific to fish consumption. However, recent work in Sweden translated this threshold to a one fish serving per week threshold of 3.4 µg/kg ww (Augustsson et al., 2021). Several of the individual compound 90th percentile concentrations exceed this threshold, indicating the sum of PFAS would exceed the cumulative threshold, further supporting our classification of PFAS as Very High Concern based on human health risks.

4. Expanding CECs Monitoring and Risk Screening in California Using New Tools and Approaches

The risk screening exercise demonstrated in Section 3 necessarily focused on several contaminant classes previously identified as of potential concern, and for which California occurrence data are available. This group of compounds represents a small subset of the thousands of chemicals in commerce; many more have yet to be examined. The USEPA's inventory of industrial chemicals now contains over 86,000 substances, of which over 42,000 are considered in active use (USEPA, 2021b); thousands more compounds are in use in the US as pharmaceuticals, pesticides, food additives, and personal care product ingredients. Additional chemicals can enter the US as part of imported goods and articles, or via environmental transport (Z. Wang et al., 2020). A further source of unique chemical contaminants are the transformation products that may form as chemicals in commerce enter the environment (Chibwe et al., 2017; Howard & Muir, 2013).

Given the expanding understanding of the occurrence and impacts of emerging contaminants, along with the rapid pace of chemical and product development globally, it is important to track broader efforts to identify and analyze new CECs and assess risks to wildlife and humans. Through the Water Boards' statewide CEC Program, these agencies are well-positioned to maintain a forward-looking approach to address emerging contaminants more efficiently and effectively. Below, we outline some of the new tools and approaches that can inform future monitoring and management priorities.

4.1. Advancing targeted analytical techniques

Starting in the 1990s, advances in targeted analytical chemistry brought to light a new and wide-ranging set of contaminants with the potential to impact ecosystems worldwide. Major improvements in analytical instruments allowed chemists to identify compounds at µg/L to ng/L levels, providing an unprecedented ability to detect unregulated and previously unknown contaminants in environmental matrices. These breakthroughs led to an upwelling of investigations into CECs in California and beyond. Challenges specific to these types of targeted analyses include the need to quantify low concentrations of analytes with potentially broad physico-chemical properties, which are present within complex matrices (Hernández et al., 2019). Analytical methods must have detection limits below toxicological thresholds to inform risk screening.

The analysis of environmental samples for emerging contaminants like those discussed in this report generally relies on chromatography-based techniques, specifically gas chromatography (GC) and liquid chromatography (LC), combined with mass spectrometry (MS). Chromatography-based techniques separate a mixture of compounds using two phases, known as the stationary and mobile phase. Different physical and chemical properties (such as size, shape, mass, boiling point, polarity, etc.) will cause compounds to interact differently with the two phases, leading to separation.

In gas chromatography (GC), volatile compounds are separated as an inert gas mobile phase passes through the stationary phase. Varying volatility and boiling point lead to different compound affinities for the mobile and stationary phases, resulting in separation of compounds as indicated by a chromatogram with peaks associated with elution from the stationary phase column (i.e., exited with the mobile phase). In a system combined with mass spectrometry (MS), these compounds are ionized and fragmented within an ionization chamber, then accelerated to a mass analyzer to detect and separate fragments using their mass-to-charge (m/z) ratio. Detection of ionized fragments allows sensitive characterization of the components of a complex mixture. GC-MS is advantageous for its speed and separation efficiency, and works best for nonpolar, volatile pollutants, and those with high thermal stability (Galindo-Miranda et al., 2019). Recent advances include use of two-dimensional GC (GC x GC), in which two GC columns are sequentially connected, enhancing the peak capacity and separation power of the instrument; and high resolution MS (HRMS) and tandem MS (MS/MS), which provide more sensitivity and versatility to the techniques.

In contrast to GC, liquid chromatography (LC) uses a liquid mobile phase to separate contaminants for identification. Most commonly, varying polarities of mobile and stationary phases are used to separate mixtures of contaminants (Daverey et al., 2019). This allows for a wide range of compounds to be detected, though this can mean these techniques require more time for analysis. Analytes of medium to high polarity are more compatible with LC-based analysis. A review of the literature suggests LC-MS/MS techniques are widely used for quantification of CECs, resulting from an increasing focus on polar compounds within classes like pharmaceuticals, current-use pesticides, bisphenols, and organophosphate esters, among others.

However, many more CECs have been identified and quantified via more exploratory targeted analyses; for these compounds, standardized methods are not yet available. Because it can take many years for standard methods for specific compounds to be developed and commercialized, it is important to consider measurements derived from non-standard methods to be able to track newly identified contaminants, including those representing regrettable substitutions in products. New methodological advances can also lead to lower detection limits and, therefore, higher detection frequencies for contaminants where existing standard methods are insufficient to assess environmental occurrence at toxicologically relevant levels. Recent monitoring study guidance suggests a maximum method reporting limit generally set to half the concentrations indicated by available thresholds of interest, such as monitoring trigger levels (Dodder et al., 2015). While there may be uncertainties associated with these more exploratory methods, an exclusive reliance on data from standardized methods would limit attention to relatively information-rich CECs and exclude action on newer concerns.

Below we list some examples among the screened classes (Section 3) where data limitations suggest analytical improvements are needed, including development and/or application of targeted analytical methods that have not yet been standardized.

Alkylphenols and alkylphenol ethoxylates – For APs and APEOs included in the occurrence database, better analytical methods are needed to quantify compounds at environmental concentrations (Section 3.1). In addition, available data indicate the need for extra precautions during sample collection to reduce the introduction of procedural contamination.

In addition, alkylphenol ethoxylates used in products generally contain ethoxylate chains between 4 and 15, compounds that cannot be characterized using available standardized methods. While a preliminary investigation of these long-chain APEOs is being completed in the San Francisco Bay Area using more exploratory methods (Lindborg et al., in prep), a major impediment to standardization is the lack of commercially available analytical standards. Additionally, alkylphenol carboxylates are the dominant degradation product for APEOs under the aerobic conditions expected in surface waters, and have also not been monitored. Alkylphenol ethoxylates are increasingly being replaced by other ethoxylated surfactants with limited monitoring data globally. DTSC is seeking to gather occurrence data on long-chain APEOs, alcohol ethoxysulfates, secondary alkane sulfonates, alcohol ethoxylates, and alkylbenzene sulfonates as part of the Priority Product review of NPEOs in industrial detergent (DTSC, 2018). Analytical methods are needed to measure these additional ethoxylate compounds.

Organophosphate esters – Testing of consumer products containing flame retardants has highlighted active use of additional organophosphate esters not commonly monitored in the environment (A. L. Phillips et al., 2017). Such findings have motivated development and application of more expansive monitoring methods including a larger variety of organophosphate ester analytes than currently examined in California matrices (Guan et al., 2019). In addition, researchers have developed a functional group-dependent screening strategy for identifying unanticipated organophosphate esters, and applied it to samples of sediment from Taihu Lake, China (Ye et al., 2021). Some of the organophosphate esters quantified via these methods are used as flame retardants or for other purposes in plastics (Wiesinger et al., 2021); the uses of others are unclear. Another monitoring data gap is characterization of key diester metabolites, such as bis(1,3-dichloro-2-propyl) phosphate, a metabolite of TDCIPP.

Brominated flame retardants – Like PBDEs, many of the brominated compounds used at high volumes globally are flame retardants. Additional brominated flame retardants (including chemical intermediates) that have not been the subject of targeted monitoring in California and may merit examination include tetrabromobisphenol A (TBBPA), 1,2,3,4,5-pentabromo-6-chlorocyclohexane (PBCC), N,N’-ethylenebis (tetrabromophthalimide) (EBTEBPI), octabromo-1,3,3-trimethyl-3-phenylindan, tetrabromophthalic anhydride, dibromopropyl-tribromophenyl ether, 1,2,5,6-tetrabromocyclooctane, and 1-bromo-4-(tribromomethyl)benzene (Howard and Muir, 2010; Strempel et al., 2012; Zhang et al., 2020).

Pharmaceuticals and metabolites – While some data exist for dozens of pharmaceuticals in California aquatic ecosystems, many more may merit examination for potential risks to wildlife. Targeted studies of California wastewater effluent can provide information on the concentrations and estimated loads of pharmaceuticals discharged to receiving waters and not yet quantified via monitoring efforts (Lin, Sutton, Sun, et al., 2018). In addition, more exploratory suspect screening studies (see Section 4.2) of wastewater effluent suggest the potential presence of additional pharmaceuticals and metabolites. For example, a study of four San Francisco Bay Area effluents revealed a strong signal for the over-the-counter antihistamine fexofenadine (Overdahl et al., 2021). This compound is widely used and is known in the literature to be recalcitrant to wastewater treatment (Kosonen & Kronberg, 2009); no quantitative California occurrence data are available.

Pharmaceutical metabolites such as erythro-dihydrobupropion and hydroxybupropion (both derived from the antidepressant bupropion), as well as gabapentin-lactam (derived from the anticonvulsant gabapentin), were also observed in Bay Area effluent (Overdahl et al., 2021); these observations substantiate a need to consider metabolites as part of a comprehensive assessment of the potential impacts of pharmaceutical-derived contaminants.

An in silico screening provides a means to assess the persistence and bioaccumulation potential of over 3,000 pharmaceuticals included in US Food and Drug Administration databases and lists of top ranked or selling drugs (Howard & Muir, 2011). Among the compounds reviewed, 275 had been observed in the environment at that time, and 399 could be considered high production volume (HPV) pharmaceuticals. Compounds highlighted for future study included those that had not yet been examined in the environment and were classified as HPV as well as persistent and bioaccumulative (56 pharmaceuticals), or persistent only (48 pharmaceuticals). Just a handful of the identified pharmaceuticals have been examined in California monitoring exercises reported to publicly accessible databases.

PFAS – While more than 4,700 PFAS are registered for commercial use, standardized environmental monitoring methods are only available for a few dozen compounds (Y. Wang et al., 2019). The increasing diversity of fluorinated chemicals in commerce (Strempel et al., 2012) suggests significant monitoring and management actions are warranted for PFAS. Rapid development and commercialization of new fluorine chemistries likewise support a class-based approach to science and management, as embodied in the broadly inclusive PFAS definition from OECD (Wang et al., 2021), and class-wide PFAS bans in specific products.

To gain a more holistic picture of PFAS exposure in wildlife, an international research team performed a fluorine mass balance on liver tissues from 11 different marine mammal species using a combination of targeted PFAS analysis, extractable organic and total fluorine determination, and suspect screening (Spaan et al., 2019). For seven samples, including four from the US Atlantic Coast (no species from the US Pacific Coast were included), the sum of targeted PFAS analytes did not fully explain the presence of extractable organic fluorine. In fact, suspect screening revealed an additional 33 PFAS present in marine mammal liver samples. Nevertheless, PFOS was typically a dominant presence in these samples, similar to available tissue monitoring data in California.

The fluorine mass balance approach described above provides an example of the utility of methods designed to more broadly characterize contamination by PFAS, including the total oxidizable precursors (TOP) assay, and total organic fluorine methods, as described previously (Section 3.9). Such methods provide an indication of the overall quantity of PFAS present in the environment, information that is particularly useful in light of the rapidly expanding universe of as-yet unidentified PFAS in commerce and in the environment.

4.2. New approaches to prioritize emerging contaminants for monitoring

While the screening exercise demonstrated in Section 3 focused on several notable classes of CECs for which suitable levels of occurrence and toxicity data are available, there are far more constituents that could be important for future monitoring and management actions. To expand beyond classes of CECs with more available data, new tools and approaches can inform and prioritize additional research. Non-targeted analysis and suspect screening, a class of analytical techniques that seek broader occurrence information with respect to the presence of diverse compounds in a sample or matrix, can be used to identify unexpected contaminants of potential interest. Likewise, systematic in silico screening of chemicals in commerce based on factors including production, use, and physico-chemical and toxicological properties can provide new insights as to as-yet unassessed compounds with potential concerns. Both methods, described briefly below, can be used to identify additional compounds of interest for targeted monitoring and risk screening.

