

UNDERSTANDING MICROPLASTIC LEVELS, PATHWAYS, AND TRANSPORT

in the San Francisco Bay Region

Authors

Rebecca Sutton, SFEI

Amy Franz, SFEI

Alicia Gilbreath, SFEI

Diana Lin, SFEI

Liz Miller, SFEI

Meg Sedlak, SFEI

Adam Wong, SFEI

Carolynn Box, 5 Gyres

Rusty Holleman, University of California-Davis
Center for Watershed Science

Keenan Munno, University of Toronto

Xia Zhu, University of Toronto

Chelsea Rochman, University of Toronto

Design and Layout

Ruth Askevold, SFEI

Ila Shimabuku, SFEI

Funded By

The Gordon and Betty Moore Foundation

With Additional Support From

Patagonia

City of Palo Alto

East Bay Municipal Utility District

Virginia Wellington Cabot Foundation

California Ocean Protection Council

San Francisco Bay Regional Monitoring Program for
Water Quality



October 2019
SFEI-ASC Publication #950

SFEI | San Francisco
Estuary Institute



5 GYRES
SCIENCE TO SOLUTIONS



Acknowledgements

The San Francisco Bay Microplastics Project would not have been possible without the enthusiastic support of many people. We are grateful to all who provided insights, advice, and funding in support of this effort, and would especially like to thank the following individuals and agencies:

The Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) for providing a foundation for much of this work, and Jay Davis (RMP Director), Melissa Foley (RMP Manager), and Don Yee (RMP Quality Assurance Officer) for their guidance and review of this document;

The RMP Microplastic Workgroup participants and its expert advisors: Anna-Marie Cook, Kara Lavender Law, and Chelsea Rochman;

RMP Emerging Contaminants Workgroup expert advisor: Kelly Moran;

The RMP Small Tributaries Loading Strategy team: Jim Scanlin (Alameda Countywide Clean Water Program), Bonnie de Berry (San Mateo Countywide Water Pollution Prevention Program), Lisa Sabin (Santa Clara Valley Urban Runoff Pollution Prevention Program), Lucile Paquette (Contra Costa Clean Water Program), Chris Sommers (EOA, Inc.), and Richard Looker and Jan O'Hara (San Francisco Bay Regional Water Quality Control Board);

Wastewater agencies and staff: Karin North, Jim Stuart, Dominic Hoang, Kyle Carbajal, Robert Hara, and the City of Palo Alto; Simret Yigzaw, Eric Dunlavey, and the City of San Jose; Cameron Kostigen-Mumper and the City of Sunnyvale; Autumn Cleave, Stephen Navarra, Laura Pagano, Karrie Ving, and the San Francisco Public Utilities Commission; Jason Mitchell and the East Bay Municipal Utility District; Shahed Abbasi, Angie Barumen, Jackie Zipkin, and the East Bay Dischargers Authority; Tri Nguyen, Mary Lou Esparza, Douglas Little, Melody LaBella, Lori Schectel, and Central Contra Costa Sanitary District; and Giti Heravian and Fairfield-Suisun Sewer District;

The captains and sailors of the Derek M. Baylis and the San Francisco Baykeeper vessel, as well as Jody Watt (Wylie Charters), and Ian Wren, Sienna Courter, and Geoff Potter (San Francisco Baykeeper), all instrumental in surface water sample collection (authorized by California Department of Fish and Wildlife Scientific Collection Permit SCP-12364);

National Oceanic and Atmospheric Administration staff including Max Delaney, Sophie De Beukelaer, Lilli Ferguson, and Kate Bimrose;

Moss Landing Marine Laboratories scientists who collected sediment and prey fish, Rusty Fairey and Marco Sigala (Scientific Collection Permit SC-008521);

All who assisted in the project's field work, with special thanks to Steph Whyte (formerly 5 Gyres), Erika Delemarre (5 Gyres), Doug George (UC Davis), and Jane Sedlak for their assistance with completion of surface water sampling; additional thanks to Erika Delemarre for her support on data organization and writing related to the Surface Water Chapter;

The microplastics analytical team at the Rochman Laboratory at the University of Toronto, under the supervision of Chelsea Rochman, Xia (Alice) Zhu, and Keenan Munno: Hannah DeFrond, Jelena Grbic, Edie Guo, Lara Werbowski, Jacqueline Bikker, Aimee Huntington, Brenda Lim, Hayley McIlwraith, Annissa Ho, Cassandra Sherlock, Natasha Klasios, Cole Brookson, Antonino Calarco, Charlotte Hung, Anna Lisa, Tina Wu, Lauren Ead, and Maheen Arshad;

Edward Gross for extensive assistance in particle tracking simulations and discussions of hydrodynamics, and John Largier for guidance on coastal dynamics in the San Francisco Bight;

Shira Bezalel (SFEI), and Andy Miller and Robin Moore (Plus M Productions) for expert photographs; and

The Gordon and Betty Moore Foundation program officers: Rachel Strader (former), Genny Biggs, and Sara Bender.

The San Francisco Bay Microplastics Project was primarily funded by the Gordon and Betty Moore Foundation, with additional financial support from the Regional Monitoring Program for Water Quality in San Francisco Bay, Patagonia, the Virginia Wellington Cabot Foundation, East Bay Municipal Utility District, the City of Palo Alto, and the California Ocean Protection Council.

Executive Summary

The San Francisco Bay Microplastics Project

Microplastics (particles less than 5 mm) are ubiquitous and persistent pollutants in the ocean and a pervasive and preventable threat to the health of marine ecosystems. Microplastics come in a wide variety of shapes, sizes, and plastic types, each with unique physical and chemical properties and toxicological impacts. Understanding the magnitude of the microplastics problem and determining the highest priorities for mitigation require accurate measures of microplastic occurrence in the environment and identification of likely sources.

To develop critical baseline data and inform solutions, the San Francisco Estuary Institute and the 5 Gyres Institute have completed the first comprehensive regional study of microplastic pollution in a major estuary. This project supported multiple scientific components to develop improved knowledge about and characterization of microparticles and microplastics in San Francisco Bay and adjacent National Marine Sanctuaries, with the following objectives:

1. Contribute to the development and standardization of sample collection and analysis methodology for microplastic research.
2. Determine a baseline for future monitoring of microplastics in San Francisco Bay surface water, sediment, and fish, and in ocean waters outside the Golden Gate.
3. Characterize pathways by which microplastics enter the Bay, including urban stormwater and treated wastewater effluent.
4. Investigate the contribution of Bay microplastics to the adjacent National Marine Sanctuaries through computer simulations.
5. Communicate to regional stakeholders and the general public through meetings and educational materials.
6. Facilitate evaluation of policy options for San Francisco Bay, with recommendations on source reduction.

This document presents the findings of this three-year project. A companion document, "San Francisco Bay Microplastics Project: Science-Supported Solutions and Policy Recommendations," has been developed by 5 Gyres using the findings of this study (Box and Cummins, 2019).

Our findings

In this report, we have distinguished between **microparticles**, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and **microplastics**, which are confirmed to be plastic through Raman or Fourier Transform Infrared (FTIR) spectroscopy. This distinction is necessary because it was not possible to examine every microparticle via spectroscopy to confirm particle composition. Moreover, for some particles, clear spectra were not easy to obtain.

Microparticles and microplastics come in a range of sizes, and the lower size limit of microplastics within a sample is operationally defined by the mesh, sieve, or filter pore size used in sample collection and analysis. We quantified microparticles using sieves with a mesh size of 0.125 mm for stormwater, wastewater, and sediment. For surface water, the standard manta trawl sample collection method typically captures particles greater than 0.355 mm, whereas for prey fish, it is possible to collect far smaller particles. When smaller size fractions are included, the overall number of microparticles and microplastics in a sample can increase significantly. To facilitate comparisons among these different types of samples, particles were grouped into uniform size categories during laboratory analysis.

STORMWATER

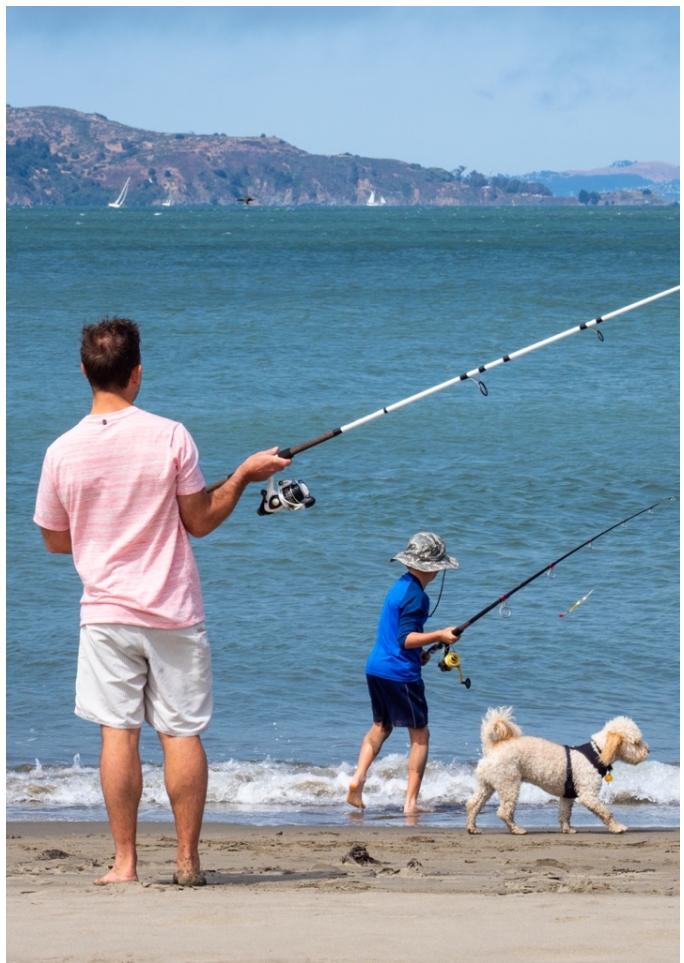
We measured microparticles and microplastics in stormwater from 12 small tributaries comprising 11% of the watershed drainage area to San Francisco Bay (6% of total flow to Bay). These tributaries varied in urban and non-urban land uses and were distributed across the region. Microparticles were identified in stormwater from all 12 small tributaries, which discharged between 1.3 and 30 microparticles per liter. Fragments (59%) and fibers (39%) constituted nearly all microparticles sampled.

Nearly half of the particles from field samples were black fragments that had a distinctive rubbery texture when handled with tweezers. Spectroscopic analysis and secondary characteristics suggested these particles may be synthetic or natural rubber. This identification is not definitive, as other techniques beyond the scope of this project are needed to confirm the particle composition. For purposes of this report, these polymers are considered a type of plastic, a common approach in the field of microplastics. The literature suggests that one potential source of these particles is vehicle tire wear.

Using an existing stormwater model developed for other contaminants, we estimated the annual discharge of microparticles via stormwater from small tributaries to be 11 trillion microparticles to the Bay. Approximately two thirds of these microparticles were estimated to be plastic, yielding an estimated annual discharge of 7 trillion microplastics per year. This

Executive Summary

estimate of microplastic load is approximately 300 times greater than the estimated annual discharge from all wastewater treatment plants discharging into San Francisco Bay.



TREATED WASTEWATER

We measured microparticles and microplastics in treated wastewater from eight wastewater treatment facilities that represent approximately 70% of the treated effluent flow discharged to San Francisco Bay. These facilities are geographically distributed, vary in effluent treatment capacity, and employ a range of treatments. Microparticles were identified in effluent from all eight facilities, which discharged an average of 0.063 microparticles per liter. Fibers were the most frequently identified type. While 19% of the fibers were unmistakably plastic, another 50% were clearly manufactured due to the presence of dyes and coloring agents, but could not be definitively identified as plastic or non-plastic. Fragments were the second most abundant shape, and of those that underwent spectroscopy, 54% were identified as plastic, with most being polyethylene (31%).

In aggregate, approximately 91 million microparticles per day are discharged by

the eight facilities. Facilities employing advanced treatment including dual media filtration had lower microparticle concentrations than facilities without this additional treatment, suggesting that enhanced treatment may reduce microparticles as well as other pollutants. Assuming similar discharges among the remaining facilities, approximately 130 million microparticles are discharged per day to the Bay in treated wastewater effluent, or approximately 47 billion microparticles annually, of which 17 billion are estimated to be plastic. This is substantially lower than the estimate developed for the annual microplastic load from the small tributaries surrounding the Bay.

SURFACE WATER

We collected surface water samples at 17 monitoring sites throughout San Francisco Bay and 11 monitoring sites within Monterey Bay, Cordell Bank, and Greater Farallones National Marine Sanctuaries. Each site was sampled twice, once during the dry season and again during the wet season following rainfall events.

Microparticles were identified in all manta trawl samples, with higher abundances overall in the Bay than in the adjacent marine sanctuaries. Levels of microparticles in the Bay are some of the highest observed globally to date. The dominant particle type was fibers, followed by fragments, with 53% of fibers and 87% of fragments identified as plastic. The composition of many fibers could not be determined, though the presence of dyes and coloring agents indicated that they were anthropogenic in origin.

Apart from fibers, polyethylene and polypropylene fragments, polystyrene foams, and polyethylene and polypropylene films made up a majority of the microparticles that underwent spectroscopy. These polymer and particle types may be linked to the breakdown of single-use plastic items, packaging, and plastic bags. Polyethylene beads were also identified in the surface waters, possibly linked to microbeads found in personal care and cleaning products.

Wet season Bay samples contained statistically higher concentrations of microparticles compared to dry season samples, suggesting that wet weather may mobilize microplastics from the surrounding watershed. Within the Bay, the wet season average abundance for non-fiber particles was 520,000 microparticles/km², while the average for fibers was 580,000 microfibers/km². A statistically significant seasonal effect was not observed in the sanctuaries, likely due to the low abundance of microparticles observed.

Manta trawl sample collection is not an ideal method for capturing fibers. Sampling methods designed to collect more representative levels of fibers, as well as especially small particles, were deployed at some sites to test their effectiveness. However, field blank samples collected and analyzed to monitor background contamination for these sampling techniques had high levels of microparticles, especially fibers. This suggests the need for sampling larger volumes and provides further evidence of the impacts of background contamination from fibers on data quality.

SEDIMENT

We collected sediment samples at 20 sites, including 18 within San Francisco Bay and two in Tomales Bay, which has minimal urban influence. Sites were selected to characterize microplastic concentrations near discharges of stormwater and wastewater in the nearshore

Executive Summary

"margins" of the Bay, in open portions of the Bay, and in a less urban reference area (Tomales Bay).

Microparticles were identified in sediment from all 20 sites. Fibers, followed by fragments, were the most abundant type of microparticles in Bay sediment, with detected concentrations ranging between 1 and 49 microfibers per gram dry weight (dw), and between 0.1 and 11 non-fiber microparticles (including fragments, films, spheres, and foams) per gram dw. The highest concentrations of microparticles were measured in Lower South Bay, which is strongly influenced by wastewater and urban stormwater discharges. Concentrations at the reference site, Tomales Bay, were among the lowest observed in the study.

Black fragments that had a rubbery texture were frequently detected in sediment samples. Spectroscopy was unable to identify the composition; however, based on secondary characteristics, these particles were similar to particles that had been previously identified as rubber by FTIR spectroscopy. These particles were also similar to the black, rubbery fragments that were abundant in stormwater, suggesting that stormwater is an important pathway for microparticles to reach Bay sediment, and that inputs from tire wear and perhaps use of recycled tires (e.g., artificial turf) may also merit further investigation.

Microparticle and microplastic concentrations in the Bay sediment were higher than those reported in the majority of other regions around the globe.

PREY FISH

To evaluate the uptake of microplastics into the food web, two prey fish species, topsmelt and Northern anchovy, were sampled at six sites in the Bay, as well as two sites in a less urban reference area (Tomales Bay). At each site, approximately 10 fish of each species were collected, and the digestive tracts were analyzed for microparticles and microplastics.

Microparticle levels in fish from San Francisco Bay were higher than levels in fish from Tomales Bay. Fibers were particularly abundant; while most fibers were dyed and therefore produced by people, few could be identified conclusively as plastic. At least 38% of fish from the Bay had consumed microparticles.

The estimated average number of microplastics was between 0.2 and 0.9 non-fiber microplastics per fish and between 0.6 and 4.5 plastic fibers per fish. While fibers were detected in all fish from the Bay regardless of species, non-fiber microparticles were more frequently detected in topsmelt compared to anchovies. The microplastic counts and detection frequencies in the Bay were comparable to counts reported in many other locations.

Executive Summary

While toxicological evaluation was not a part of this study, these results indicate that microplastics are entering Bay food webs. Microplastics have been shown to transfer up food chains and cause adverse effects in fish, but the magnitude and types of effects are difficult to predict because of the diversity of microplastic morphologies and compositions. There is a need for further ecotoxicological studies that evaluate the effects of microplastics at environmentally relevant concentrations. However, even with more ecotoxicological data, establishing risk thresholds will be challenging given the heterogeneous nature of this class of contaminants.

TRANSPORT MODEL

A novel three-dimensional hydrodynamic transport model was developed to simulate microparticle and microplastic movement in the Bay and the adjacent marine sanctuaries. The model was validated and accurately captured water surface elevations, velocities, and salinity. This model is unique in its spatial coverage from small scale (e.g., meters) in sloughs and mud flats within the Bay to shelf-scale (e.g., tens of kilometers) dynamics in the coastal ocean. The transport model includes the effects of wind and tides, as well as inflows from stormwater, wastewater, and freshwater from the Sacramento-San Joaquin River Delta.

The model incorporated estimated microparticle and microplastic loads from stormwater and wastewater, and simulated particle trajectories throughout the Bay and into the coastal ocean. The rising and settling characteristics of particles were estimated based on laboratory measurements of chemical composition, shape, and size.

Model output was analyzed to estimate spatial distributions of predicted surface water concentrations and potential deposition to sediment, as well as time scales for particles to be exported from the Bay. The fate of microplastics was found to be highly sensitive to particle buoyancy, and even minimal sinking rates led to retention of particles within the Bay. The model indicated that, for microplastics originating in San Francisco Bay, only buoyant particles were likely to travel any significant distance beyond the Golden Gate and into the nearby National Marine Sanctuaries. The transport model and the manta trawl particle abundance data were in good agreement, showing that the average abundance of particles was higher in the Bay than in the coastal ocean. Good agreement was also observed between the model-predicted microparticle abundances near the bottom of the Bay and measured sediment concentrations, showing the greatest abundance of microparticles in Lower South Bay.

CONCEPTUAL MODEL AND DATA SYNTHESIS

We refined a conceptual model of major pathways of microplastic pollution for San Francisco Bay, including a comprehensive review of likely sources to urban stormwater runoff and treated wastewater discharges. This study synthesis indicated identification of specific plastic

Executive Summary

polymers is essential for pinpointing potential sources of microplastics, as well as predicting the movement of these particles within and through estuarine ecosystems.

Comparison of urban stormwater and wastewater indicated that beyond the large differences in estimated loads to the Bay, there were also considerable differences in relative proportions of different polymers. The large contribution of black, rubbery fragments was a dominant feature in urban stormwater samples. Meanwhile, wastewater samples indicated influence from multiple sources, including plastics used in textiles (acrylic, polyester), as well as microbeads in personal care and cleaning products and microplastics likely derived from the breakdown of larger single-use items (polyethylene).

Comparison of surface water and sediment samples likewise indicated that polymer type was generally the most influential variable in determining whether relative contributions of different types of microplastics were preferentially concentrated in one matrix or the other. Buoyant polymers and foams were more likely to be found in surface water, while denser particles were often found in sediment.

Key data gaps for San Francisco Bay remain, including additional information on the sources and pathways of microplastics, the exposure of Bay aquatic organisms and associated risk for adverse impacts, more comprehensive information resulting from essential improvements in methodology, and the effects of current and future solutions implemented to reduce microplastic pollution.

LESSONS LEARNED: RECOMMENDED BEST PRACTICES FOR FUTURE STUDIES

The field of microplastics pollution is in its infancy, and there are not yet widely accepted standards for sample collection, laboratory analysis, quality assurance/quality control (QA/QC), or reporting of microplastics in environmental samples. This project included the development of recommended best practices for collection, processing, analysis, and reporting microplastics in environmental media. We recommend factors to consider in microplastic study design, particularly in regards to site selection and sampling methods. We also highlight the need for standard QA/QC practices such as collection of field and laboratory blanks, use of methods beyond microscopy to identify particle composition, and standardized reporting practices, including suggested vocabulary for particle classification.

Table of Contents

<i>Acknowledgements</i>	ii
<i>Executive Summary</i>	iv
Introduction	1
<i>Microplastics: An emerging concern</i>	2
<i>What are microplastics?</i>	3
<i>Current state of scientific knowledge</i>	8
<i>Microplastics as a concern in San Francisco Bay</i>	11
<i>The San Francisco Bay Microplastics Project</i>	13
<i>Looking forward</i>	16
<i>Glossary of key terms specific to this report</i>	17
<i>References</i>	18
Stormwater	27
<i>Highlights</i>	28
<i>Objectives</i>	29
<i>Methods</i>	30
<i>Results</i>	40
<i>Discussion</i>	50
<i>Conclusions</i>	59
<i>References</i>	60
Wastewater	65
<i>Highlights</i>	66
<i>Objectives</i>	68
<i>Methods</i>	70
<i>Results</i>	80
<i>Discussion</i>	92
<i>Conclusions</i>	102
<i>References</i>	103

Surface Water.....	108
<i>Highlights.....</i>	109
<i>Objectives.....</i>	110
<i>Methods.....</i>	112
<i>Results.....</i>	128
<i>Discussion.....</i>	145
<i>Conclusions.....</i>	156
<i>References.....</i>	158
Sediment	167
<i>Highlights.....</i>	168
<i>Objectives.....</i>	169
<i>Methods.....</i>	170
<i>Results.....</i>	175
<i>Discussion.....</i>	185
<i>Conclusions.....</i>	193
<i>References.....</i>	194
Prey Fish	200
<i>Highlights.....</i>	201
<i>Objectives.....</i>	202
<i>Methods.....</i>	204
<i>Results.....</i>	209
<i>Discussion.....</i>	218
<i>Conclusions.....</i>	231
<i>References.....</i>	233
Transport Model.....	240
<i>Highlights.....</i>	241
<i>Objectives.....</i>	243
<i>Methods.....</i>	244
<i>Results.....</i>	252
<i>Discussion.....</i>	277

<i>Conclusions</i>	280
<i>References</i>	281
Synthesis.....	283
<i>Highlights</i>	284
<i>Objectives</i>	285
<i>Methods</i>	286
<i>Microplastics sources and pathways in the San Francisco Bay Area</i>	287
<i>Microplastics transport and fate in San Francisco Bay and adjacent ocean environment</i>	295
<i>Key data gaps remain for microplastics in San Francisco Bay</i>	302
<i>Conclusions</i>	308
<i>References</i>	309
Lessons Learned.....	316
<i>Introduction</i>	317
<i>Study design: Site selection</i>	321
<i>Study design: Field sampling</i>	327
<i>Laboratory analysis</i>	335
<i>Reporting results</i>	341
<i>Conclusions</i>	347
<i>References</i>	348
Appendices.....	356
<i>Stormwater</i>	356
<i>Wastewater</i>	359
<i>Surface Water</i>	360
<i>Sediment</i>	382
<i>Prey Fish</i>	387

CHAPTER

1

INTRODUCTION

by Liz Miller



Microplastics: An emerging concern

Microplastics, tiny pieces of plastic smaller than 5 millimeters, have been recognized as a pervasive and preventable global threat to the health of aquatic ecosystems. Microplastics are ubiquitous pollutants in the ocean (Farady, 2019), and have been observed in every setting examined, even remote locations such as the Arctic (Bergmann et al., 2019; Lusher et al., 2015) and deep sea sediment (Bergmann et al., 2017; Van Cauwenberghe et al., 2013).