Suspect screening and non-targeted analysis – Recent analytical advances have led to the development of novel techniques applied to environmental matrices to identify unanticipated contaminants that cannot be observed using existing targeted methods. These techniques typically rely on high resolution mass spectrometry (HRMS; Section 4.1), or tandem HRMS, to perform two types of analysis: a) suspect screening, which involves matching unknown sample features to compounds within libraries of spectra; and b) non-targeted analysis, which involves determination of the chemical structures of unknown compounds that are not present in libraries (Newton et al., 2018). While identification of unknown compounds via suspect screening is relatively straightforward and reliable, it is necessarily limited by the compound-specific spectra available in existing spectral libraries. In contrast, non-targeted analysis often provides only partial information about an unknown compound, leading to more tentative identifications (Schymanski et al., 2014). One advantage of this class of techniques is the ability to conduct retrospective analysis of spectra collected previously, permitting identification of unknowns via matching to spectra of more recently characterized compounds (Angeles et al., 2020; Creusot et al., 2020).

Suspect screening and non-targeted analysis methods may use LC, best suited for identifying polar compounds, or GC, best suited for identifying nonpolar compounds, as noted previously. A recent study compared the identification rate of 1269 substances by LC- and GC-based methods and found moderate overlap (40%) in the number of compounds detected (E. M. Ulrich et al., 2019). For a more comprehensive screening for the presence of unanticipated contaminants, applying both LC- and GC-based methods may be useful. One limitation of current suspect screening and non-targeted analysis methods is that while they can indicate the presence of identified contaminants, they generally do not provide quantitative information, a key constraint in estimating exposure and risk (Panagopoulos Abrahamsson et al., 2020). Another limitation is that these methods may not be as sensitive as a targeted method for detection of specific compounds (A. Wang et al., 2018).

Forward-looking regional monitoring programs have begun to incorporate suspect screening and non-targeted analysis into monitoring activities to detect unexpected contaminants that may pose risks to wildlife (Anderson et al., 2012; Crimmins et al., 2018; Sutton et al., 2017). In California, these techniques have been applied to wastewater (B. Du, Tian, et al., 2020; Overdahl et al., 2021), recycled water (Ishida et al., 2022), streams (Mehinto et al., 2017), stormwater runoff (Chang et al., 2021; B. Du, Tian, et al., 2020; Moschet et al., 2017; Tian et al., 2021), estuarine water (Overdahl et al., 2021), bivalves (Sutton and Kucklick, 2015), seabirds (Millow et al., 2015), and marine mammals (Cossaboon et al., 2019; Mackintosh et al., 2016; Shaul et al., 2015; Sutton & Kucklick, 2015; Trego et al., 2018). Of particular interest, non-targeted approaches can be used to identify contaminants linked to adverse impacts, known as effect-directed analysis; such an approach was recently applied to samples of urban streams and stormwater collected along the Pacific Coast, leading to the identification of a tire-derived toxicant responsible for acute mortality in pre-spawn adult coho salmon (Tian et al., 2021). Strategic use of suspect screening and non-targeted analysis can provide valuable insights to inform future monitoring and management priorities for CECs.

In silico screening – A contrasting approach to developing priorities for future work is to screen large inventories of chemicals in commerce using computational or “in silico” tools that provide estimates of hazard traits. The USEPA and regulatory agencies of many other countries maintain chemical databases listing compounds approved for production and use. Those chemicals with known molecular structures can be screened using in silico models that rely on quantitative structure property or activity relationships (QSPR/QSARs) to estimate persistence, bioaccumulation, toxicity, and/or long-range transport potential, enabling identification of those that could be prioritized for environmental monitoring.

Notable examples of in silico screening include those focused on identifying hazardous compounds approved for commercial use in the US and Canada (Howard & Muir, 2010), the EU (Strempel et al., 2012), and China (X. Zhang et al., 2020). Some screening efforts are moving towards a more international scope, combining databases of chemicals in commerce from many nations, given the potential for global trade and environmental transport to carry chemicals across national boundaries (Muir et al., 2019; Z. Wang et al., 2020). Review of these studies indicates a number of specific compounds within chemical or functional (use-based) classes that are discussed in the next sections.

Limitations of in silico screening approaches relate to the comprehensiveness of the chemical databases, as well as the ability of available computational tools to accurately predict hazard traits for a broad range of chemical structures. The types of chemicals included within available databases can be limited by use type, production volume, or confidential business information. To address the specific data gap of pharmaceuticals, Howard and Muir (2011) developed a pharmaceuticals database from two US FDA databases and supplemented with additional lists of widely used drugs, which they then screened for persistence and bioaccumulation using QSPRs and professional judgment. In addition, many entries within the regulatory databases are not discrete chemicals, but instead are “unknown or variable composition complex reaction products and biological materials” (UVCBs). Complex mixtures like these are challenging to assess without specific information on the components present, and are typically excluded from screenings. A recent, global screening study noted that chemical inventories are not available in countries in Africa, South America, and West Asia; this study found an unexpectedly high number of chemicals registered only in developing countries for which inventories were available, indicating that studies focusing on chemicals registered in developed countries are not sufficiently comprehensive for identification and prioritization of potential environmental contaminants (Z. Wang et al., 2020). A final constraint on use of regulatory databases for in silico screening is that they do not include unintentionally produced chemicals such as byproducts, impurities, and transformation products. An initial attempt to address this data gap (Howard & Muir, 2013) identified and evaluated possible byproducts, impurities, and transformation products related to a refined list of over 600 chemicals previously identified through a prior screening study as likely to be persistent and bioaccumulative (Howard and Muir, 2010).

The computational tools used to predict hazard traits pose additional constraints on in silico screening approaches. Model-derived predictions of persistence, bioaccumulation, toxicity, and/or long-range transport potential are subject to uncertainty relating to the availability and quality of input data for individual chemicals, as well as other sources. An analysis by Strempel et al. (2012) suggests that uncertainty in persistence data contributes most to the uncertainty in the number of potentially persistent, bioaccumulative, and toxic (PBT) chemicals identified within EU inventories of chemicals in commerce. In addition, these predictive QSPR/QSAR tools were developed and trained using well-characterized substances that are likely not representative of the full suite of compounds present in chemical inventories (Muir et al., 2019). As a result, they may be biased towards prioritizing compounds that are similar in structure to more established chemicals of concern. Chemical classes with unique physico-chemical properties, such as PFAS, are generally not adequately modeled using these tools (Muir et al., 2019). Overall, in silico screening studies have focused on neutrally charged organic compounds, including neutral forms of acidic or basic molecules, and have excluded inorganic and organometallic compounds, as well as many polymers and surfactants.

Despite these limitations, in silico screening studies have indicated a number of chemical or functional (use-based) classes that may be appropriate to prioritize in future environmental monitoring activities. Complementary use of suspect screening and non-targeted analysis can further hone efforts to identify chemical structures associated with hazard traits like persistence and bioaccumulation, which can inform both monitoring and management (Zhang et al., 2019).

Use of the analytical and computational screening approaches described above has resulted in identification of a broad range of chemicals that may merit quantitative monitoring in environmental matrices. Listed below are classes of CECs, defined by shared chemical and/or functional properties, that have been highlighted by recent studies, including brominated compounds (beyond flame retardants); chlorinated compounds; siloxanes; plastic monomers, additives, and processing aides; vehicle and tire-related compounds; and quaternary ammonium compounds (QACs). Based on available data, these may be of potential interest for future monitoring in California.

4.2.1. Brominated compounds (beyond flame retardants)

Multiple in silico screening methods have looked beyond the well-established concerns with brominated flame retardants, identifying other brominated aromatic or cyclic compounds as persistent, bioaccumulative, and potentially toxic (Howard and Muir, 2010; Strempel et al., 2012; Zhang et al., 2020). Uses include chemical intermediates for the production of flame retardants and liquid crystal or electroluminescent displays (Howard and Muir, 2010; Strempel et al., 2012; Zhang et al., 2020).

Additional brominated compounds have been examined in various aquatic matrices via targeted analysis. A class of bromine- and chlorine-containing compounds, polyhalogenated carbazoles, has also been examined in San Francisco Bay sediment and biota (Y. Wu et al., 2017). These compounds may be derived from natural sources, and/or result from impurities in the final products of halogenated indigo dye formulations. Polyhalogenated carbazoles appear to have dioxin-like activity; a comparison of estimated toxic equivalence suggests current levels pose less concern than polychlorinated biphenyls (PCBs) (Wu et al., 2017); however, cumulative risk due to coexposure to both carbazoles and PCBs was not assessed.

In addition to targeted monitoring, non-targeted analysis and suspect screening studies of blubber samples from multiple marine mammal species along the coast of California have identified PBDEs, PBBs, mixed brominated/chlorinated diphenyl ethers (B/CDEs), polybrominated methoxy PBDEs and B/CDEs, methoxy PBBs, polybrominated dibenzofurans, brominated and mixed dimethyl and methyl bipyrroles, bromo methyl biphenyl, brominated indoles, and polybrominated hexahydroxanthene derivatives

(Shaul et al., 2015; Trego et al., 2018; Cossaboon et al., 2019). Additional brominated compounds observed in both bird eggs (California black skimmer (*Rynchops niger*); Millow et al., 2015) and marine mammal blubber (Shaul et al., 2015; Trego et al., 2018; Cossaboon et al., 2019) include 4-bromophenol and brominated anisoles. The bird eggs were also found to contain 2-bromo-1,3-diphenyl-1,3-propanedione (Millow et al., 2015). While PBDEs and PBBs are anthropogenic, the other brominated compounds detected have mixed, natural, or unknown sources (Shaul et al., 2015; Trego et al., 2018; Cossaboon et al., 2019), which is a likely factor in why they were not identified via in silico screenings of chemical inventories. Quantitative environmental monitoring and toxicity data are limited for the majority of these compounds.

Additional brominated compounds recommended for monitoring via in silico screening have not been the subject of targeted monitoring in California. These include compounds used to manufacture liquid crystal and electro-photosensitive materials (Su et al., 2019; X. Zhang et al., 2020); while direct emissions of these compounds to the environment may not occur, they could be present as residual impurities in products and emitted during use and from the waste stream.

An in silico screening comparison of a 95,000-compound chemical inventory to a smaller a set of 2,781 chemicals from the EU's former list of "new chemicals" that were introduced on the European market after 1981 (European List of Notified Chemical Substances, ELINCS) revealed notable trends in chemical composition (Strempel et al., 2012). Specifically, among the potential PBT chemicals identified in the newer ELINCS compounds, there is a shift from brominated (and chlorinated) substances toward fluorinated substances. Brominated compounds make up 9.1% of potential PBT chemicals in the full chemical inventory, while only 1.4% of potential PBT chemicals in the ELINCS inventory. In contrast, 29% of potential PBTs in the ELINCS list contain (non-PFAS) fluorine, compared to just 14% in the full list. This finding suggests that a broad emphasis on brominated compounds as part of California's CEC Program may not be supported at this time.

4.2.2. Chlorinated compounds

As with brominated compounds, multiple in silico screening methods have looked beyond legacy chlorinated contaminants such as PCBs and DDT, identifying other chlorinated aromatic or cyclic compounds as persistent, bioaccumulative, and potentially toxic (Howard and Muir, 2010; Strempel et al., 2012; Zhang et al., 2020). The compounds identified have diverse uses including as flame retardants, antimicrobial active ingredients, plastic ingredients, and chemical intermediates; in some cases, uses are unknown (Howard and Muir, 2010; Zhang et al., 2020).