Aquatic organisms at every trophic level are exposed to microplastics (Besseling et al., 2019; de Sá et al., 2018; Wright et al., 2013), but the health risks posed are uncertain due to variations based on the plastic type, shape, species, and dose concentration (Group of Chief Scientific Advisors, 2019). Ingested microplastics can impact the biochemical and physiological processes of many different types of animals (Burns and Boxall, 2018; Foley et al., 2018; Prokić et al., 2019; Wright et al., 2013; Shaoliang Zhang et al., 2019). Microplastics can also expose organisms to potentially harmful chemicals, especially plastic additives such as flame retardants, plasticizers, or dyes (Fries et al., 2013; Rochman et al., 2019). Microplastics and harmful plastic chemical ingredients and additives can also be transferred up food chains (Chagnon et al., 2018; Farrell and Nelson, 2013; Mattsson et al., 2017; Setälä et al., 2014; Tosetto et al., 2017).

Once introduced into the environment, microplastics become a highly persistent part of marine ecosystems (Farady, 2019). The half-lives of microplastics are not known with certainty due to the broad variety of polymers in use and varying environmental conditions; however, it is widely believed that these particles are very resistant to environmental (bio)degradation and will remain stable in the environment long after their release (European Chemicals Agency, 2019a, 2019b; Science Advice for Policy by European Academies, 2019).

Plastics are ubiquitous in modern society. Plastic is a worldwide trillion-dollar industry (Worm et al., 2017), and annual plastic production is projected to increase from nearly 340 million metric tons in 2016 to over 1.1 billion metric tons by 2050 (United Nations Environment Programme, 2019). Recovery and recycling, however, lag behind; as of 2015, only around 9% of all plastic waste ever generated had been recycled (Geyer et al., 2017). Approximately 5–13 million metric tons of plastics are estimated to end up in marine waters each year (Jambeck et al., 2015). Assuming no major changes to waste management infrastructure, the total amount of plastic waste available to enter the marine environment could increase to 150 million metric tons or more by 2025 (Worm et al., 2017).

The sources and generation of micro-sized pieces of plastics, the pathways by which these tiny particles reach estuarine and marine environments, their transport, distribution, and fate

Introduction

within these ecosystems, and the levels to which they are taken up into the food web are complex and not well understood. Numerous independent scientific efforts are underway to begin to characterize this class of emerging, unregulated contaminants and address the many scientific questions and data gaps that have been identified.

In 2015, a preliminary screening study of microplastics in the San Francisco Bay indicated that levels of contamination might be higher than observed in other large, urbanized water bodies (Sutton et al., 2016). These findings suggested the need for a comprehensive regional study to characterize microplastics in the Bay, the potential pathways by which they enter the Bay, and the circulation patterns that drive spatial variation within the Bay and deliver microplastics to the ocean. The scientific understanding generated by such an effort is critical to informing effective solutions for plastic pollution. California is leading the nation in plastic pollution reduction efforts, and passed the first statewide single-use carry-out plastic bag ban (California Department of Resources Recycling and Recovery (CalRecycle), 2019). The state has an enforceable goal of zero trash in any ocean waters, bays, or rivers by 2030 (California State Water Resources Control Board, 2017), but trash is defined in the regulation as debris larger than 5 mm, and does not include microplastics. In the Bay Area, many communities with watersheds that drain directly to the Bay are passing local ordinances to ban single-use plastic items.

Based on the increasing amount of plastic debris collecting in aquatic habitats, current policies are inadequate to address the growing and widespread threat of microplastic pollution. Data are therefore essential to understanding and minimizing the impacts of microplastics on San Francisco Bay and the adjacent ocean environment.

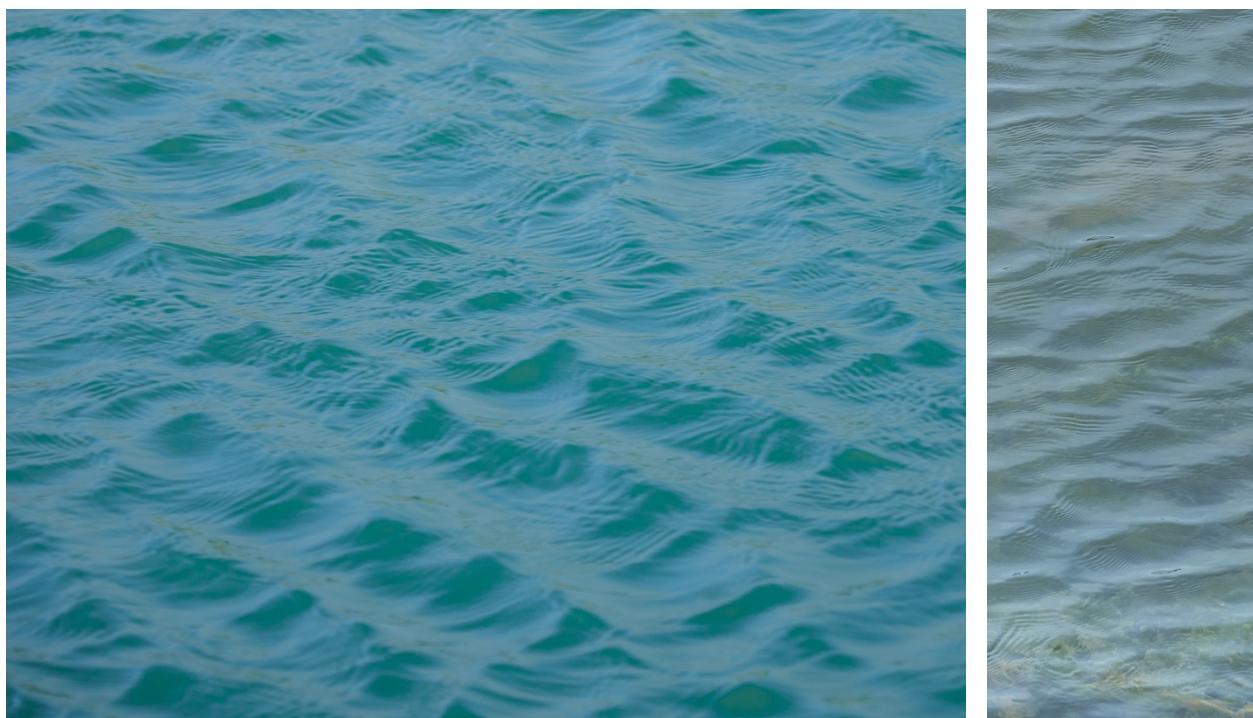
What are microplastics?

Microplastics are commonly defined as plastic particles smaller than 5 mm in at least one external dimension (Burns and Boxall, 2018; Mausra and Foster, 2015; Thompson et al., 2009). The lower size limit of microplastics within a sample is operationally defined by the mesh, sieve, or filter pore size used in sample collection and analysis. For example, manta trawl collection of surface water samples typically captures particles greater than 0.355 mm, which corresponds to the mesh size of the net, the collection end of the sampling apparatus, and the sieves used. Some particles smaller than this operational lower size limit may be captured due to aggregation or association with larger materials. In contrast, particles like fibers, which are long and thin, may escape capture depending on their orientation and movement relative to the net (Barrows et al., 2017; Covernton et al., 2019; Kang et al., 2015). Particles smaller than 0.0001 mm (or 0.1 μm) are generally defined as nanoparticles (Thompson et al., 2009), and have received little study.

Introduction

The term *micoplastics* encompasses a wide variety of plastic materials, each with unique physical and chemical characteristics (Figure 1.1). The term *plastic* is quite broad and generally refers to any synthetic water-insoluble polymer (typically of petrochemical origin) that can be molded on heating and manipulated into various shapes designed to be maintained during use (Burns and Boxall, 2018). Common polymers are polyethylene (PE), polypropylene (PP), polystyrene (PS), nylon (polyamide), polyethylene terephthalate (PET, or polyester in the case of fibers), acrylic (polyacrylonitrile and related polymers), and polyvinyl chloride (PVC), among others (Boucher and Friot, 2017; Hidalgo-Ruz et al., 2012). Rubber, whether natural (isoprene) or synthetic (e.g., styrene-butadiene), is also considered a plastic. Many plastics contain chemical additives to impart desirable characteristics, including flame retardants, plasticizers, and dyes. Dyes and coloring agents can pose a particular problem during sample analysis.

Microplastics are frequently described as primary or secondary, to distinguish between the general types of sources from which they originate. Primary microplastics are designed and manufactured to be small for a variety of uses, including pellets for plastic production (e.g., "nurdles"), abrasive blasting, paints and adhesives, agricultural applications, and for use in personal care products (European Chemicals Agency, 2019a). Primary microplastics are released to the environment as a consequence of the use of products that intentionally contain or release them via discharge to wastewater treatment plants, spills, and weathering during maintenance and use (Boucher and Friot, 2017). In contrast, secondary microplastics originate from the degradation of larger plastic items, regardless of when this breakdown occurs (Science Advice for Policy by European Academies, 2019).



Microparticles vs. microplastics

In this report, we have distinguished between **microparticles**, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and **microplastics**, which have been confirmed to be plastic through Raman or Fourier Transform Infrared (FTIR) spectroscopy. This is an important and necessary distinction because—in this study and as a general rule—some particles that visually appear to be plastic are anthropogenic, but made of other materials (e.g., glass, metal, cotton), and not all potentially plastic particles can be confirmed as plastic, either due to resource constraints or analytical challenges.

The first studies in the field of microplastics relied on identification using only visual techniques, such as microscopy. Studies using additional, advanced laboratory techniques demonstrated that a portion of these visually identified particles were erroneously characterized as plastics, an issue that becomes more likely with decreasing particle size (Hidalgo-Ruz et al., 2012).

In more recent studies, laboratory analysis of microplastics typically involves microscopy as a first step to identify particles that visually appear to be plastic, followed by a confirmation step to establish that the particle is, in fact, plastic, and to determine what type. In the present study, Raman or FTIR spectroscopy were applied to a subset of particles to measure the ability of the analysts in picking anthropogenic particles and to identify the composition of each via matches in standard and custom-built spectral libraries (Munno et al., in review) to get an idea of the types of materials that are in our samples. In many cases, it was possible to identify whether a particle was made up of, for example, polyethylene, polyester, or a non-plastic material, like cotton.

Unfortunately, spectroscopic identification of the chemical composition of particles is not always possible. The presence of a chemical such as a dye can mask the spectrum of the underlying material and prevent identification. In these cases, the microparticle could be identified as being of anthropogenic origin (i.e., manufactured) based on the presence of the dye, but not necessarily classified as plastic. In this report, we refer to such particles as **anthropogenic unknown**.

In other cases, sufficient spectral information allowed confirmation that the dyed particle was plastic, but could not indicate the specific type of plastic. In this report, we refer to such particles as **anthropogenic synthetic**. The presence of dyes can also reduce the ability to distinguish between non-plastic materials for some particles, leading to the additional non-plastic category **anthropogenic cellulosic** (e.g., cotton, rayon, modal, or Lyocell).

Therefore, the use of spectroscopy or other chemical identification methods is essential for accurately characterizing microplastic pollution. Given the high number of particles detected in

some environmental samples, it is time- and resource-prohibitive to conduct spectroscopy on each one, and thus subsampling often becomes necessary. The terms used in this report, microparticles vs. microplastics, are used carefully to provide transparency around the level of certainty regarding particle composition.

Morphologies of microparticles and microplastics

Microparticles and microplastics come in a range of shapes or morphologies (Figure 1.1). Particles are commonly classified into different shapes or categories, which in some cases provide insights as to the potential source of individual particles (Free et al., 2014; McCormick et al., 2014; Rochman et al., 2019). In this study, we use the following categories:

- ◆ Fragment — irregularly-shaped particle; may come from the breakdown of larger plastic debris;
- ◆ Sphere/Pellet — hard, rounded, or spherical particle; may come from pelletized pre-production material for plastic or microbeads intentionally added to consumer products;
- ◆ Film — thin plane; may come from the breakdown of film-like debris, such as plastic bags and wraps;
- ◆ Foam — lightweight, sponge-like particle; may come from the breakdown of foam plastic debris; and
- ◆ Fiber — thin or fibrous particle (sometimes also referred to as **microfibers**); may come from textiles as well as fishing gear and cigarette filters.

Preliminary work characterizing samples collected for this project led to the identification of an additional particle type category, fiber bundle, consisting of a number of fibers that cannot be disentangled. Individual fibers within a bundle may be of similar or differing chemical composition and color. Because there were relatively few fiber bundles observed, this category was grouped with fibers in many of the analyses in this report.

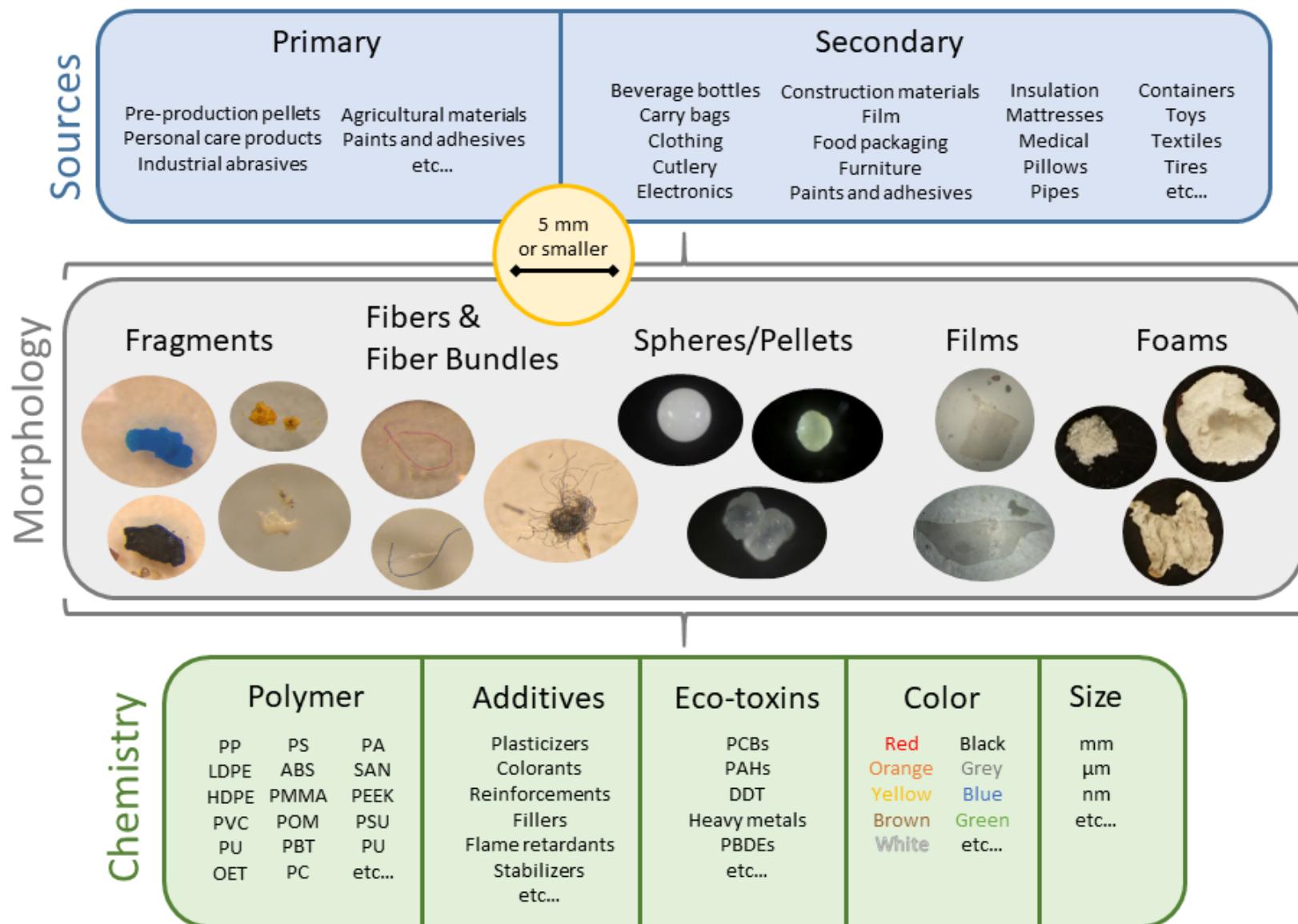


Figure 1.1. Microplastics are a diverse class of contaminants. Figure adapted from Rochman et al., 2019; Tanaka and Takada, 2016; and Wessel et al., 2016.

Current state of scientific knowledge

Prevalence of microplastics in the environment

The world's first fully synthetic polymer, Bakelite or polyoxybenzylmethyleneglycolanhydride, a thermosetting phenol formaldehyde resin, was invented in 1907 by Leo Baekeland, who also first coined the term *plastic*, after the Greek word *plastikos*, meaning moldable (Worm et al., 2017). Baekeland's invention—an inexpensive, nonflammable, and versatile material—was slow to pick up steam, but by the 1950s, mass production of plastics had transformed every aspect of human material consumption. Since the 1960s, plastic production has increased by approximately 8.7% annually (Jambeck et al., 2015). In 2017, 348 million metric tons of plastic were produced worldwide (Plastic Europe, 2018), with future projections indicating plastic production is likely to double by 2030 (Azoulay et al., 2019).

However, many of the properties of plastics that make them attractive for a myriad of applications also pose new challenges. The stability of plastic polymers means that plastics are extremely persistent and resistant to degradation. Only around 9% of plastic ever generated has been recycled, whereas almost 80% has accumulated in landfills or the natural environment (Geyer et al., 2017). Plastic litter from terrestrial sources accounts for approximately 80% of the plastics found in marine litter (Cole et al., 2011). Globally, 5 to 13 million tons are estimated to enter the ocean every year (Jambeck et al., 2015). Estimates of plastics in the Great Pacific Garbage Patch show a doubling in concentration in the last decade (Lebreton et al., 2018). Plastic litter, especially microplastics, is exceedingly difficult to remove once introduced into the environment. Although recent policy actions may reduce some plastic use (e.g., straw, bag, and food-packaging bans) and discharge (e.g., trash control measures), the reservoir of plastics in use today is large, and, without significant management actions, is likely to result in continued discharge to and accumulation in the environment, including San Francisco Bay.

With current plastics use, microplastics are continuously produced, whether intentionally or unintentionally, and move into the environment. The distribution of microplastics in ecosystems can be influenced by a range of processes, including plastic production and use, waste and wastewater management, runoff, infiltration, freshwater flow, wind action, ocean currents, and the movement of animals and humans across and between ecosystems (Cole et al., 2011). The physical and chemical properties of individual microplastics also control how they are transported, their retention times in environmental compartments, and where they will ultimately end up (Rochman et al., 2019).

Microplastics have been detected across terrestrial and aquatic systems around the world, from air (Allen et al., 2019; Bergmann et al., 2019; Dris et al., 2016) to surface waters (Barrows

Introduction

et al., 2018; Cole et al., 2011; Law, 2017; van Sebille et al., 2015) to deep-sea sediments (Bergmann et al., 2017; Van Cauwenbergh et al., 2013). They are now widely recognized as globally ubiquitous. Fibers appear to be especially pervasive (Barrows et al., 2018), and may be frequently underestimated due to their ability to slip through nets (Barrows et al., 2017; Covernton et al., 2019; Kang et al., 2015). Models estimating the order of magnitude of past, present, and future microplastic concentrations in the environment based on global plastic production data predict a 50-fold increase in microplastics in ocean surface water by 2100 compared to the present-day concentrations, and adverse ecological effects in aquatic life by the second half of the 21st century, if they are not occurring already (Everaert et al., 2018). However, the sources, pathways, transport, and fate of microplastics in the environment are complex, and research is still in its infancy, with much remaining uncertain.

Effects on aquatic life

In general, there is a dearth of studies identifying the impacts of microplastic exposure to organisms at ecologically relevant concentrations. However, the literature indicates a clear risk of adverse impacts to Bay species if microplastic concentrations increase in the future.

Microplastics can affect aquatic life through both physical and chemical mechanisms. In general, microplastics may be ingested (either directly or indirectly due to their presence in ingested prey) or taken up through the gills or other permeable tissues. Microplastics may physically block feeding structures, impair respiration by clogging gills, or cause lacerations (Burns and Boxall, 2018; Foley et al., 2018; Prokić et al., 2019; Wright et al., 2013; Shaoliang Zhang et al., 2019). The chemical effects of microplastics depend on the plastic polymers; the presence of chemical additives such as flame retardants, plasticizers, and dyes; and other organic pollutants sorbed to the microplastic. Once in tissues, microplastics and the chemicals they contain may elicit an adverse immune or stress response by causing production of reactive oxygen species, inflammation, and cell damage (Adam et al., 2019; Burns and Boxall, 2018; Duis and Coors, 2016; European Chemicals Agency, 2019a, 2019b; Foley et al., 2018; Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, 2015; Prokić et al., 2019; Science Advice for Policy by European Academies, 2019; Wright et al., 2013; Shaoliang Zhang et al., 2019). Smaller ingested microplastic particles may also translocate out of the gut to other tissues. Once in prey species, microplastics and the potentially harmful chemicals they contain can be transferred to higher trophic levels (Mattsson et al., 2017; Setälä et al., 2014; Tusetto et al., 2017).