Among the top four chlorinated compounds recommended for monitoring by Howard and Muir (2010), two have been the subject of targeted monitoring in the State of California: the flame retardant Dechlorane Plus, and the antimicrobial active ingredient triclocarban. Dechlorane Plus has been monitored in sediment and biota in San Francisco Bay (Sutton, Chen, et al., 2019), with frequent, low level (< 2 µg/kg dw) detections in sediment, and less frequent but similarly low detections in bivalves and harbor seal blubber. Dechlorane Plus is not a member of the CEC classes reviewed in this risk screening exercise. In contrast, triclocarban is a member of the personal care and cleaning products class, and has been included in several monitoring efforts (Section 3.8). Of note, triclosan was excluded from the Howard and Muir (2010) priorities due to its frequent inclusion in CECs monitoring efforts, consistent with the data availability observed in California (see Section 3.8).

Beyond these priorities, additional chlorinated compounds have been the subject of targeted monitoring in California. Polyhalogenated carbazoles, mentioned above, have been examined in San Francisco Bay sediment and biota (Wu et al., 2017). Comparison of the chlorinated carbazole levels in fish and harbor seals suggested biomagnification of these compounds. Concentrations of polyhalogenated carbazoles were comparable to those of PCBs in the sediment of the Bay, but two to three orders of magnitude lower in biota. As noted above, estimated risks of these compounds appears to be lower than that of PCBs, though cumulative risks associated with exposure to both classes have not been evaluated.

While PCBs, referenced above, are generally considered a legacy contaminant class primarily derived from Aroclor mixtures, some specific PCB congeners have sources associated with other products in current use. It may be appropriate to consider these PCB congeners as emerging contaminants. One example is PCB-11, also known as 3,3'-dichlorobiphenyl, which is inadvertently manufactured in the production of pigments used for inks, dyes, paints, and textiles (Rodenburg et al., 2010). PCB-11 is semi-volatile and can leach out of consumer products and enter wastewater effluent and stormwater flows into receiving waters. While monitoring data for PCB-11 and other PCBs with non-Aroclor sources (PCB-47, 51, 68, 206, 207, 208, 209) were not compiled as part of this risk screening exercise, it is likely that considerable data for California exist wherever PCB analysis of the full 209 congeners has been performed. The toxicity information associated with non-Aroclor PCBs is variable; for example, PCB-11 does not bioaccumulate in the food web, and does not exhibit dioxin-like toxicity because of its lower number of chlorines. As a result, congener-specific risk screening approaches may be appropriate.

Chlorinated paraffins are an additional class of contaminants not reviewed in this risk screening, and for which few California data are available. They are primarily used as lubricants and coolants in the metal forming and cutting industries, and plasticizers and flame retardants in plastics. Minor uses include paints, rubber formulation, adhesives and sealants. Chlorinated paraffins are considered persistent, bioaccumulative, and toxic. Short-chain (C10-C13) chlorinated paraffins were characterized in San Francisco Bay biota in 2006-2007 (S. Klosterhaus et al., 2013). Seal blubber contained the highest total concentrations (25-50 ng/g wet weight [ww]), followed by cormorant eggs (4–6 µg/kg ww), and then sport fish (<1–1 µg/kg ww). Medium- and long-chain chlorinated paraffins (C14-C17 and C≥18, respectively) have not been evaluated. Short-chain chlorinated paraffin production in the US stopped in 2012 as part of a settlement negotiated with USEPA (USEPA, 2012a). As global phase-out of short-chain chlorinated paraffins occurs, the use of medium- and long-chain alternatives is expected to increase. Recent monitoring studies in other locations report detections of a broad range of chlorinated paraffins in sediment and biota (X. Du et al., 2018; Yuan et al., 2019; Zeng et al., 2015; Zeng, Lam, Chen, et al., 2017; Zeng, Lam, Horii, et al., 2017).

In addition to targeted monitoring, non-targeted analysis and suspect screening studies of bird eggs and marine mammal blubber samples from the California coast have identified multiple classes of chlorinated aromatic or cyclic compounds. Of particular note is the occurrence of known or suspected impurities and transformation products of the legacy pesticide DDT (Millow et al., 2015; Shaul et al., 2015; Sutton and Kucklick, 2015; Mackintosh et al., 2016; Trego et al., 2018; Cossaboon et al., 2019). While the geographic ranges of each species examined may be quite broad, many of the samples were collected in southern California, a region that includes coastal sites where significant quantities of DDT waste barrels have been observed on the ocean floor (Xia, 2021). While DDT is not considered an emerging contaminant, the presence of unanticipated DDT-derived impurities and transformation products emphasizes the potential significance of these types of compounds in environmental monitoring and risk evaluation (Howard and Muir, 2013).

Additional anthropogenic chlorinated compounds identified via non-targeted analysis and suspect screening of tissue samples from California include compounds derived from other legacy pesticides as well as PCBs, and polychlorinated terphenyls (Shaul et al., 2015; Sutton and Kucklick, 2015; Trego et al., 2018; Cossaboon et al.. 2019). Methyl triclosan, a transformation product of triclosan, was identified via suspect screening in bivalves from San Francisco Bay (Sutton and Kucklick, 2015); both triclosan and methyl triclosan were later quantified in prey fish from the southernmost portion of this Bay (Lin et al., *in prep*.). Dichlorinated PAHs, produced during combustion, were present in tissue samples and may have both natural and anthropogenic sources (Shaul et al., 2015; Sutton and Kucklick, 2015; Trego et al., 2018; Cossaboon et al., 2019). Other chlorinated compounds, such as chlorinated and mixed halogenated dimethyl and methyl bipyrroles, and methoxy chlorinated diphenyl ethers, have natural or unknown sources (Shaul et al., 2015; Trego et al., 2018; Cossaboon et al., 2019). An additional class of contaminants observed in fish tissues from the Great Lakes, which could merit examination in California, are chlorinated methoxyphenols; the sources of these compounds are unknown (Fernando et al., 2018). Quantitative environmental monitoring and toxicity data are limited for the majority of these compounds.

In silico screening studies have identified other chlorinated compounds of interest that have not been examined in California matrices by targeted or non-targeted methods. Howard and Muir (2010) suggested the need for data on bis(4-chlorophenyl)sulfone, used in the production of thermostable plastics; dibutyl chlorendate, a plasticizer and flame retardant; and multiple cyclopentane-based compounds, used as chemical intermediates in the production of Dechlorane Plus and other compounds. Zhang et al.

(2020) identified additional chemical intermediates, such as 5-chloro-2-methyl-3-nitrobenzotrifluoride and 2,4-dichloro-N-(1,2-dichloroethyliden)aniline; 4,4’-(1,1-dioxido-3H-2,1-benzoxathiazole-3,3-diyl)bis(2,6-dichlorophenol), used in lithography and opto-chemical sensors and as a chemical reagent in chlorophenol blue dye; and 4,4’-dichlorophenyl sulfide, used in lubricants, flame retardants, pesticides, and pharmaceuticals. Other compounds identified had no known uses (3,3’,4,4’,-tetrachloro-azobenzene; bis(3,4-dichlorophenyl)-diazene-1-oxide; 1,2,3,4,7,7-hexachloro-5-phenyl-bicyclo[2.2.1]hepta-2,5-diene). Several mixed halogenated (brominated and chlorinated) compounds were mentioned in the previous section on brominated compounds.

As noted previously, an in silico screening of ELINCS, the EU’s former list of “new chemicals,” revealed a shift from chlorinated (and brominated) substances toward fluorinated substances. (Strempel et al., 2012). Specifically, chlorinated compounds make up 31% of potential PBT chemicals in the full EU chemical inventory, but just 13% of potential PBT chemicals in the newer chemicals described in the ELINCS inventory. This decline suggests a potential shift in new chemical development away from chlorinated chemistries, though the shift is not as dramatic as that observed from brominated chemistries.

4.2.3. Siloxanes

An organosilicon-based chemical class defined by the presence of Si−O−Si functional groups, siloxanes (or silicones) have wide-ranging uses in personal care and cleaning products, pharmaceuticals, medical products, cooking items, plastics, paper, paints, sealants, adhesives, building materials, electronics, and in industrial applications including dry cleaning (Rücker & Kümmerer, 2015). Specific to dry cleaning, increased use of a volatile cyclic methylsiloxane, decamethylcyclopentasiloxane (D5), may have occurred in California due to current use as an alternative to perchloroethylene (Ceballos et al., 2021), the subject of a major emissions control effort in the state.

Widely used linear and cyclic siloxanes have been observed in environmental matrices worldwide (Rücker and Kümmerer, 2015). Targeted data in California may be limited to a summary of monitoring data reported in bivalves collected from San Francisco Bay (Lin, Sutton, Shimabuku, et al., 2018). However, broader suspect screening studies of the state’s aquatic ecosystems indicated the presence of D5 in estuarine waters (Overdahl et al., 2021). Multiple in silico screening studies have prioritized members of this class of compounds based on factors including high production volume and estimated persistence and bioaccumulation or bioconcentration, particularly for cyclic methylsiloxanes (Howard and Muir, 2010; Muir et al., 2019; Zhang et al., 2020). Experiments to characterize bioaccumulation and bioconcentration of volatile cyclic methylsiloxanes in the environment have resulted in conflicting evidence, which may in part be due to the difficulty of quantifying these compounds in water (Rücker and Kümmerer, 2015).

4.2.4. Plastic monomers, additives, and processing aides

Globally, annual plastic production is anticipated to increase from nearly 350 million tons in 2017, to over 2 billion tons in 2050, driving increasing production and use of plastic monomers, additives, and processing aides (UN (United Nations), 2019). A recent systematic investigation of plastic ingredients in use worldwide identified over 10,000 chemical substances (Wiesinger et al., 2021). More than 2,400 of these compounds were suggested to be of potential concern due to persistence, bioaccumulation, and/or toxicity. Of particular relevance to the present risk screening exercise, over 1,600 substances were flagged for chronic aquatic toxicity. While this investigation of plastic monomers, additives, and processing aides is the most comprehensive to date, the number of substances identified is likely to be an underestimate of the total number of substances in plastics due to lack of transparency concerning plastic ingredients, difficulty in aggregating all publicly available data, as well as the presence of non-intentionally added substances (e.g., impurities) in plastics.

Many of the classes examined in this statewide risk screening exercise are widely used in plastics, including alkylphenol ethoxylates, bisphenols, organophosphate esters, and phthalates. Non-targeted analysis and suspect screening approaches can facilitate identification of additional members of these classes not yet covered by either standardized or more exploratory targeted methods.

Broader suspect screening efforts applied to California matrices have identified the presence of additional plastic-related compounds outside of these classes. For example, the plasticizer N-butylbenzenesulfonamide was observed in roadway runoff from southern California (B. Du, Tian, et al., 2020), and additives such as bis(2,2,6,6-tetramethyl-4-piperidyl) sebacate (also known as Tinuvin 770) and didodecyl 3,3’-thiodipropionate oxide were identified in San Francisco Bay water (Overdahl et al., 2021). The presence of a variety of plastic ingredients in California’s aquatic ecosystems is consistent with observations of significant concentrations of microplastics in San Francisco Bay (Sutton, Lin, et al., 2019; X. Zhu et al., 2021), as these microplastics can serve as carriers of plastic contaminants into the environment. Ecotoxicity information is scarce for many plastic ingredients, limiting the understanding of risks posed by these chemicals. Given the widespread and increasing use of plastics locally and globally, additional monitoring and toxicity evaluations of plastic-related chemicals and classes would be valuable to inform risk screening and management in California.

4.2.5. Vehicle and tire-related contaminants

The recent effects-directed non-targeted analysis identification of a tire-derived contaminant, 6PPDQ (N-(1,3-dimethylbutyl)-N′-phenyl-p-phenylenediamine quinone), which causes acute mortality to coho salmon (Tian et al., 2021), highlights the need to expand monitoring activities associated with chemicals related to tires and other vehicle components and fluids. Contaminants specific to these sources have not been a major focus of emerging contaminants research previously, resulting in a data gap highly relevant to urban-influenced aquatic ecosystems.