Microplastics have been associated with impaired feeding, lower growth rates, decreased respiration, altered behavior, impaired reproduction, and mortality in many different organisms (Burns and Boxall, 2018; Foley et al., 2018; Prokić et al., 2019; Wright et al., 2013; Shaoliang Zhang et al., 2019). However, there is wide variation in the type and dose-

Introduction

dependence of effects reported in the literature, indicating a great deal of uncertainty and variation in toxicity and sensitivity. The observed variability in type and magnitude of effects is likely due to inconsistent methods (including exposures and analyses); species differences; differences in microplastic sources, shapes, sizes, and chemistry; and whether the exposure to microplastics occurs alone or in conjunction with exposure to other contaminants. Irregularly shaped microplastics (fragments and fibers), especially those that have experienced environmental weathering, appear more likely to cause adverse effects than smooth, virgin particles (Choi et al., 2018; Jabeen et al., 2018). Smaller particles may also cause more adverse effects (Critchell and Hoogenboom, 2018; Ding et al., 2018; Mattsson et al., 2017), possibly because they can more easily translocate from the gut into the bloodstream and other tissues (Ding et al., 2018; Mattsson et al., 2017; Messinetti et al., 2019). The toxicity of microplastics can also increase and alter the toxicity of other environmental contaminants when organisms are co-exposed to microplastics and another contaminant, as has been shown for heavy metals (Barboza et al., 2018; Rainieri et al., 2018; Wen et al., 2018), persistent organic pollutants (Pannetier et al., 2019a, 2019b; Rainieri et al., 2018), and emerging contaminants (Chen et al., 2017; Shanshan Zhang et al., 2019). They can also act as a vector for pathogens (Viršek et al., 2017).

In 2019, the European Chemical Agency (ECHA), which oversees the use of chemicals in the European Union, released a recommendation for regulating microplastics in the environment. As background for this recommendation, ECHA conducted an extensive review of the scientific literature on occurrence, fate, and effects of microplastics, as well as discussions with stakeholders and scientific experts, and determined that risks arising from intentional uses of microplastics that result in releases to the environment are not currently adequately controlled (European Chemicals Agency, 2019a). Their conclusion was based on the lack of ecotoxicity data for calculation of risk thresholds, clear evidence of microplastic persistence, and uncertainty in regards to bioaccumulation potential. As a result, ECHA concluded that microplastics should be classified as a non-threshold substance for the purposes of risk assessment, with any release to the environment assumed to result in risk.

There is an urgent need for ecotoxicological studies that evaluate the effects of microplastics at environmentally relevant concentrations in organisms at multiple life stages. However, even with more ecotoxicological data, establishing risk thresholds will be challenging given the heterogeneous nature of this class of contaminants. Threshold values for a single contaminant in a given environmental media are normally set to protect the most sensitive species, but in the case of microplastics, the adverse impacts are potentially more wide-ranging and specific to the microplastic composition, size, and shape. As a result, ECHA and other agencies in the Bay Area, California, and worldwide have begun to recommend taking cost-effective microplastic pollution prevention actions now, despite uncertainties.

Microplastics as a concern in San Francisco Bay

Study setting: San Francisco Bay and adjacent National Marine Sanctuaries

San Francisco Bay is the largest estuary on the west coast of the Americas. The climate of wet winters and dry summers means that inputs of freshwater and associated chemical and biological constituents are highly seasonal. Seawater enters through the narrow deep channel at the Golden Gate, and freshwater is delivered primarily by the Sacramento and San Joaquin rivers. About 90% of the flow into the northern portion of the Bay comes from these rivers, which drain a combined watershed extending approximately 800 km from northwest to southeast, bounded by the Cascade and Sierra Nevada mountains. San Francisco Bay receives water that drains over 40% of the state, including a major inland agricultural area that includes several large cities such as Sacramento, Fresno, and Bakersfield. Southern California and the Central Valley agricultural production are also largely dependent upon water diverted from the freshwater inflows to the estuary. Active flushing of the northern and central portions of the Bay results in lower salinities and, in general, cleaner water quality than the southern portion of the Bay, which only receives about 10% of the freshwater flow.

The Bay is a unique ecosystem and an important habitat for a wide range of species. It provides spawning, nurturing, and hatching grounds, as well as a migration route stop-over for anadromous and marine species and birds on the Pacific Flyway. The estuary, rivers, and surrounding wetlands provide habitat for a large number of threatened and endangered species, including mammals like the Point Reyes mountain beaver (*Aplodontia rufa phaea*); birds like the California least tern (*Sternula antillarum brownii*) and the California clapper rail (*Rallus longirostris obsoletus*); and fish like steelhead trout (*Oncorhynchus mykiss*), delta smelt (*Hypomesus transpacificus*), and Chinook salmon (*Oncorhynchus tshawytscha*).

Outside the Golden Gate are three National Marine Sanctuaries: Greater Farallones, Cordell Bank, and Monterey Bay. Covering one of the most diverse and bountiful marine environments in the world, these protected areas are home to important underwater ecosystems, including eelgrass (*Zostera marina*), and an array of vulnerable or endangered species, including the black abalone (*Haliotis cracherodii*), tidewater goby (*Eucyclogobius newberryi*), short-tailed albatross (*Phoebastria albatrus*), and blue whales (*Balaenoptera musculus*).

The Bay Area is highly urbanized and home to over seven million people. As a result, the Bay assimilates approximately 1.2 billion cubic meters of treated wastewater per year, discharged from 42 wastewater treatment plants (San Francisco Regional Water Quality Control Board - Region 2, 2019). Stormwater flows to the Bay are highly variable but are estimated to be 1.7

Introduction

billion cubic meters per year from small tributaries surrounding the Bay (McKee et al., 2013); this may be compared to 25 billion cubic meters per year from the Sacramento and San Joaquin Rivers. The area also supports diverse industries. The Bay Area's four largest cities (San Francisco, San Jose, Oakland, and Fremont) collectively have approximately 3,200 manufacturing companies spanning such diverse sectors as electronics and semiconductors, metalworking, apparel, food and beverage, and lifestyle products (Bay Area State of Urban Manufacturing, 2016).

San Francisco Bay is a world leader in water quality monitoring, supporting a pioneering regional monitoring program that demonstrates how regional collaboration can provide the science needed to protect and improve water quality in a treasured ecosystem (Trowbridge et al., 2016). Because of its hydrodynamics, the Bay acts as a long-term trap for persistent contaminants, with recovery taking decades or longer when contamination is extensive (e.g., mercury and polychlorinated biphenyls).

Given all of these characteristics, the Bay is a prime example of an ecosystem that merits investigation of the scope of contamination and potential impacts of anthropogenic contaminants such as microplastics.

Previous study of the Bay

A preliminary screening study by the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) showed that, with an average abundance of 700,000 particles/km², Bay surface water appeared to have higher microplastic levels than other urban water bodies sampled in North America (Sutton et al., 2016). While this initial study also showed that treated wastewater from facilities that discharge into the Bay contained considerable quantities of microparticles, especially microfibers, higher levels of fragments in surface water suggested that additional pathways of pollution, such as urban stormwater runoff, may also be important contributors of microparticles and microplastics to the Bay. These findings indicated a need for further study of microplastics in the Bay.

In 2016, the RMP developed a strategy for continued study of microplastics in San Francisco Bay. To create this strategy, the RMP convened stakeholders to articulate management questions specific to microplastic pollution, and then conducted a one-day workshop that brought together local stakeholders and international technical experts to develop an understanding of the state of the science and determine consensus priorities for future work. The resulting Microplastic Monitoring and Science Strategy established a preliminary conceptual model focusing on sources and pathways of microplastic pollution in the Bay, and

Introduction

provided a multi-year plan to develop methods, monitor environmental concentrations, and evaluate control options (Sutton and Sedlak, 2017).

The Strategy was used to develop the objectives and study design for the three-year project described in this report. A refined conceptual model of microplastic sources and pathways is presented in Chapter 8 Synthesis.

The RMP evaluates emerging, unregulated contaminants, which currently includes microplastics, using a tiered risk-based framework intended to guide monitoring and management activities (Lin et al., 2018; Sutton et al., 2017). In response to the 2019 ECHA recommendation that microplastics should be classified as a non-threshold substance for the purposes of risk assessment, with any release to the environment assumed to result in risk, the RMP chose to elevate the status of microplastics in this framework to moderate concern for the Bay. This indicates a higher priority for microplastic studies focusing on status and trends, fate, effects, sources, pathways, and loadings, and suggests a need for the development of an action plan or management strategy, aggressive regional pollution prevention, and low-cost control or treatment actions.



The San Francisco Bay Microplastics Project

Study objectives and rationale

To develop critical baseline data and inform solutions, the San Francisco Estuary Institute and the 5 Gyres Institute embarked on a three-year collaborative project to complete the first comprehensive study of microplastic pollution of a major estuary and adjacent ocean. This project supported multiple scientific components to develop improved knowledge about microparticles and microplastics in San Francisco Bay and the Greater Farallones, Cordell Bank, and Monterey Bay National Marine Sanctuaries, with the following objectives.

- 1. Contribute to the development and standardization of sample collection and analysis methodology for microplastics research.*

There are currently no widely accepted standards for sample collection, laboratory analysis, quality assurance/quality control (QA/QC), or reporting of microplastics in environmental

Introduction

samples, making comparisons between studies and locations difficult. The lack of standardized methods has been widely acknowledged as a significant challenge for the field (Filella, 2015; Hermsen et al., 2018; Hidalgo-Ruz et al., 2012; Rochman et al., 2019; Shaoliang Zhang et al., 2019). Very few studies of microplastics in the environment use protocols that are standard to trace chemical analyses, including consistent collection of a range of QA/QC samples such as field and laboratory blanks, field duplicates, and model matrix spikes.

This project is unique in its documentation of the importance of rigorous techniques for field and laboratory procedures. In particular, quality assurance is necessary to evaluate the observations between studies and to distinguish between environmental variability, analytical data quality, and measurement drift. The diversity of microplastics also requires careful design of QA/QC methods in sampling and analytical measurement techniques, as different field and laboratory methods are better at recovering or measuring specific sizes, shapes, or types of microplastics.

Microplastics as a class pose special challenges for both sample collection and analysis because the ubiquitous use of plastics can lead to both widespread—and often widely varying or non-uniform—background contamination of samples, as commonly observed in both field and laboratory blanks or control samples (Hermsen et al., 2018; Koelmans et al., 2019; Lusher et al., 2017; Torre et al., 2016; Wesch et al., 2017; Woodall et al., 2015). At this time, there are no standard methods for uniformly addressing the influence of background contamination via correction of field sample values based on observations in accompanying field and/or laboratory blank samples (Brander et al., *in review*).

As a result, in this report we present blank sample data (microparticle counts) alongside field sample data, and have not corrected field sample data to account for potential background contamination. To provide guidance in interpreting the results, we developed **thresholds for data qualification** specific to each matrix and each particle morphology using standard techniques that are based on the average of the field and laboratory blanks and blank variability. Data are qualified as uncertain when they are below the average of field and laboratory blanks plus two times the blank standard deviation. Values below these thresholds may be strongly influenced by background contamination during sample collection and analysis. These thresholds are discussed in detail in the relevant chapters.

This conservative approach allows individual researchers to make their own assessments of the data. Our transparent and public datasets may be used for future calculations and refinements as our understanding of sources of background contamination improves and standard methods for blank correction are established within the scientific community.

2. *Determine a baseline for future monitoring of microplastics in San Francisco Bay surface water, sediment, and fish, and in ocean waters outside the Golden Gate.*

Introduction

Data are essential to understanding the impacts to and resilience of San Francisco Bay and its adjacent ocean areas. Monitoring trends and measuring the efficacy of management actions requires knowledge of microplastic baseline abundance, type, and composition in waters, sediment, and biota. This project aimed to establish the first comprehensive assessment of quantities and characteristics of microparticles and microplastics in the Bay environment.

3. *Characterize pathways by which microplastics enter the Bay, including urban stormwater and treated wastewater effluent.*

Evaluation of potential sources and pathways of microplastics may aid in identifying management actions. Determining relative microplastic abundance and characteristics in pathways to the Bay will allow design and prioritization of management actions. This project was unique in its methodologically consistent, side-by-side evaluation of two pathways, urban stormwater runoff and treated wastewater. The resulting data provided key information concerning the relative contribution of each pathway to different types of microplastics in the Bay.

Each of these pathways is influenced by different sources of microplastics, as described in Chapter 8 Synthesis. Source control (e.g., reduced use of plastic items identified as original sources of pollution) is typically found to be the most effective and least expensive pollution prevention option, and may be the primary tool applied to reduce microplastic pollution.

4. *Investigate the contribution of Bay microplastics to adjacent National Marine Sanctuaries through computer simulations.*

This project developed one of the first estuarine-marine microplastics transport models linking the Bay to the adjacent marine sanctuaries. Monitoring data were used to evaluate the model's predictive skill. The modeling results helped us understand the transport of microplastics through the environment and can be used to evaluate the potential effects of management actions.

5. *Communicate to regional stakeholders and the general public through meetings and educational materials.*

This project was designed to generate resources to inform and educate regional policy-makers, stakeholders, the wider scientific community, and the general public. These resources include this report, a summary sheet for the general public of how this science can inform microplastic policies, and films. In addition, this work is being translated into open access manuscripts and technical presentations. All data generated will be made publicly available through the California Environmental Data Exchange Network (CEDEN) in December 2019.

Introduction

6. *Facilitate evaluation of policy options for San Francisco Bay, with recommendations on source reduction.*

Understanding microplastics in the Bay from a scientific perspective is critical to informing and designing effective policy solutions, interventions, and innovations targeting individual, commercial, and industrial behavior, and waste and watershed management. Current policies that affect wastewater and stormwater treatment procedures and definitions of pollution are inadequate to address this growing and widespread threat.

The 5 Gyres Institute led an accompanying regional effort with input from multiple stakeholders to generate scientifically supported recommendations for solutions to plastic pollution in the region. These solutions and policy recommendations are presented in a companion document titled “San Francisco Bay Microplastics Project Science-Supported Solutions and Policy Recommendations” (Box and Cummins, 2019).

Looking forward

Microplastic pollution is a complex global issue and its management will require multi-stakeholder participation. Accurate measures of the sources, sinks, and reservoirs of microplastics in the environment are necessary to form an understanding of the magnitude of the problem, identify the highest priorities for mitigation, and inform effective management strategies.

This project was one of the first to conduct a comprehensive and integrated assessment using techniques adapted from the field of trace environmental chemistry. Microplastic concentrations and characteristics throughout the Bay and surrounding ocean were assessed in five different environmental matrices (stormwater, treated wastewater effluent, surface water, sediment, and prey fish), showcasing the power of large collaborative efforts in environmental sampling and analysis. This work resulted in seminal findings regarding microparticle source, pathway, and load characterization, as well as the development of novel methods for sampling and analysis of microplastics in different matrices, and development of the first model to link estuarine and ocean environments. This science spurred innovative discussions on solutions and science-driven recommendations for policies to mitigate microplastic pollution. The scientific information, tools, and recommended solutions developed via the San Francisco Bay Microplastics Project are intended to catalyze similar efforts to understand and reduce plastic pollution around the globe. Future monitoring of the Bay will continue to improve our understanding of microplastic pollution and track the effectiveness of management strategies. With sound science, we remain hopeful that as a society we can move forward to identify and implement successful solutions.

Glossary of key terms specific to this report

Microplastics — microparticles that have been definitively determined to be plastic

Microparticles — particles that appear to be plastic and are smaller than 5 mm in at least one external dimension

Microfibers — anthropogenic fibers smaller than 5 mm in diameter

Anthropogenic unknown — of anthropogenic origin (because it is dyed or has other chemical additive), but whether plastic or natural material could not be determined

Anthropogenic synthetic — interpretation of Raman or FTIR spectrum indicates the material is plastic, but does not indicate which polymer is present

Anthropogenic cellulosic — evidence indicates the material is human in origin, due to the presence of a color (i.e., not clear or white) or the Raman or FTIR spectrum of dyes or other synthetic compounds; the underlying material is cellulosic

Threshold for data qualification — data are qualified as uncertain when they are below the average of field and laboratory blanks plus two times the blank standard deviation specific to each matrix and each particle morphology

References

- Adam, V., Yang, T., Nowack, B., 2019. Toward an ecotoxicological risk assessment of microplastics: Comparison of available hazard and exposure data in freshwaters. *Environmental Toxicology and Chemistry* 38, 436–447. <https://doi.org/10.1002/etc.4323>
- Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Durández Jiménez, P., Simonneau, A., Binet, S., Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience* 12, 339–344. <https://doi.org/10.1038/s41561-019-0335-5>
- Azoulay, D., Villa, P., Arellano, Y., Gordon, M., Moon, D., Miller, K., Thompson, K., 2019. Plastic & Health: The Hidden Costs of a Plastic Planet. Center for International Environmental Law.
- Barboza, L.G.A., Vieira, L.R., Branco, V., Figueiredo, N., Carvalho, F., Carvalho, C., Guilhermino, L., 2018. Microplastics cause neurotoxicity, oxidative damage and energy-related changes and interact with the bioaccumulation of mercury in the European seabass, *Dicentrarchus labrax* (Linnaeus, 1758). *Aquatic Toxicology* 195, 49–57. <https://doi.org/10.1016/j.aquatox.2017.12.008>
- Barrows, A.P.W., Cathey, S.E., Petersen, C.W., 2018. Marine environment microfiber contamination: Global patterns and the diversity of microparticle origins. *Environmental Pollution* 237, 275–284. <https://doi.org/10.1016/j.envpol.2018.02.062>
- Barrows, A.P.W., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab vs. neuston tow net: A microplastic sampling performance comparison and possible advances in the field. *Analytical Methods* 9, 1446–1453. <https://doi.org/10.1039/C6AY02387H>
- Bay Area State of Urban Manufacturing, 2016. SFMade. BayAreaMfg.org
- Bergmann, M., Mütsel, S., Primpke, S., Tekman, M.B., Trachsel, J., Gerdts, G., 2019. White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. *Science Advances* 5, eaax1157. <https://doi.org/10.1126/sciadv.aax1157>
- Bergmann, M., Wirzberger, V., Krumpen, T., Lorenz, C., Primpke, S., Tekman, M.B., Gerdts, G., 2017. High quantities of microplastic in Arctic deep-sea sediments from the HAUSGARTEN Observatory. *Environmental Science & Technology* 51, 11000–11010. <https://doi.org/10.1021/acs.est.7b03331>

Introduction

- Besseling, E., Redondo-Hasselerharm, P., Foekema, E.M., Koelmans, A.A., 2019. Quantifying ecological risks of aquatic micro- and nanoplastic. Critical Reviews in Environmental Science and Technology 49, 32–80. <https://doi.org/10.1080/10643389.2018.1531688>
- Boucher, J., Friot, D., 2017. Primary Microplastics in the Oceans: A Global Evaluation of Sources. Gland, Switzerland: International Union for Conservation of Nature (IUCN), 43pp.
- Box, C., and Cummins, A. 2019. San Francisco Bay Microplastics Project Science-Supported Solutions and Policy Recommendations. Prepared by The 5 Gyres Institute.
- Brander, S., Renick, V., Foley, M., Steele, C., Woo, M., Lusher, A., Carr, S.A., Helm, P.A., Box, C., Cherniak, S., Andrews, R., Rochman, C.M., in review. Sampling and QA/QC, or how many blanks do I need? A guide for scientists investigating the occurrence of microplastics across matrices.
- Burns, E.E., Boxall, A.B.A., 2018. Microplastics in the aquatic environment: Evidence for or against adverse impacts and major knowledge gaps. Environmental Toxicology and Chemistry 37, 2776–2796. <https://doi.org/10.1002/etc.4268>
- California Department of Resources Recycling and Recovery (CalRecycle), 2019. Single-Use Carryout Bag Ban (SB 270) [WWW Document]. Plastics Programs and Resources. URL <https://www.calrecycle.ca.gov/plastics/carryoutbags> (accessed 9.9.19).
- California State Water Resources Control Board, 2017. Statewide Water Quality Control Plans for Trash [WWW Document]. URL https://www.waterboards.ca.gov/water_issues/programs/trash_control/ (accessed 9.9.19).
- Chagnon, C., Thiel, M., Antunes, J., Ferreira, J.L., Sobral, P., Ory, N.C., 2018. Plastic ingestion and trophic transfer between Easter Island flying fish (*Cheilopogon rapanouensis*) and yellowfin tuna (*Thunnus albacares*) from Rapa Nui (Easter Island). Environmental Pollution 243, 127–133. <https://doi.org/10.1016/j.envpol.2018.08.042>
- Chen, Q., Yin, D., Jia, Y., Schiwy, S., Legradi, J., Yang, S., Hollert, H., 2017. Enhanced uptake of BPA in the presence of nanoplastics can lead to neurotoxic effects in adult zebrafish. Science of the Total Environment 609, 1312–1321. <https://doi.org/10.1016/j.scitotenv.2017.07.144>
- Choi, J.S., Jung, Y.-J., Hong, N.-H., Hong, S.H., Park, J.-W., 2018. Toxicological effects of irregularly shaped and spherical microplastics in a marine teleost, the sheepshead minnow (*Cyprinodon variegatus*). Marine Pollution Bulletin 129, 231–240. <https://doi.org/10.1016/j.marpolbul.2018.02.039>

Introduction

Cole, M., Lindeque, P., Halsband, C., Galloway, T.S., 2011. Microplastics as contaminants in the marine environment: A review. *Marine Pollution Bulletin* 62, 2588–2597.
<https://doi.org/10.1016/j.marpolbul.2011.09.025>

Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. *Science of the Total Environment* 667, 124–132. <https://doi.org/10.1016/j.scitotenv.2019.02.346>

Critchell, K., Hoogenboom, M.O., 2018. Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (*Acanthochromis polyacanthus*). *PLoS ONE* 13, e0193308. <https://doi.org/10.1371/journal.pone.0193308>

de Sá, L.C., Oliveira, M., Ribeiro, F., Rocha, T.L., Futter, M.N., 2018. Studies of the effects of microplastics on aquatic organisms: What do we know and where should we focus our efforts in the future? *Science of the Total Environment* 645, 1029–1039.
<https://doi.org/10.1016/j.scitotenv.2018.07.207>

Ding, J., Zhang, S., Razanajatovo, R.M., Zou, H., Zhu, W., 2018. Accumulation, tissue distribution, and biochemical effects of polystyrene microplastics in the freshwater fish red tilapia (*Oreochromis niloticus*). *Environmental Pollution* 238, 1–9.
<https://doi.org/10.1016/j.envpol.2018.03.001>

Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin* 104, 290–293.
<https://doi.org/10.1016/j.marpolbul.2016.01.006>

Duis, K., Coors, A., 2016. Microplastics in the aquatic and terrestrial environment: Sources (with a specific focus on personal care products), fate and effects. *Environmental Sciences Europe* 28. <https://doi.org/10.1186/s12302-015-0069-y>

European Chemicals Agency, 2019a. Annex XV Restriction Report – Microplastics (Version 1.1). European Chemicals Agency (ECHA).