Tire-related contaminants with potential water quality impacts have been identified in California matrices. 6PPDQ has been observed in stormwater in the San Francisco Bay Area, and roadway runoff in Los Angeles (Tian et al., 2021). DTSC has highlighted six other tire ingredients of interest to the Safer Consumer Products Program (DTSC, 2021). One class, octylphenol ethoxylates, was included in the risk screening exercise (Section 3.1). The other five chemicals or classes include: benzothiazoles; chlorinated paraffins (see Section 4.2.2); 1,3-diphenylguanidine; (methoxymethyl) melamines; and polycyclic aromatic hydrocarbons. A limited number of studies employing suspect screening and/or targeted analyses, some still in progress, have identified these compounds in urban runoff and/or ambient aquatic ecosystems in California (B. Du, Tian, et al., 2020; Overdahl et al., 2021; Sutton, 2021).

Tires are not the only potential source of vehicle-related emerging contaminants. An earlier study aiming to use non-targeted analysis and suspect screening to develop a chemical signature for stormwater associated with coho salmon mortality included an examination of tire particle leachate, as well as leachates from used motor oil, used gear oil, power steering fluid, automatic transmission fluid, and dilute solutions of windshield wiper fluid and antifreeze (Peter et al., 2018). Along with tires, motor oil was identified as a significant contributor to the chemical features observed in roadway runoff. In contrast, each of the other fluids examined could be linked to less than 5% of the chemical signature in runoff observed using non-targeted analysis. Suspect screening of urban runoff has also identified ethoxylated surfactants in urban runoff, including alkylphenol ethoxylates with longer ethoxymer chains (see Section 3.1), and polyethylene and polypropylene glycols (B. Du, Tian, et al., 2020; Peter et al., 2018). These surfactants may have multiple vehicle-related sources; octylphenol ethoxylates are commonly used in tires, as noted above, and polyethylene and polypropylene glycols and related surfactants are used in antifreeze and other automotive products. An examination of a stormwater-influenced site in San Francisco Bay using suspect screening methods confirmed that these unique contaminants transported via urban runoff are present in ambient ecosystems (Overdahl et al., 2021).

4.2.6. Quaternary ammonium compounds

QACs are a major class of primarily cationic surfactants with a chemical structure typically consisting of a positively charged nitrogen atom connected to four hydrophobic alkyl or benzyl groups. This chemical class includes hundreds of individual chemicals and chemical mixtures, with many considered high production volume chemicals in the US (USEPA, 2016). Commercial mixtures contain QACs with varying alkyl chain lengths; variation in the anion (such as chloride or methosulfate) associated with the QAC salt adds an additional layer of complexity that can limit the utility of Chemical Abstract Service (CAS) registry names and numbers for harmonization of monitoring data. Three subclasses of QACs that have been the subject of environmental monitoring are alkyltrimethyl ammonium compounds (ATMACs), benzylalkyldimethyl ammonium compounds (BACs or benzalkonium compounds), and dialkyldimethyl ammonium compounds (DADMACs) (X. Li et al., 2018; Pati & Arnold, 2020; C. Zhang et al., 2015). These and other QACs were added to the California Biomonitoring Priority Chemicals List in June 2021 (https://biomonitoring.ca.gov/sites/default/files/downloads/PriorityChemicalsList_June2021.pdf).

QACs are used in a wide variety of applications. ATMACs, BACs, and DADMACs with shorter alkyl side chains are used as antibacterial active ingredients in cleaning products, antibacterial soaps, health care products, and hand wipes (California Biomonitoring, 2019; D. Li et al., 2020; C. Zhang et al., 2015). Nearly half of the USEPA's N List of disinfecting products proven to effectively kill SARS-CoV-2 contain QACs, suggesting recent increases in use are likely (Hora et al., 2020). Thanks to their antimicrobial properties, these QACs may also replace triclosan and triclocarban (Section 3.8) in some types of products.

Some QACs can also function as preservatives in personal care and cleaning products. QACs with longer alkyl side chains have desirable surfactant and anti-static properties that are routinely used in fabric softeners, hair conditioners, as dispersants, and in oil and gas recovery (California Biomonitoring, 2019; Lara-Martín et al., 2010; Melin et al., 2014). Many of the consumer, commercial, and health care uses of QACs result in discharges to wastewater, indicating a major pathway for environmental contamination is likely to be via treated effluent and land application of biosolids (Pati & Arnold, 2020; C. Zhang et al., 2015). Outdoor uses may lead to discharges via urban and/or agricultural runoff (California Biomonitoring, 2019; Pati and Arnold, 2020).

In aquatic ecosystems, QACs undergo significant partitioning to sediment. Targeted monitoring of sediment samples indicates QACs are present in San Francisco Bay. Eleven surface sediment samples and one sediment core sample were tested for 14 QACs, including ATMACs, BACs, DADMACs, and others (Arnold, 2021; Miller et al., 2020). Seven QACs were detected at two or more surface sediment sites (C12, C14, C18 BACs; C12, C16, C18 DADMACs; benzethonium), with the greatest sums detected in Grizzly Bay (North Bay) and Lower South Bay. Of note, no QACs were detected at quantifiable levels at upstream sites in the Sacramento and San Joaquin Rivers. BAC-C18 was the most widely detected (nine sites), while DADMAC-C18 had the highest concentrations (range 42–830 µg/kg dw; median 290 µg/kg dw). The analyzed sediment core spans roughly 60 years of sediment deposition (approximately 1951-2009) and contains the same seven QACs. DADMAC-C18 was found in all layers at the highest concentrations, and averaged 50 times greater than the next most pervasive compound, BAC-C18. The concentration profile of DADMAC-C18 with depth suggests a declining temporal trend in sediment, possibly due to declining use of this QAC. A key uncertainty is the effect of partitioning to sediment on the ecotoxicity of QACs.

Potential recent and rapid changes in the use and discharge of QACs relating to the COVID-19 pandemic may be examined via sustained monitoring of effluent or receiving waters, a major data gap and the subject of current research (e.g., Arnold, 2021). Suspect screening indicates QACs are present in San Francisco Bay water samples

(Overdahl et al., 2021), but quantitative measurements are not yet available. Similarly, little to no data are available for the rest of the state.

5. Recommended Priorities for Monitoring and Management of CECs in California Waters

The risk screening of available occurrence data for emerging contaminants in ambient aquatic environments of California (Section 3), in combination with the review of notable data gaps (Section 4), ultimately informs recommended state priorities for monitoring and management. Additional factors that influence these recommendations include: trends in use and environmental release; persistence in the environment; geographic distribution of data; anticipated exposure to wildlife and humans; availability of toxicity information; limitations of available analytical methods; and current and/or upcoming management actions.

As noted previously, we evaluated CECs using a class-based approach, in which individual chemicals are grouped as members of broader classes defined primarily by chemical similarities (e.g., phthalates), or by function or use in society (e.g., pharmaceuticals). This approach is favored by multiple state agencies grappling with monitoring and management of emerging contaminants. Members of a chemical class often have similar properties with respect to persistence, bioaccumulation, and/or toxicity, such that considering broad classes of emerging contaminants in risk screening exercises generally results in more accurate and efficient evaluations of risk. This approach can help predict potential concerns of poorly studied compounds that are chemically similar to a better understood compound. Similarly, members of a use class might be addressed by a single management action that affects product requirements (e.g., flammability standards that impact use of flame retardants) or disposal practices (e.g., pharmaceutical take-back programs). Finally, the class-based approach provides flexibility to address shifts in manufacturing toward potentially regrettable substitutes, which can result when management actions address individual chemicals and neglect consideration of the broader class as used in commerce.

Monitoring recommendations for ambient ecosystems are generally guided by several essential concepts and themes:

> Use analytical methods that **quantify multiple members within classes of concern**, and provide **sufficient sensitivity** to ascertain whether measured levels exceed monitoring trigger levels or toxicity thresholds. For many of the classes reviewed in this document, robust occurrence datasets to conduct risk screening are available for only one or a few contaminants, with major data gaps and uncertainties associated with other members of the class. Such a narrow focus for targeted analytes may not be justified based on occurrence data reported elsewhere in the world, and would not be likely to identify the impacts of market shifts including regrettable substitutions. An additional constraint to use of available occurrence data is that many studies have employed methods with detection limits that exceed the toxicity thresholds for many emerging contaminants of interest. It is not possible to know whether a sample examined with a relatively insensitive method, which indicates no detectable concentration

of a contaminant, might actually have levels of the contaminant exceeding its toxicity threshold. A recent monitoring study guidance document for California recommended maximum method reporting limits for some individual emerging contaminants, with values generally set to half the concentrations indicated by available monitoring trigger levels (Dodder et al., 2015).

Expand the geographic coverage of screening level emerging contaminants monitoring in the state, to provide sufficient occurrence data for assessment of water, sediment, and tissue in freshwater, estuarine, and marine settings. Screening level studies can be performed in water bodies likely to be representative of general urban influence, or may be located in settings thought to represent worst-case scenarios for the state in terms of occurrence of specific contaminant classes and associated risk. Useful models of site selection and study design for screening level monitoring within multiple Regional Water Quality Control Board (RWQCB) regions include studies of the Russian River in the North Coast RWQCB (K. Maruya et al., 2018); status and trends monitoring and special studies conducted by the Bay RMP (https://www.sfei.org/rmp); an ongoing study of CECs in the Sacramento-San Joaquin Delta (https://deltarmp.org/water-quality-monitoring/); and studies of effluent-dominated waterways in southern California (Sengupta et al., 2014; Maruya et al., 2016).

For contaminants already addressed via significant management actions, **evaluate recovery by assessing temporal trends**. Such monitoring activities can provide information as to whether the management action is having the intended effect on ambient water quality. Recovery can be quantified in representative ecosystems, and/or settings thought to embody a worst-case exposure scenario. Monitoring studies designed to assess trends require careful consideration of site selection and monitoring frequency, as variables like these will determine the magnitude of an observable trend.

Management recommendations for the Water Boards provided below are similarly guided by a few essential concepts and themes:

A foundational focus for management of emerging contaminants is **source identification**. Many of the contaminants and classes screened in this report are present in multiple sources within urban settings. In this context, the term source represents the original product or use from which contaminants are released, such as a consumer product. Effective management requires an examination of the breadth of these sources, with attention to the potential for each to contribute to significant levels of environmental contamination. For chemical classes with diverse uses, we generally recommend a desktop assessment of uses with respect to their potential to contribute to environmental contamination, to inform identification of targets (e.g., specific consumer products or applications) for management actions.

Pollution prevention is generally considered the most effective and efficient means of addressing environmental contamination. Sources to prioritize for action include those that contribute relatively large amounts of contaminants to the ambient environment, as well as those that are more readily amenable to control. Specific pollution prevention actions will vary depending on the source, and can include product reformulation, chemical or product bans or phase-outs, limits on product use, extended producer responsibility, reduced releases into the environment, and improved disposal options, among others.

Cross-agency collaboration is essential for effective outcomes. The Water Boards may not have the regulatory authority to address CECs via controls on specific products and their uses. However, the Water Boards are in an excellent position to identify emerging contaminants of concern and related sources due to potential impacts in aquatic ecosystems. Identifying high priority CECs and sources for which urgent monitoring and management actions are recommended would be useful for partner agencies such as CDPR and DTSC's Safer Consumer Products program, and could inform workload, staffing, and priorities. Cross-agency collaboration will allow for better utilization of available state tools, integration with existing efforts where possible, and will help avoid duplicative efforts. In particular, communication with partner agencies that can take management actions related to control of specific sources to prevent pollution from CECs is important.

Additional coordination with similar agencies and organizations focused on major ecosystems outside of California, such as the Chesapeake Bay, Puget Sound, Gulf of Mexico, and Great Lakes could increase opportunities for engagement. Many of the classes and compounds of interest in California are also priorities for action elsewhere.