European Chemicals Agency, 2019b. Annex to the Annex XV Restriction Report – Microplastics. European Chemicals Agency (ECHA).

Everaert, G., Van Cauwenberghe, L., De Rijcke, M., Koelmans, A.A., Mees, J., Vandegehuchte, M., Janssen, C.R., 2018. Risk assessment of microplastics in the ocean: Modelling approach and first conclusions. *Environmental Pollution* 242, 1930–1938.
<https://doi.org/10.1016/j.envpol.2018.07.069>

Introduction

- Farady, S.E., 2019. Microplastics as a new, ubiquitous pollutant: Strategies to anticipate management and advise seafood consumers. *Marine Policy* 104, 103–107.
<https://doi.org/10.1016/j.marpol.2019.02.020>
- Farrell, P., Nelson, K., 2013. Trophic level transfer of microplastic: *Mytilus edulis* (L.) to *Carcinus maenas* (L.). *Environmental Pollution* 177, 1–3. <https://doi.org/10.1016/j.envpol.2013.01.046>
- Filella, M., 2015. Questions of size and numbers in environmental research on microplastics: Methodological and conceptual aspects. *Environmental Chemistry* 12, 527.
<https://doi.org/10.1071/EN15012>
- Foley, C.J., Feiner, Z.S., Malinich, T.D., Höök, T.O., 2018. A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. *Science of the Total Environment* 631–632, 550–559. <https://doi.org/10.1016/j.scitotenv.2018.03.046>
- Free, C.M., Jensen, O.P., Mason, S.A., Eriksen, M., Williamson, N.J., Boldgiv, B., 2014. High-levels of microplastic pollution in a large, remote, mountain lake. *Marine Pollution Bulletin* 85, 156–163. <https://doi.org/10.1016/j.marpolbul.2014.06.001>
- Fries, E., H. Dekiff, J., Willmeyer, J., Nuelle, M.-T., Ebert, M., Remy, D., 2013. Identification of polymer types and additives in marine microplastic particles using pyrolysis-GC/MS and scanning electron microscopy. *Environmental Science: Processes & Impacts* 15, 1949–1956.
<https://doi.org/10.1039/C3EM00214D>
- Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. *Scientific Advances* 3, e1700782. <https://doi.org/10.1126/sciadv.1700782>
- Group of Chief Scientific Advisors, 2019. Scientific Opinion: Environmental and Health Risks of Microplastic Pollution (Scientific Opinion 6/2019).
- Hermsen, E., Mintenig, S.M., Besseling, E., Koelmans, A.A., 2018. Quality criteria for the analysis of microplastic in biota samples: A critical review. *Environmental Science & Technology* 52, 10230–10240. <https://doi.org/10.1021/acs.est.8b01611>
- Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: A review of the methods used for identification and quantification. *Environmental Science & Technology* 46, 3060–3075. <https://doi.org/10.1021/es2031505>
- Jabeen, K., Li, B., Chen, Q., Su, L., Wu, C., Hollert, H., Shi, H., 2018. Effects of virgin microplastics on goldfish (*Carassius auratus*). *Chemosphere* 213, 323–332.
<https://doi.org/10.1016/j.chemosphere.2018.09.031>

Introduction

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. *Science* 347, 768–771.
<https://doi.org/10.1126/science.1260352>

Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, 2015. Sources, Fate and Effects of Microplastics in the Marine Environment: A Global Assessment.

Kang, J.-H., Kwon, O.Y., Lee, K.-W., Song, Y.K., Shim, W.J., 2015. Marine neustonic microplastics around the southeastern coast of Korea. *Marine Pollution Bulletin* 96, 304–312.
<https://doi.org/10.1016/j.marpolbul.2015.04.054>

Koelmans, A.A., Mohamed Nor, N.H., Hermsen, E., Kooi, M., Mintenig, S.M., De France, J., 2019. Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Research* 155, 410–422. <https://doi.org/10.1016/j.watres.2019.02.054>

Law, K.L., 2017. Plastics in the marine environment. *Annual Review of Marine Science* 9, 205–229. <https://doi.org/10.1146/annurev-marine-010816-060409>

Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., Marthouse, R., Hajbane, S., Cunsolo, S., Schwarz, A., Levivier, A., Noble, K., Debeljak, P., Maral, H., Schoeneich-Argent, R., Brambini, R., Reisser, J., 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. *Scientific Reports* 8, 4666. <https://doi.org/10.1038/s41598-018-22939-w>

Lin, D., Sutton, R., Shimabuku, I., Sedlak, M., Wu, J., Holleman, R., 2018. Contaminants of Emerging Concern in San Francisco Bay: A Strategy for Future Investigations 2018 Update. SFEI Contribution No. 873. San Francisco Estuary Institute, Richmond, CA.

Lusher, A.L., Tirelli, V., O'Connor, I., Officer, R., 2015. Microplastics in Arctic polar waters: The first reported values of particles in surface and sub-surface samples. *Scientific Reports* 5. <https://doi.org/10.1038/srep14947>

Lusher, A.L., Welden, N.A., Sobral, P., Cole, M., 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Analytical Methods* 9, 1346–1360. <https://doi.org/10.1039/C6AY02415G>

Masura, B., Baker, J., Foster, G., Arthur, C., 2015. Laboratory Methods for the Analysis of Microplastics in the Marine Environment (Technical Memorandum NOS-OR & R-48), NOAA Marine Debris Program. National Oceanic and Atmospheric Administration.

Introduction

- Mattsson, K., Johnson, E.V., Malmendal, A., Linse, S., Hansson, L.-A., Cedervall, T., 2017. Brain damage and behavioural disorders in fish induced by plastic nanoparticles delivered through the food chain. *Scientific Reports* 7, 11452. <https://doi.org/10.1038/s41598-017-10813-0>
- McCormick, A., Hoellein, T.J., Mason, S.A., Schluep, J., Kelly, J.J., 2014. Microplastic is an abundant and distinct microbial habitat in an urban river. *Environmental Science & Technology* 48, 11863–11871. <https://doi.org/10.1021/es503610r>
- McKee, L.J., Lewicki, M., Schoellhamer, D.H., Ganju, N.K., 2013. Comparison of sediment supply to San Francisco Bay from watersheds draining the Bay Area and the Central Valley of California. *Marine Geology* 345, 47–62. <https://doi.org/10.1016/j.margeo.2013.03.003>
- Messinetti, S., Mercurio, S., Scarì, G., Pennati, A., Pennati, R., 2019. Ingested microscopic plastics translocate from the gut cavity of juveniles of the ascidian *Ciona intestinalis*. *The European Zoological Journal* 86, 189–195. <https://doi.org/10.1080/24750263.2019.1616837>
- Munno, K., De Frond, H., O'Donnell, B., Rochman, C., in review. Increasing the accessibility for characterizing microplastics: Introducing new application-based and spectral libraries of plastic particles (SLoPP & SLoPP-E).
- Pannetier, P., Cachot, J., Clérando, C., Faure, F., Van Arkel, K., de Alencastro, L.F., Levasseur, C., Sciacca, F., Bourgeois, J.-P., Morin, B., 2019a. Toxicity assessment of pollutants sorbed on environmental sample microplastics collected on beaches: Part I-adverse effects on fish cell line. *Environmental Pollution* 248, 1088–1097. <https://doi.org/10.1016/j.envpol.2018.12.091>
- Pannetier, P., Morin, B., Clérando, C., Laurent, J., Chapelle, C., Cachot, J., 2019b. Toxicity assessment of pollutants sorbed on environmental microplastics collected on beaches: Part II-adverse effects on Japanese medaka early life stages. *Environmental Pollution* 248, 1098–1107. <https://doi.org/10.1016/j.envpol.2018.10.129>
- Plastic Europe, 2018. Plastics: The Facts An analysis of European plastics production, demand and waste data. www.plasticseurope.org
- Prokić, M.D., Radovanović, T.B., Gavrić, J.P., Faggio, C., 2019. Ecotoxicological effects of microplastics: Examination of biomarkers, current state and future perspectives. *TrAC Trends in Analytical Chemistry* 111, 37–46. <https://doi.org/10.1016/j.trac.2018.12.001>
- Rainieri, S., Conllledo, N., Larsen, B.K., Granby, K., Barranco, A., 2018. Combined effects of microplastics and chemical contaminants on the organ toxicity of zebrafish (*Danio rerio*). *Environmental Research* 162, 135–143. <https://doi.org/10.1016/j.envres.2017.12.019>

Introduction

Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K., Athey, S., Huntington, A., McIlwraith, H., Munno, K., Frond, H.D., Kolomijeca, A., Erdle, L., Grbic, J., Bayoumi, M., Borrelle, S.B., Wu, T., Santoro, S., Werbowski, L.M., Zhu, X., Giles, R.K., Hamilton, B.M., Thaysen, C., Kaura, A., Klasios, N., Ead, L., Kim, J., Sherlock, C., Ho, A., Hung, C., 2019. Rethinking microplastics as a diverse contaminant suite. *Environmental Toxicology and Chemistry* 38, 703–711. <https://doi.org/10.1002/etc.4371>

San Francisco Regional Water Quality Control Board - Region 2, 2019. San Francisco Basin Plan (downloaded Jan 23 2019 from website). www.waterboards.ca.gov

Science Advice for Policy by European Academies, 2019. A Scientific Perspective on Microplastics in Nature and Society (Version 2019.1.1).

Setälä, O., Fleming-Lehtinen, V., Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution* 185, 77–83.
<https://doi.org/10.1016/j.envpol.2013.10.013>

Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. *Marine Pollution Bulletin* 109, 230–235. <https://doi.org/10.1016/j.marpolbul.2016.05.077>

Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay. SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA.

Sutton, R., Sedlak, M., Sun, J., Lin, D., 2017. Contaminants of Emerging Concern in San Francisco Bay: A Strategy for Future Investigations 2017 Revision. SFEI Contribution No. 815. San Francisco Estuary Institute, Richmond, CA.

Tanaka, K., Takada, H., 2016. Microplastic fragments and microbeads in digestive tracts of planktivorous fish from urban coastal waters. *Scientific Reports* 6, 34351.
<https://doi.org/10.1038/srep34351>

Thompson, R.C., Moore, C.J., vom Saal, F.S., Swan, S.H., 2009. Plastics, the environment and human health: Current consensus and future trends. *Philosophical Transactions of the Royal Society B* 364, 2153–2166. <https://doi.org/10.1098/rstb.2009.0053>

Torre, M., Digka, N., Anastasopoulou, A., Tsangaridis, C., Mytilineou, C., 2016. Anthropogenic microfibres pollution in marine biota. A new and simple methodology to minimize airborne contamination. *Marine Pollution Bulletin* 113, 55–61.
<https://doi.org/10.1016/j.marpolbul.2016.07.050>

Introduction

Tosetto, L., Williamson, J.E., Brown, C., 2017. Trophic transfer of microplastics does not affect fish personality. *Animal Behaviour* 123, 159–167.
<https://doi.org/10.1016/j.anbehav.2016.10.035>

Trowbridge, P.R., Davis, J.A., Mumley, T., Taberski, K., Feger, N., Valiela, I., Ervin, J., Arsem, N., Olivieri, A., Carroll, P., Coleman, J., Salop, P., Sutton, R., Yee, D., McKee, L.J., Sedlak, M., Grosso, C., Kelly, J., 2016. The Regional Monitoring Program for Water Quality in San Francisco Bay, California, USA: Science in support of managing water quality. *Regional Studies in Marine Science, US Monitoring Programs* 4, 21–33. <https://doi.org/10.1016/j.rsma.2015.10.002>

United Nations Environment Programme, 2019. Global Chemicals Outlook II From Legacies to Innovative Solutions: Implementing the 2030 Agenda for Sustainable Development – Synthesis Report.