Safer disposal practices that limit environmental contamination are essential for chemicals of significant concern in today's products. Products that are designed to be disposed of down the drain can be particularly difficult to address. Attempts to promote recycling and a circular economy may have the unintentional consequence of maintaining the presence of problematic chemicals in products and in the waste stream. Bans and phase-outs may result in the need to dispose of large quantities of older products; disposal options should not create additional environmental burdens. Education and outreach activities can be particularly important for adoption of safer disposal practices.

5.1. Monitoring and management recommendations for classes of concern

In this section, we provide broad Water Boards monitoring and management recommendations specific to each of the classes included in our risk screening exercise. We recommend an overall monitoring approach that is intended to generate data to directly inform management; a comprehensive statewide occurrence dataset is generally not required prior to action. Specific management options are likely to depend on the types of sources or uses that are prioritized for action. In many cases, a useful initial step in management of chemical classes is an assessment of available information about the many possible uses in urban settings, to provide guidance for impactful and/or actionable targets.

5.1.1. Alkylphenols and alkylphenol ethoxylates

Risk tier: Possible Concern. There are important data gaps in our understanding of occurrence levels to evaluate risks. Only a small subset of analytes are monitored (i.e., NP, OP, short-chain NPEOs and OPEOs), while long-chain ethoxylates have not been monitored. Additionally, water and sediment monitoring data are limited, particularly in marine ecosystems. Some monitoring records are insufficiently sensitive to inform risk screening.

Additional factors: Persistence. APEOs degrade to short-chain APEOs and APs (including NP and OP) that are the most persistent and toxic. Persistent compounds build up in the environment over time, suggesting that risks will increase if existing uses and discharges continue (Cousins et al., 2019).

Past and current management actions: Federal and state bans or phase-out of AP/APEOs in a few discrete products have been implemented. This includes California bans for specific household cleaning products via California Air Resources Board regulations (CCR title 17, §§ 94507-94517), and voluntary nationwide phase-out from consumer laundry detergent. DTSC is currently evaluating use of APEOs in industrial laundry detergent and tires.

Monitoring recommendations:

- We recommend additional screening level monitoring of AP/APEOs in aquatic ecosystems in California. This recommendation is driven primarily by limitations in available occurrence data, and is additionally informed by the persistence of NP and OP in the environment. Ideally, studies should use methods that have detection limits at or below toxicity thresholds to inform risk screening. A recent monitoring study guidance document for the state recommended maximum method reporting limits of half the monitoring trigger levels, where available (Dodder et al., 2015). Current monitoring has focused on a small subset of analytes (i.e., NP, short-chain NPEOs, OP, short-chain OPEOs), and levels of long-chain ethoxylates have not been reported. Additionally APEOs are a subclass within a larger class of ethoxylated surfactants, which may also lead to environmental contamination. Collaborations with academic researchers can address these data gaps using more exploratory methods when standardized methods are not yet available. For example, there is an ongoing collaboration with Duke University to investigate a broader array of these compounds in the San Francisco Bay (Lindborg et al., in prep). Studies should include a broad array of analytes where available. Monitoring of water and sediment in freshwater, estuarine, and marine settings of California is needed. Ecosystems strongly influenced by wastewater and urban runoff may be considered worst case scenarios of particular interest to assess potential risks associated with APEOs.

Management recommendations:

- We recommend that the Water Boards assess the uses of AP/APEOs in society, with respect to their potential to contribute to environmental contamination, to inform identification of specific targets of management actions. A desktop assessment of the diverse uses of AP/APEOs, while limited by significant data gaps (e.g., approximate volumes used in different product types, transport from products to the environment), may nevertheless provide insights that permit a prioritization of sources to be addressed by management actions. Prioritization could include factors such as whether the function or use of the APEOs in a product is readily substitutable with a safer alternative, and/or whether the source is likely to be a major contributor to environmental contamination. Recent investigations using non-targeted analysis have detected AP/APEOs in urban stormwater runoff and associated receiving waters (Overdahl et al., 2021; Tian et al., 2020). These findings emphasize the need to further investigate the stormwater pathway to inform source identification (including both indoor and outdoor uses) and management actions. For example, AP/APEOs are also used in tire rubber, industrial cleaning products, paints and coatings, and building materials (EPA, 2012; Lamprea et al., 2018), which have potential pathways to urban stormwater runoff.
- We recommend that the Water Boards support DTSC in current efforts to evaluate and mitigate impacts from NP/NPEOs in industrial laundry detergents and OP/OPEOs in tires (DTSC, 2021). Should the Water Boards or others provide evidence that other products or uses of AP/APEOs are likely to result in contamination of aquatic environments, this could inform future selection of Priority Products within DTSC's Safer Consumer Products Program. Existing restrictions on use of APEOs within the European Union, where these compounds are considered a Very High Concern (ECHA, 2013), suggest that safer alternatives to these compounds may already exist for many products.
- We recommend that the Water Boards and others manage AP/APEOs using a class-based approach that acknowledges APEOs as a subclass within a larger class of ethoxylated surfactants to reduce the likelihood of regrettable substitutions. Alkylphenol ethoxylates are increasingly being replaced by other ethoxylated surfactants with limited monitoring data globally. DTSC is seeking to gather occurrence data on long-chain APEOs, alcohol ethoxysulfates, secondary alkane sulfonates, alcohol ethoxylates, and alkylbenzene sulfonates as part of the Priority Product review of NPEOs in industrial detergent (DTSC, 2018).

5.1.2. Bisphenols

Risk tier: Moderate Concern. This classification is driven primarily by observations of bisphenol A (BPA) in California freshwater, as well as in estuarine sediment. Bisphenol A is the most well known and widely studied member of this chemical class.

Additional factors: Regrettable substitution. In some cases, manufacturers have begun to substitute other bisphenols in place of BPA in specific products or applications. Thus, while BPA use remains high, the production of alternatives such as BPF and BPS is increasing (Rochester and Bolden, 2015). Despite this shift within industry, occurrence data on bisphenols other than BPA are extremely limited in California. Studies elsewhere indicate BPF and BPS are commonly observed in environmental matrices (Chen et al., 2016).

Past and current management actions: Federal or state legislative bans of BPA in a few discrete products have been implemented. These bans have predominantly applied to plastic products that come in contact with food, including baby bottles, sippy cups, and sports bottles. California also requires point-of-sale notification of the potential for BPA exposure through the protective linings on metal food and beverage cans.

Monitoring recommendations:

- We recommend additional screening level monitoring of bisphenols in aquatic ecosystems of California. This recommendation is driven primarily by the lack of data for bisphenols other than BPA, in light of observations of regrettable substitution. Ideally, studies should include BPA, BPF, and BPS, at minimum, and use methods with detection limits well below toxicity thresholds. Recent state pilot study guidance for CECs recommends a minimum reporting limit for BPA of 0.03 μg/L in water (Dodder et al., 2015); reporting limits of around 0.002 μg/L are readily achievable. Monitoring of water and sediment in freshwater, estuarine, and marine settings of California is needed, with a focus on screening level monitoring at present (as opposed to an evaluation of temporal trends). Ecosystems strongly influenced by wastewater and urban runoff may be considered worst case scenarios of particular interest to assess potential risks associated with bisphenols.

Management recommendations:

- We recommend that the Water Boards assess the uses of bisphenols in society with respect to their potential to contribute to environmental contamination, to inform identification of specific targets of management actions. Prioritization could include factors such as whether the function or use of the bisphenol in a product is readily substitutable with a safer alternative, and/or whether the source is likely to be a major contributor to environmental contamination. Of additional interest is whether a particular use of bisphenols results in recirculation of the compound via recycling, potentially resulting in further human or ecological exposures, as has been observed for BPA in paper products (Liao & Kannan, 2011)

- We recommend that the Water Boards and others manage bisphenols using a class-based approach that acknowledges and accounts for the observed substitution of other bisphenols for BPA by some manufacturers, a substitution that may prove to be regrettable. Previous legislative attempts to address bisphenols beyond BPA include California's 2019 "Skip the Slip" legislation (AB161), which aimed to phase out thermal paper receipts due to concerns relating to BPA and BPS, replacing them with electronic receipts; and New York's 2018 children's products bill (AO9997), which would have expanded an existing ban on the sale of children's products containing BPA to include BPAF, BPAP, BPB, BPF, BPS, and BPZ. Neither of these bills passed. Information from the Water Boards could inform pollution prevention actions via legislation or other mechanisms.
- Should the Water Boards or others provide evidence that a product or use of BPA and/or other bisphenols is likely to result in contamination of aquatic environments, this could inform future selection of Priority Products within DTSC's Safer Consumer Products Program. We recommend supporting this regulatory process. The Safer Consumer Products Program provides another approach to address chemicals of concern in products and avoid switching to regrettable substitutions. Should DTSC select a product containing a bisphenol as a Priority Product, manufacturers of that product would then conduct an alternatives analysis to evaluate possible alternatives and select an appropriate option. The USEPA prepared an assessment of alternatives to BPA in thermal receipt paper, and identified a number of concerns relating to other bisphenols that were under consideration (USEPA, 2015). This assessment suggested that substitution with another member of the chemical class could be problematic. The DTSC Safer Consumer Products Program alternatives analysis process may be particularly useful when a clearly safer alternative is not readily available for a particular function or use of a bisphenol. At present, DTSC has not indicated that it plans to list a bisphenol-containing product as a Priority Product.

5.1.3. Organophosphate esters

Risk tier: Very High Concern. This classification is driven primarily by observations of TDCIPP in freshwater and estuarine settings in California.

Additional factors: Regrettable substitution, changing use trends. Organophosphate esters are typically used as plastic ingredients or as additive flame retardants in a variety of products. Following bans on PBDEs, regrettable substitution resulted in increasing use of some organophosphate esters as flame retardants in upholstered furniture (Stapleton et al., 2012). A later study of polyurethane foam products indicated declines in use of flame retardants in sofas and loveseats, likely a result of a change to California's flammability standard for foam furniture (Cooper et al., 2016).

Past and current management actions: In 2013, the state flammability standard for foam furniture (TB 117) was updated from an open flame test to a smoldering test for compliance purposes (TB 117-2013), essentially eliminating the need for manufacturers to add chemical flame retardants like organophosphate esters to upholstered furniture. Among the flame retardants identified in foam products, fewer products purchased after 2014 contained TDCIPP, a potential consequence of the listing of this compound in 2011 via California's Proposition 65 due to its carcinogenic potential. Testing data suggested that TDCIPP may have been replaced by other organophosphate esters, like TCPP, by some manufacturers (Cooper et al., 2016).

Additional state management actions have targeted flame retardant uses of organophosphate esters. In 2017, DTSC's Safer Consumer Products Program listed children's foam-padded sleeping products with TDCIPP or TCEP as Priority Products. Manufacturers of these products generally elected to eliminate use of these flame retardants rather than complete an alternatives analysis. DTSC later tested 21 products and found none to contain the specified compounds at levels suggesting that they were intentionally added, indicating compliance with regulations (DTSC, 2019).

A subsequent state ban on the manufacture, distribution, and sale of upholstered furniture, mattresses, and children's products containing any flame retardant, effective in 2020, is expected to limit the presence of organophosphate esters in these products in California. However, it is important to note that organophosphate ester flame retardants are still used in many other products not covered by these management actions. Organophosphate esters also have many other non-flame retardant uses, particularly as plastic ingredients, which have not been addressed at this time.

The USEPA recently issued a regulation on one member of this class, phenol, isopropylated phosphate (3:1) (PIP (3:1); CASRN 68937-41-7), which the agency has determined meets the requirements for expedited action under the Toxic Substances Control Act. The final rule "prohibits the processing and distribution of PIP (3:1), PIP (3:1)-containing products, and PIP (3:1)-containing articles, with specified exclusions; prohibits or restricts the release of PIP (3:1) to water during manufacturing, processing, distribution, and commercial use; and requires persons manufacturing, processing, and distributing in commerce PIP (3:1) and products containing PIP (3:1) to notify their customers of these prohibitions and restrictions and to keep records" (EPA-HQ-OPPT-2021-0598-0001). The USEPA recently extended the compliance deadline to October 31, 2024, at the request of manufacturers. One isomer of PIP (3:1), tris(2-isopropylphenyl) phosphate, has been observed at low levels in estuarine waters of California (90th percentile concentration 0.0023 µg/L). This new regulation is expected to reduce amounts of PIP (3:1) being manufactured, processed, distributed in commerce, used and disposed, which should lead to lower levels in the environment.

Monitoring recommendations:

- We recommend additional monitoring for organophosphate esters in aquatic ecosystems of California. Screening level data are urgently needed to expand the breadth of data with respect to compounds and ecosystems for this class of Very High Concern. Given the testing data suggesting substitution of one compound for another within this class, methods that cover a broad number of analytes are preferred. In particular, we recommend including compounds with concentrations approaching or greater than associated thresholds, such as TDCIPP, TPhP, EHDPP, and TBOEP. Freshwater and marine data generated using methods with sufficiently low detection limits are particularly lacking in the state.

- In addition, more robust monitoring to assess temporal trends in light of existing management actions to limit flame retardants in some products may be warranted. Such studies could be conducted on water bodies considered generally representative of urban influence in California, and/or worse-case exposure scenarios.

Management recommendations:

- We recommend that the Water Boards conduct a desktop assessment of the uses of organophosphate esters in society with respect to their potential to contribute to environmental contamination, to inform identification of specific targets of management actions. Factors such as the potential for a source to release organophosphate esters to the environment, as well as the availability of safer substitutes, could drive a management strategy. In addition, widespread use in plastic products can lead to recirculation via recycling, increasing potential human and ecological exposures (Kajiwara et al., 2022).
- We recommend that the Water Boards and others manage organophosphate esters using a class-based approach informed by past observations of shifts in industry that can include regrettable substitutions (Blum et al., 2019). DTSC's selection of a Priority Product with specific members of the organophosphate ester class provides one example of an approach designed to avoid regrettable substitution through an evaluation of whether the chemical of concern is necessary, followed by a careful analysis of potential alternatives. Likewise, the legislation banning all flame retardants in specific products provides comprehensive action on a use class, which is more impactful than a chemical-by-chemical ban.
- We recommend that the Water Boards consider engaging as a stakeholder in efforts to evaluate existing flammability standards in various products. These flammability standards were established by state and federal agencies or international industry groups in an attempt to mitigate risks caused by fire, and have resulted in widespread use of chemical additive flame retardants. Recent investigations indicate that flame retardants may not provide a significant fire safety benefit as currently used in products (Charbonnet et al., 2020). Where safety can be achieved without the use of these compounds, reductions can be achieved in specific product types through flammability standards that reduce or eliminate requirements for use, or through flame retardant bans via legislation. As noted above, California's Bureau of Home Furnishings recently reexamined

the state flammability standard for upholstered furniture, updating it to eliminate the need for flame retardants while increasing fire safety.

5.1.4. Phthalates

Risk tier: Possible Concern. This classification is driven primarily by limited detections using analyses with method detection limits exceeding toxicity thresholds, as well as limited data availability in California ecosystems.

Because this risk screening focused only on concentrations in ambient waters of California, it does not adequately address all risks of phthalates, which also occur in sediment, where risks may be equal or higher.

Additional factors: None.

Past and current management actions: Federal and state legislative bans, motivated by human health concerns, address a subset of more well-studied phthalates in key product types. The use of several phthalates is restricted in children's toys and child care products to levels no greater than 0.1% (CPSC, 2017). This federal regulation was preceded by similar bans implemented by California and other states (Eichler et al., 2019). California also lists six phthalates via Proposition 65 for concerns relating to carcinogenicity, neurotoxicity, reproductive toxicity, and/or birth defects: butyl benzyl phthalate (BBzP), dibutyl phthalate (DBP), di-(2-ethylhexyl) phthalate (DEHP), di-isodecyl phthalate (DIDP), diisononyl phthalate (DINP), and di-n-hexyl phthalate (DnHP). Meanwhile, the USEPA has initiated risk evaluations for a number of individual phthalates via the Toxic Substances Control Act (https://www.epa.gov/assessing-and-managing-chemicals-under-tsca/chemicals-undergoing-risk-evaluation-under-tsca). Far more extensive bans based on human health concerns have already occurred in Europe via chemicals management and cosmetics regulations (Eichler et al., 2019).

Monitoring recommendations:

- We recommend additional screening level monitoring of phthalates in surface waters, particularly in estuarine and marine settings where data are limited. This recommendation is driven by the limitations of available monitoring data. Methods that have detection limits at or below toxicity thresholds are essential to inform risk screening. A recent monitoring study guidance document for the state recommended maximum method reporting limits of half the monitoring trigger levels, where available (Dodder et al., 2015). Ecosystems significantly influenced by wastewater and/or stormwater discharges may be considered worst case scenarios of particular interest to assess potential risks associated with phthalates.

Management recommendations:

- We recommend that the Water Boards assess the uses of phthalates in society with respect to their potential to contribute to environmental contamination, to inform identification of specific targets of management actions. In addition to their potential to release phthalates to environmental matrices, factors such as the availability of safer substitutes could be used to prioritize sources for management.
- We recommend that the Water Boards and others manage phthalates using a class-based approach that acknowledges and accounts for the potential for regrettable substitution. Information from the Water Boards could inform pollution prevention actions via legislation or other mechanisms.
- Should the Water Boards or others provide evidence that a product or use of one or more of these phthalates is likely to result in contamination of aquatic environments, this could inform future selection of Priority Products via DTSC's Safer Consumer Products Program. We recommend supporting this regulatory process. At present, DTSC has not indicated that it will list a phthalate-containing product as a Priority Product. As noted previously, the alternatives analysis required by this program supports informed rather than regrettable substitution of chemicals of concern.

5.1.5. Brominated flame retardants, including polybrominated diphenyl ethers

Risk tier: PBDEs and other brominated flame retardants are placed in separate tiers.

- PBDEs: Very High Concern. This classification is driven by the potential for more highly exposed fish to exceed advisory tissue levels recommended to protect people eating high fish diets.
- Other brominated flame retardants: Possible Concern. This classification is driven by limited occurrence data and limited toxicity thresholds.

Additional factors: Persistence and bioaccumulation; declining temporal trends for PBDEs (no trends observed for other brominated flame retardants). Statistically significant declines in congeners associated with PBDE mixtures banned in California in the mid-2000s (Penta and Octa), such as PBDE 047, have been observed in multiple matrices in San Francisco Bay (Sutton et al., 2015), though similar trends have not yet been observed via the statewide Surface Water Ambient Monitoring Program's (SWAMP) Stream Pollution Trends (SPoT) monitoring program (B. M. Phillips et al., 2020). In contrast, concentrations of PBDE 209—the primary congener in the commercial mixture Deca, phased out of US production in 2013—have not shown signs of decline in California ecosystems (Sutton et al., 2015; Phillips et al., 2020).

Past and current management actions: Impactful management actions at both state and federal levels include bans and/or phase-outs of all three commercial mixtures of PBDEs. Recent management actions in California, including revised flammability standards and flame retardant bans on upholstered furniture, mattresses, and children's products, may have reduced overall use of brominated (and other) flame retardants in key consumer products. However, flame retardants remain in wide use in other products, such as electronics and some construction materials, for which regulatory or voluntary industry flammability standards remain.

Monitoring recommendations:

- We recommend monitoring of PBDEs in multiple matrices to assess temporal trends and verify the anticipated declines of PBDEs in environmental matrices following bans and phase-outs. Although PBDEs are expected to decline in environmental matrices following bans and phase-outs, recovery may take decades (Oram et al., 2008). In particular, monitoring of PBDE 209 in sediment is needed, as the source of this congener, the commercial mixture Deca, was phased out less than a decade ago, and recovery has not yet been observed. Such monitoring could occur in select locations considered representative of urban-influenced freshwater and marine environments, and could be paired with water monitoring, given the potential for exceedances of the aquatic toxicity threshold. It is important to note that PBDE 209 presents a number of analytical challenges (Kierkegaard et al., 2009), such that particular care must be exercised in sample collection and analysis.
- We recommend monitoring of PBDEs in sport fish and local seafood from urban-influenced water bodies to assess the need for public health interventions. This recommendation is driven by the persistence and bioaccumulation of PBDEs. Populations that consume high fish diets, including subsistence fishers and members of local tribal communities, could face higher exposures to PBDEs in more contaminated locations. Water Boards monitoring of water, sediment, and sport fish could help to identify sites of particular interest to public health agencies.
- We recommend screening level monitoring of other brominated flame retardants. At present, occurrence data for brominated flame retardants other than PBDEs are quite limited, resulting in the Possible Concern classification. We recommend screening level monitoring of these compounds in settings representing worst case scenarios, such as urban-influenced aquatic environments, to provide additional data for risk screening purposes.

Management recommendations:

- Given the anticipated impacts of the bans and phase-outs, we do not have recommendations for further management actions on PBDEs.
- We recommend that the Water Boards consider engaging as a stakeholder in efforts to evaluate existing flammability standards in various products. These product-specific flammability standards have led to widespread use of these compounds, as noted above for organophosphate esters. The Water Boards can consider participating in efforts to evaluate existing flammability standards in various products; where safety can be achieved without the use of flame

retardants (brominated or otherwise), reductions are possible through updated flammability standards or legislation.

5.1.6. Select current-use pesticides (and degradates) with urban applications

We evaluated risks for three pesticides or pesticide groups, which were prioritized based on guidance from CDPR and criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, 90th percentile concentrations near or exceeding toxicity thresholds, and selection of compounds that represent diverse insecticide classes.

Risk tier: The three pesticides or pesticide groups were classified in risk tiers independently because they are chemically diverse and have different modes of action.

- Imidacloprid: High Concern
- Fipronil (and degradates): Very High Concern
- Pyrethroids (including bifenthrin, cyfluthrin, permethrin): High Concern

Because this risk screening focused only on concentrations in ambient waters of California, it does not adequately address all risks of pyrethroids and fipronil, both of which also occur in sediment, where risks are expected to be equal or higher.

Additional factors: Trends in use. All of these pesticides were developed to replace organophosphate insecticides, and use has increased since the 1990s.

Past and current management actions: Unlike other classes described in this report, current-use pesticides are subject to significant levels of regulation. Pesticides are regulated at the federal level by the USEPA, and at the state level by CDPR, which has specific authority to protect California's aquatic ecosystems from harmful impacts caused by pesticides. Through extensive freshwater monitoring and risk assessment, the agency has identified the three pesticides included in this risk screening as posing potential risks to aquatic life. Management actions initiated by CDPR for fipronil and pyrethroids are designed to reduce outdoor urban uses by professional pest control operators, which is anticipated to ultimately reduce levels in urban runoff and receiving waters. No trends in occurrence data have been observed for pyrethroids at this time (Budd et al., 2020); not enough time has passed since adoption of fipronil label changes to adequately assess trends.

Monitoring recommendations:

- Water Boards pesticides monitoring activities in ambient ecosystems should be coordinated with CDPR to maximize the breadth of knowledge and best inform management. Relative to many other CECs, management-relevant monitoring data for current-use pesticides are more abundant in California. CDPR's Surface Water Protection Program conducts pesticides monitoring in freshwater environments across the state to inform regulation. Freshwater ecosystems are generally the focus of toxicity concerns and mitigation efforts.

- Should a specific concern be identified for estuarine or marine settings, we recommend that the Water Boards act to address any occurrence data gaps. The Water Boards and relevant regional or municipal agencies may be better positioned than CDPR to expand the more limited pesticides occurrence data available in California's estuarine and marine ecosystems, providing data that could inform CDPR's regulatory process. At present, CDPR does not monitor estuarine or marine settings.

Management recommendations:

- We recommend that the Water Boards provide strategic support to CDPR, as requested and appropriate, in disseminating information via education and outreach efforts to regulated entities, other stakeholders, and the public. As the agency charged with protecting ambient ecosystems from pesticide impacts, CDPR is the lead for pesticide pollution prevention efforts in California. CDPR's registration process, which has been expanded to more fully address aquatic ecosystem protection, is the state's primary means of preventing pesticides water pollution (Moran et al., 2020). For registered pesticides, CDPR evaluates monitoring data and, where appropriate, may examine the feasibility of different mitigation options designed to prevent pesticide pollution, including both pilot studies and modeling exercises, prior to implementing management options (Moran et al., 2020).

  Collaborative education and outreach efforts supported by the Water Boards have historically played an important role in raising consumer awareness and gaining real-world experience with alternative pest control practices ahead of and in parallel with CDPR's science-based regulatory work. For example, the urban-specific "Our Water Our World" program that was initially supported by Water Board grant funding has long identified and encouraged implementation of pyrethroids alternatives. Currently, similar efforts are underway by some municipal wastewater treatment plants and the San Francisco Bay Regional Water Board to address imidacloprid, fipronil, and some pyrethroids that are used as topical flea control products including spot-on products and flea collars. Studies of pet wash water (Teerlink et al., 2017) and municipal wastewater (Sadaria et al., 2017) indicate pet flea control products may contribute significant levels of pesticides to receiving waters. Efforts are also underway to educate pet owners, veterinarians, and other relevant stakeholders about the negative impacts of these pesticides, and the availability of safer alternatives, particularly oral medications (e.g., https://baywise.org/residential/pets/). Although the USEPA has not indicated plans to take action on these products, CDPR is currently evaluating fipronil and imidacloprid pet spot-on treatments for potential human health risk and is collecting monitoring data on all three chemical classes at municipal wastewater treatment plants. While pesticide regulators consider their next steps, these educational efforts could be expanded and diversified with the support of state, regional, and municipal agencies.

5.1.7. Select pharmaceuticals

We evaluated the associated risks for ten pharmaceuticals, which were selected based on several criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, 90th percentile concentrations near or exceeding toxicity thresholds, selection of compounds that represent diverse use classes, and prioritization of use categories where there are available management actions.

Risk tier: Each pharmaceutical was classified in a risk tier independently because pharmaceuticals are chemically diverse and may have different modes of action.

- 17alpha-ethinylestradiol and 17beta-estradiol: Possible Concern. Methods used to monitor these compounds in California surface waters are insufficiently sensitive, with detection limits several orders of magnitude greater than ecotoxicity thresholds.
- Ibuprofen (NSAID): High Concern.
- Metoprolol (beta blocker): Moderate Concern.
- Gemfibrozil (lipid-lowering): Moderate Concern.
- Fluoxetine (antidepressant): High Concern.
- Azithromycin (antibiotic): High Concern.
- Clarithromycin, erythromycin, and ciprofloxacin (antibiotics): Moderate Concern.

Additional factors: Predicted trends in use. A growing and aging population would be expected to require more pharmaceuticals, leading to increased release of these compounds to the environment, particularly through the wastewater pathway. Additionally, development of new drug treatments will also change the levels and types of compounds discharged.

Past and current management actions: Diverse policies and initiatives from government agencies and the medical community are aimed at reducing unnecessary use of antibiotics, due to human health concerns arising from the evolution of antibiotic resistance bacteria (CDC, 2017). Inappropriate prescription of antibiotics, for example in response to viral infections for which they are ineffective, is widely observed in the US (Chua et al., 2019). Multiple interventions aimed at clinicians and patients have been shown to reduce unnecessary antibiotic prescriptions (Rowe & Linder, 2019; Seppa, 2014). More broadly, California’s comprehensive, producer-funded take-back program to provide safe disposal of pharmaceuticals is a means of addressing improper disposal of drugs.

Monitoring recommendations:

- We recommend additional screening level monitoring of pharmaceuticals in aquatic ecosystems in California. We recommend additional focus on freshwater and marine ecosystems, which have either applied methods that are insufficiently sensitive to evaluate risk, or have not been monitored. Ideally, studies should use methods that have detection limits at or below toxicity thresholds to inform risk screening. A recent monitoring study guidance document for the state recommended maximum method reporting limits of half the monitoring trigger levels, where available (Dodder et al., 2015). Ecosystems strongly influenced by wastewater may be considered worst case scenarios of particular interest to assess potential risks associated with pharmaceuticals. Considering the essential use of pharmaceuticals for protection of human health, we recommend focusing on monitoring that can inform management actions, such as efforts to reduce unnecessary use of antibiotics.

Management recommendations:

- We recommend that the Water Boards seek out and support active efforts to limit unnecessary antibiotic use led by medical and veterinary/agricultural stakeholders as a means of addressing these contaminants at the source. Water quality managers may have more management options available to manage antibiotic use compared to other pharmaceutical use categories; such efforts will require engagement with a broad variety of stakeholders to identify actions that can address both public health and water quality concerns.
- Water quality managers can continue to support efforts to reduce unnecessary discharges of drugs through expansion of existing drug take-back programs that support proper disposal of unused drugs. Wastewater is generally considered a minor pathway by which these contaminants enter the wastewater pathway.

5.1.8. Select personal care and cleaning product ingredients

We reviewed occurrence data and screened risks for four personal care and cleaning product ingredients, which were prioritized based on several criteria including availability and quality of monitoring data, availability and quality of toxicity thresholds, 90th percentile concentrations near or exceeding toxicity thresholds, function in products, and prioritization of use categories where there are available management actions.

Risk tier: Each compound was classified in a risk tier independently because PCCPs are chemically diverse and may have different modes of action.

- Triclocarban: Moderate Concern.
- Triclosan: Low Concern.
- Galaxolide: Low Concern.
- Tonalide: Possible Concern. There are no ecotoxicity thresholds available to evaluate risks.

Because this risk screening focused only on concentrations in ambient waters of California, it may not adequately address all risks of these four compounds, which also occur in sediment.

Additional factors: Persistence. The specific compounds reviewed here are persistent in the environment. Persistent compounds build up over time, suggesting that risks will increase if existing uses and discharges continue (Cousins et al., 2019).

Past and current management actions: Major management actions have already been implemented to reduce triclosan and triclocarban release from specific products; this most importantly includes the FDA's ban of these compounds and other specified antimicrobials in over-the-counter soap and wash products, over-the-counter topical antiseptics in health-care settings, and over-the-counter hand sanitizers (FDA, 2016). Additionally, triclosan is no longer registered in agricultural film and paper mulch, paints and stains, HVAC exchange coils, and apparel textiles uses (USEPA, 2020a).

The FDA's Fair Packaging and Labeling Act requires personal care products to disclose ingredients, but musks and other ingredients in fragrances are exempt as a "trade secret" (i.e., confidential business information). Likewise, fragrance ingredient disclosure is not required at the federal level for cleaning products or other consumer goods. Fortunately, recent legislation in California has begun to increase requirements for fragrance ingredient disclosure through various mechanisms for both personal care products (California Fragrance and Flavor Ingredient Right to Know Act of 2020 [SB 312]) and cleaning products (Cleaning Product Right to Know Act of 2017 [SB 258]). In addition, some manufacturers voluntarily disclose fragrance ingredients down to 100 ppm (100,000 µg/L).

Monitoring recommendations:

- We recommend additional monitoring of triclosan and triclocarban in aquatic ecosystems in California, with particular focus on freshwater and marine ecosystems, which have either applied methods that are insufficiently sensitive to evaluate risk, or have received little attention. Screening level sediment monitoring may be useful. Triclocarban may be considered a higher priority, given its classification in the Moderate Concern tier.
- We recommend monitoring of triclosan and triclocarban to assess potential temporal trends and overall efficacy of the recent management action to limit use in over-the-counter soaps and similar products. This action may be anticipated to reduce environmental contamination. Monitoring could be conducted at a limited number of water bodies considered representative or worst case scenarios with respect to influence of municipal wastewater.
- We also recommend additional screening level monitoring of galaxolide and tonalide because there were limited data available in California. Ideally, studies should use methods that have detection limits at or below toxicity thresholds to inform risk screening. A recent monitoring study guidance document for the state recommended maximum method reporting limits of half the monitoring trigger levels, where available (Dodder et al., 2015). Monitoring could be conducted at a limited number of water bodies considered worst case scenarios with respect to influence of municipal wastewater, a pathway particularly relevant to personal care and cleaning product discharges. Screening level sediment monitoring may be useful.

Management recommendations:

- We recommend that the Water Boards support outreach efforts with pesticides regulators and other agencies and organizations to reduce unnecessary use of antimicrobial products. There is growing recognition for the need for antibiotic and antimicrobial stewardship to address the growing public health risk and cost of antibiotic resistance (CDC, 2017), as noted in the previous discussion on antibiotic pharmaceuticals (Section 5.1.7). Antibiotics specifically kill or limit the growth of bacteria, while antimicrobial is a broader term that includes antibiotics and other (primarily registered pesticidal) compounds that can kill or limit the growth of other microbes as well (e.g., parasites, viruses, fungi). Antibiotic and antimicrobial stewardship includes using these compounds when necessary and appropriate, and reducing use when not, as well as regulation of the use of those antimicrobials registered as pesticides by the USEPA and CDPR.
- We recommend that the Water Boards and others manage triclosan and triclocarban using a class-based approach to managing antimicrobials that acknowledges and accounts for the observed substitution of other antimicrobials for triclosan and triclocarban by some manufacturers, a substitution that may prove to be regrettable. QACs are a class of antimicrobial active ingredients that are used as replacements for triclosan and triclocarban, as well as in many other cleaning and consumer applications, and may warrant further monitoring (see Section 4.2.6).
- We recommend that the Water Boards support regulations that require disclosure of chemical ingredients in consumer products, providing information on which fragrance ingredients are in use and may pose a water quality concern. Fragrances are widely used in products discharged down the drain.

5.1.9. Per- and polyfluoroalkyl substances

Risk tier: Very High Concern. This classification is based primarily on human health risks from ingesting contaminated fish.

Additional factors: Persistence, bioaccumulation, regrettable substitution. Most toxicity, fate, and transport studies have focused on two compounds, PFOA and PFOS. While these two compounds have largely been phased out of production in North America and Europe, there is growing evidence that they are being replaced by other PFAS that are not regulated and not as well understood. Despite this shift within industry, current analytical methods are limited in their ability to identify and quantify the numerous PFAS in commercial use. This makes comprehensive environmental monitoring and risk assessment a significant scientific challenge.

Past and current management actions: In the US, production of PFOS was phased out by 2002, and production of PFOA was phased out by 2015. PFOS was restricted under the global Stockholm Convention in 2009, while PFOA was similarly restricted in 2019, allowing some specific exceptions to a total ban for both compounds.

Additional efforts to control sources of PFAS in California have adopted a class-based approach, as opposed to a focus on individual chemicals. DTSC's Safer Consumer Products Program has established a clear rationale for management actions directed at the entire PFAS class (Bălan et al., 2021), and has begun to apply this approach to carpets and rugs made or sold in California, as well as treatments for converted textiles and leathers. Similarly, state bans on PFAS in paper-based food packaging and products intended for infants and children, both of which take effect in 2023, rely on a class-wide approach, rather than bans of individual compounds. In addition, California bill SB 1044 (Allen), which was signed into law in 2020, prohibits the use of PFAS in class B firefighting foam, with certain exceptions, starting in 2022. The USEPA has recently proposed reporting requirements for a wide range of individual PFAS from specific industry categories (USEPA, 2021a).

A significant wealth of data is currently being generated as a result of the State Water Board's statewide PFAS Action Plan (https://www.waterboards.ca.gov/pfas/). Several actions associated with this Action Plan were implemented in 2019 including: 1) Division of Water Quality issuance of three statewide investigative orders that require sampling and analysis of certain PFAS at select airports, landfills, and chrome plating facilities; and 2) Division of Drinking Water issuance of drinking water system orders that require sampling and analysis of certain PFAS from over 600 public water supply wells. Moving forward, the State Water Board's Action Plan includes additional sampling outwards from the identified impacted public water supply wells; development of strategies for domestic well sampling in impacted areas; focused watershed-based source and public water system investigations; sampling at wastewater treatment facilities; source investigations at refineries and bulk terminals; integration of data collected from Department of Defense facilities; and data analysis and visualization to inform the public and decision makers.

Monitoring recommendations:

- We recommend additional screening level monitoring of PFAS in aquatic ecosystems of California. This recommendation is informed by the Very High Concern risk tier, as well as factors including persistence and bioaccumulation. Monitoring of water, sediment, and fish in freshwater, estuarine, and marine settings of California is needed. Ecosystems strongly influenced by municipal wastewater and urban runoff are of particular interest to assess potential risks associated with PFAS from urban uses, in addition to other settings that may be influenced by industrial applications.
- We recommend sustained monitoring of PFAS at select representative sites to assess potential temporal trends in response to current and future management actions. This class of CECs is receiving considerable attention from state and federal regulatory agencies; it would be useful to ascertain the impacts of various actions on environmental occurrence.
- The Water Boards may also consider strategic use of broader and non-standardized methods to screen for the presence of PFAS in aquatic

ecosystems. While standardized, targeted analyses using LC-MS/MS and GC-MS/MS are improving our ability to detect environmental concentrations of certain PFAS, these methods have limited ability to detect the large and growing number of PFAS in use and present in various matrices. Additional methods can be used to assess broader PFAS occurrence. For example, there are various analytical methods (e.g., combustion ion chromatography (CiC), particle induced gamma-ray emission (PIGE) spectroscopy) that reduce the complexity of multiple PFAS to a simplified parameter of total organic fluorine in a sample. The total oxidizable precursor (TOP) assay takes a different approach to simplifying the complexity of PFAS in a sample by oxidizing the precursor PFAS to form terminal perfluoroalkyl acids that can be identified and quantified using traditional LC-MS/MS (Houtz & Sedlak, 2012). Suspect screening and non-targeted analytical methods are more exploratory methods (see Section 4.2) that greatly enable our ability to tentatively identify PFAS for which we do not have available standards. All of these different analytical methods quantify a different fraction of the PFAS present in the sample, provide different levels of detail about the chemical identities, and have different shortcomings. Results from a regional study of PFAS in wastewater (Mendez et al., 2020) show a significant presence of PFAS precursors that were not captured by standard LC-MS/MS target methods.

Management recommendations:

- We recommend that the Water Boards support efforts to require disclosure of PFAS in commerce, as well as production and consumption levels, and use locations and emissions, including byproducts and impurities to manage PFAS. Valuable analyses that identify potential uses of PFAS in general (Glüge et al., 2020) and within the construction and building products sector (Fernández et al., 2021) already exist, providing a higher level of insight as to sources than exists for many other classes reviewed here. The availability of this information is crucial for identifying what PFAS chemistries should be monitored in the environment for risk assessments, as well as informing what sources and uses should be reduced or eliminated. Requirements for PFAS disclosure would also help consumers and organizations to avoid products containing PFAS to further add market pressure to limit use of PFAS. The USEPA has recently proposed a PFAS reporting requirements for specific industry categories (USEPA, 2021a). Chemical inventories must have sufficient detail and tight reporting scheme to effectively inform management (Steindal & Grung, 2021).
- We recommend that the Water Boards conduct a desktop assessment of the uses of PFAS in society with respect to their potential to contribute to environmental contamination, to inform identification of specific targets of management actions. Such an assessment would be greatly strengthened by disclosure requirements recommended above.
- We recommend that the Water Boards address impacts of PFAS to aquatic ecosystems through cross-agency collaborations at the local, state, and federal level. This includes the Water Boards leveraging the ongoing PFAS Action Plan

to identify PFAS sources and pathways to inform monitoring and management strategies moving forward. Additionally, while the Water Boards can take the lead on monitoring environmental occurrence, other agencies may have regulatory authorities suited to provide essential information on the uses of PFAS, regulate the uses of PFAS, or address human health concerns, particularly relating to fish consumption.

- We recommend that the Water Boards and others manage PFAS using a class-based approach, and support policies that limit PFAS to "essential uses." The concept of essential uses was defined in the Montreal Protocol, which phased out the use of ozone-depleting chlorofluorocarbons except for use that is "necessary for health, safety or is critical for the functioning of society" and for which "there are no available technically and economically feasible alternatives." A class-based approach that reduces nonessential uses of PFAS is considered precautionary; advocates for the essential-use approach point to a long list of concerns about PFAS as a class including chemical persistence and human and ecological health risks, as well as the lack of comprehensive analytical methods (Cousins et al., 2019). Considering the current wide uses of PFAS in consumer, medical, and industrial applications, limiting PFAS uses and identifying safer alternatives may require a phased path to understanding the uses of PFAS, refining the definition or criteria for "essential uses," and continuing to identify safer alternatives to further limit uses of PFAS (Cousins et al., 2019).
- We recommend that the Water Boards support future management of PFAS through DTSC's Safer Consumer Products Program, especially selection of Priority Products that can impact water quality. DTSC has already established a clear rationale for management actions directed at the entire PFAS class (Bălan et al., 2021). In addition to action on carpets and rugs made or sold in California, DTSC has listeded another Priority Product as a next step in this process, treatments containing PFAS for use on converted textiles or leathers.

5.2. Additional classes and tools of interest

Our risk screening proof-of-concept (Section 3) focused on several contaminant classes previously identified as of potential concern, and for which California occurrence data are available. While this risk screening encompassed over 100 compounds, this represents a small subset of the thousands of chemicals in commerce; many more have yet to be examined. The Water Boards are uniquely well-positioned to establish and maintain a forward-looking approach to address emerging contaminants through the statewide CEC Program.

In Section 4, we outlined some of the new tools and approaches that can inform future monitoring and management priorities. First, we emphasize the need for continued improvements in targeted analytical methods to allow quantification of greater numbers of related contaminants at levels well below toxicological thresholds. This concept has been integrated into the monitoring recommendations for classes of CECs outlined in Section 5.1. It is important to note that interim use of non-standard methods developed by research laboratories may be useful to achieve method detection limits suitable to inform risk assessment.

We further recommend strategic application of suspect screening and non-targeted analysis to provide early insights regarding the presence of unanticipated contaminants in environmental matrices. As described in Section 4.2, these novel approaches can reveal the occurrence of compounds that scientists and regulators might not have identified or prioritized, due to the gaps in knowledge concerning chemical production, product manufacturing, and contaminant transport, fate, and toxicity. In California, models for use of these techniques include multiple studies of southern California marine mammal blubber using GC-based techniques (Shaul et al., 2015; Mackintosh et al., 2016; Trego et al., 2018; Cossaboon et al., 2019) and a study of San Francisco Bay estuarine water and wastewater effluent using LC-based techniques (Overdahl et al., 2021). In particular, suspect screening and non-targeted analysis could be applied to pathways such as wastewater and stormwater, as contaminant concentrations may be higher in these matrices relative to ambient water and sediment. These exploratory techniques are best applied using thoughtful study design including selection of representative and worst case scenario sampling locations to inform monitoring and management priorities.

Prior studies using these analytical screening methods, along with complementary in silico screening of chemical inventories, have revealed some interesting chemicals and classes that were not included in the present risk screening, and merit further monitoring in California matrices. Among the compounds highlighted in Section 4.2, we specifically recommend additional monitoring of the tire-derived contaminant 6PPDQ, which causes acute mortality to coho salmon (Tian et al., 2021). While California data are limited at this time, 6PPDQ has been observed at toxicologically relevant levels in stormwater in the San Francisco Bay Area, and roadway runoff in Los Angeles. Coho salmon populations are listed as threatened or endangered in different regions of California. To support local coho populations, additional monitoring can indicate whether this contaminant is present at harmful levels in specific tributaries, particularly those likely to be influenced by roadways with high levels of vehicle use.

An additional priority for monitoring is the broad functional class of compounds used as plastic ingredients (monomers, additives, and processing aides; Section 4.2.4). As some of the more well-studied plastic ingredients reviewed in this risk screening fall out of favor with some manufacturers, other chemicals enter the market as replacements. For example, a study of German rivers has identified a declining trend in concentrations of the phthalate DEHP in water samples, and an increasing trend of di(2-propylheptyl) phthalate, over a period between the mid-2000s and 2017 (Nagorka & Koschorreck, 2020). Other common replacements for phthalate plasticizers, such as di(isononyl)cyclohexane-1,2-dicarboxylate (DINCH) and tris(2-ethylhexyl) trimellitate (TOTM), were also observed in these rivers shortly after their introduction into commerce. A commonly used substitute in the US, di-(2-ethylhexyl) terephthalate (DEHT, also known as DOTP; not a phthalate), is widely observed in national human biomonitoring studies (Silva et al., 2019), but has not been characterized in California or US aquatic ecosystems. We recommend strategic monitoring of ambient aquatic environments and contaminant pathways (e.g., wastewater effluent and urban stormwater) using newer targeted analytical methods designed to quantify these plasticizers, as well as other plastic ingredients identified in the literature as occurring in aquatic environments. Occurrence data can be used for risk screening, and may also serve as baseline data relevant to broader statewide efforts to manage the impacts of plastic and microplastics.

A third monitoring recommendation concerns chlorinated paraffins, a class of contaminants not reviewed in this risk screening, and for which few California data are available. Among the brominated and chlorinated compounds described in Sections 4.2.1 and 4.2.2, chlorinated paraffins are of particular interest due to high production volumes, the likelihood that medium- and long-chains have been substituted for short-chain chlorinated paraffins when US production of the latter was halted in 2012, and available monitoring data outside of California in sediment and biota (Zeng et al., 2015; Zheng, Lam, Chen, et al., 2017; Zeng, Lam, Horii et al., 2017; Du et al., 2018; Yuan et al,. 2019). In contrast, a comparison of new and existing chemicals in commerce in the EU suggests a shift away from development of brominated substances (Strempel et al., 2012), a finding that argues against expanded monitoring of brominated compounds. A more modest shift away from chlorinated compounds was also observed, along with a shift towards fluorine chemistries.

A fourth monitoring recommendation is focused on widely used linear and cyclic siloxanes. These compounds have been observed in environmental matrices worldwide (Rücker and Kümmerer, 2015), but there are very few data in California. In particular, increased use of decamethylcyclopentasiloxane (D5) may have occurred in California due to current use as an alternative to perchloroethylene in dry cleaning (Ceballos et al., 2021). Quantification of these compounds can be challenging, particularly in water, with careful attention to sample collection and analysis needed. Therefore, pilot studies to evaluate concentrations in various matrices of ecosystems with higher levels of potential contamination, and/or pathways, are recommended.

We provide these recommendations in support of the Water Boards CEC Program. Further priorities are likely to emerge as the Water Boards refine and implement a risk screening approach based on the one outlined in the present report. Our demonstration of this screening method, informed by the chemical and use classes of individual contaminants, represents a pragmatic means of evaluating risks of CECs to inform monitoring and management actions.

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Appendices A - E

Appendix A: Occurrence data, toxicity thresholds, and risk quotients for key CEC classes in California waters

Table A.1 - A.9 (Supporting Document)

Appendix B: Occurrence data, toxicity thresholds, and risk quotients for key CEC classes in California sediment

Table B.1 - B.9 (Supporting Document)

Appendix C: Occurrence data, toxicity thresholds, and risk quotients for key CEC classes in California biota

Table C.1 - C.9 (Supporting Document)

Appendix D: ECHA REACH registration documentation used for risk screening

Table D (Supporting Document)

Appendix E: Stakeholder Guidance

(Supporting Document)