Van Cauwenberghe, L., Vanreusel, A., Mees, J., Janssen, C.R., 2013. Microplastic pollution in deep-sea sediments. *Environmental Pollution* 182, 495–499.
<https://doi.org/10.1016/j.envpol.2013.08.013>

van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B.D., van Franeker, J.A., Eriksen, M., Siegel, D., Galgani, F., Law, K.L., 2015. A global inventory of small floating plastic debris. *Environmental Research Letters* 10, 124006. <https://doi.org/10.1088/1748-9326/10/12/124006>

Viršek, M.K., Lovšin, M.N., Koren, Š., Kržan, A., Peterlin, M., 2017. Microplastics as a vector for the transport of the bacterial fish pathogen species *Aeromonas salmonicida*. *Marine Pollution Bulletin* 125, 301–309. <https://doi.org/10.1016/j.marpolbul.2017.08.024>

Wen, B., Jin, S.-R., Chen, Z.-Z., Gao, J.-Z., Liu, Y.-N., Liu, J.-H., Feng, X.-S., 2018. Single and combined effects of microplastics and cadmium on the cadmium accumulation, antioxidant defense and innate immunity of the discus fish (*Sympodus aequifasciatus*). *Environmental Pollution* 243, 462–471. <https://doi.org/10.1016/j.envpol.2018.09.029>

Wesch, C., Elert, A.M., Wörner, M., Braun, U., Klein, R., Paulus, M., 2017. Assuring quality in microplastic monitoring: About the value of clean-air devices as essentials for verified data. *Scientific Reports* 7. <https://doi.org/10.1038/s41598-017-05838-4>

Wessel, C.C., Lockridge, G.R., Battiste, D., Cebrian, J., 2016. Abundance and characteristics of microplastics in beach sediments: Insights into microplastic accumulation in northern Gulf of Mexico estuaries. *Marine Pollution Bulletin* 109, 178–183.
<https://doi.org/10.1016/j.marpolbul.2016.06.002>

Introduction

- Woodall, L.C., Gwinnett, C., Packer, M., Thompson, R.C., Robinson, L.F., Paterson, G.L.J., 2015. Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments. *Marine Pollution Bulletin* 95, 40–46.
<https://doi.org/10.1016/j.marpolbul.2015.04.044>
- Worm, B., Lotze, H.K., Jubinville, I., Wilcox, C., Jambeck, J., 2017. Plastic as a persistent marine pollutant. *Annual Review of Environment and Resources* 42, 1–26.
<https://doi.org/10.1146/annurev-environ-102016-060700>
- Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution* 178, 483–492.
<https://doi.org/10.1016/j.envpol.2013.02.031>
- Zhang, Shanshan, Ding, J., Razanajatovo, R.M., Jiang, H., Zou, H., Zhu, W., 2019. Interactive effects of polystyrene microplastics and roxithromycin on bioaccumulation and biochemical status in the freshwater fish red tilapia (*Oreochromis niloticus*). *Science of the Total Environment* 648, 1431–1439. <https://doi.org/10.1016/j.scitotenv.2018.08.266>
- Zhang, Shaoliang, Wang, J., Liu, X., Qu, F., Wang, Xueshan, Wang, Xinrui, Li, Y., Sun, Y., 2019. Microplastics in the environment: A review of analytical methods, distribution, and biological effects. *TrAC Trends in Analytical Chemistry* 111, 62–72.
<https://doi.org/10.1016/j.trac.2018.12.002>

CHAPTER

2

Microparticles and Microplastics
IN BAY AREA STORMWATER : by Alicia Gilbreath



Highlights

- ◆ This study measured microparticles and microplastics in stormwater from 12 small tributaries comprising 11% of the watershed drainage area to San Francisco Bay (6% of total flow to the Bay; excludes input from the large Sacramento-San Joaquin River watershed). These tributaries varied in urban and non-urban land uses and were distributed across the region.
- ◆ No methods existed for volumetric quantification of microparticles suitable for estimating loads in stormwater. We developed methods for this study and examined their performance using standard quality assurance measures. Microparticles captured on sequential 355 and 125 micron sieves were manually counted using visual techniques; 859 (approximately 7%) of the microparticles were then analyzed using spectroscopy to determine whether they were microplastics.
- ◆ Microparticles were identified in stormwater from all 12 small tributaries. Concentrations ranged from 1.3 to 30 microparticles per liter. The microparticle concentrations observed are consistent with those observed in some studies and higher than others previously reported in the literature. However, direct comparison is challenging due to different collection techniques, mesh or sieve sizes, and analytical procedures.
- ◆ Fragments (59%) and fibers (39%) constituted nearly all microparticles sampled.
- ◆ Black fragments that had a rubbery texture were abundant in stormwater and constituted 48% of all microparticles in samples. It was not possible to identify the composition of these particles using Raman or Fourier Transform Infrared (FTIR) spectroscopy; however, analysts reported that based on secondary characteristics, these particles were similar to particles previously identified as rubber by FTIR spectroscopy.
- ◆ An estimated load of 7.2 trillion microplastics are conveyed from small tributaries to the Bay each year based on watersheds modeling conducted using the previously developed Regional Watershed Spreadsheet Model (RWSM) for the San Francisco Bay Area. The stormwater microplastics load estimate is approximately 300 times greater than the estimated microplastics load from wastewater.
- ◆ The results of the watersheds modeling effort suggested that industrial areas may be linked to higher microparticle concentrations in stormwater. We recommend additional investigation into sources of microplastics in the landscape, including a greater number of relevant landscape attributes (e.g., imperviousness, proximity to roadways), to more fully explore factors that are potentially related to higher levels of microparticles in stormwater.

Objectives

Stormwater runoff is believed to be one of the primary pathways for plastic pollution to enter the Bay (BASMAA, 2014a; Boucher and Friot, 2017; EOA, 2014; GESAMP, 2016; Sutton and Sedlak, 2017). Primary microplastics from industry and other activities (e.g., plastic nurdles), as well as secondary sources of larger plastics fragmented by photooxidative degradation or physical abrasion (e.g., tire abrasion due to roadway wear), can be entrained in stormwater runoff from the landscape and enter drainage systems that discharge to local creeks, rivers, and the Bay.

This is the first study to evaluate microplastics in stormwater entering San Francisco Bay from multiple watersheds, a data gap noted in the prior Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) microplastics screening study (Sutton et al., 2016). Trash monitoring studies have been conducted in local storm drains, and have demonstrated the ubiquity of larger plastic items within urban litter (e.g., BASMAA, 2014b; BASMAA, 2016; EOA, 2014); it is a logical extension of this work that urban stormwater runoff likely plays a major role in mobilizing both macro- and microplastics from the landscape to the Bay.

Through assessing microparticle and microplastic abundances and characteristics of stormwater collected from tributaries to the San Francisco Bay, this study sought to address the following objectives.

1. **Quantify the abundance of microparticles and microplastics in stormwater.**
Understanding the abundance of microparticles and microplastics in stormwater is important for evaluating the sources, pathways, loadings, and processes [Management Question (MQ) 3] leading to microplastics in the Bay [(MQ1); see the Microplastic Monitoring and Science Strategy for San Francisco Bay (Sutton and Sedlak, 2017) for further detail on the MQs].
2. **Characterize types of microparticles and microplastics found in stormwater and their chemical composition.** Understanding the types of microparticles and microplastics found in stormwater will help determine the sources of microplastics in stormwater. This could help inform future decisions about management measures (MQ5) that could contribute to possible future reductions of microplastics transported to the Bay via stormwater.
3. **Assess microparticle and microplastic concentrations in relation to watershed attributes.** Are concentrations from urban sites higher than rural, open, and undeveloped spaces? Are certain land uses associated with higher concentrations of microparticles and microplastics? Evaluating these questions allows development of

improved conceptual models that can point to management actions that may be effective in reducing microplastic pollution (MQ5).

4. **Calculate estimates of microparticle and microplastic loads via stormwater into the Bay.** Developing first order estimates of microparticle and microplastic loads from the various transport pathways provides information relevant to MQ3, as well as aids in prioritizing management actions (MQ5).
5. **Develop and test new methods for collecting stormwater samples.** A key step in quantifying the abundance of microparticles and microplastics (MQ1) is establishing appropriate field and laboratory methods for measurement. A review of the literature revealed no existing methods for volumetric quantification of microparticles suitable for estimating loads in stormwater. We developed methods for this study and examined method performance using standard quality assurance measures.

This study was designed to test the following hypotheses (Sedlak et al., 2017).

- ◆ Microplastics will be present in stormwater.
- ◆ Rural-dominated watersheds will have lower concentrations of microparticles and microplastics than urban-dominated watersheds.
- ◆ Concentrations of microplastics in stormwater and wastewater will be comparable; however, the composition will be different. (See Chapter 8 Synthesis for further exploration of this hypothesis.)

Methods

In this report, we have distinguished between microparticles, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and microplastics, which have been confirmed to be plastic through Raman or FTIR spectroscopy. The upper size boundary for microparticles and microplastics is typically defined as 5 mm, while the lower size boundary is operationally defined by the interaction between the mesh pore size and the characteristics (e.g., shape or other physical properties) of each particle that prevent it from passing through the sieve.

Stormwater site selection

Stormwater samples were collected at 12 sites distributed around San Francisco Bay (Figure 2.1; Table 2.1). Sites were selected based on drainage area (5–323 km²), land use, and geographical distribution around the Bay. Sites that overlapped with Bay fish and sediment sampling (e.g., San Leandro Bay sites near the Coliseum and the Lower South Bay Guadalupe site) were a high priority for sampling.

Altogether, the watersheds sampled comprise 11% of the total small tributaries¹ area draining into the Bay (i.e., 763 km² of 6,725 km²; excludes areas upstream of major reservoirs) and 6% of the total flow to the Bay via the small tributaries. Land-use characteristics varied across the watersheds, with total urban area within the watersheds ranging widely from 9%–98%² (Figure 2.2). Below their reservoirs, Lower Coyote Creek and Guadalupe River (two of the 12 watersheds sampled) are the 4th and 8th largest tributaries to the Bay, respectively. These two watersheds are typical of larger Bay Area watersheds with rural areas in the upper watershed and more urban areas closer to the Bay. They also both have homeless encampments at some locations along the banks; homeless encampments have been associated with trash hot spots (City of San Jose, 2016; CRWQCB SF Bay, 2015) and can be large contributors of litter in local waterways. Taken together, these characteristics facilitated a broad survey of microplastics in small tributaries around the Bay.

¹ Small tributaries are all of the stormwater conveyances to San Francisco Bay, excluding input from the large Sacramento-San Joaquin River watershed.

² The land use dataset is now over ten years old and may not accurately represent current land use given levels of redevelopment.

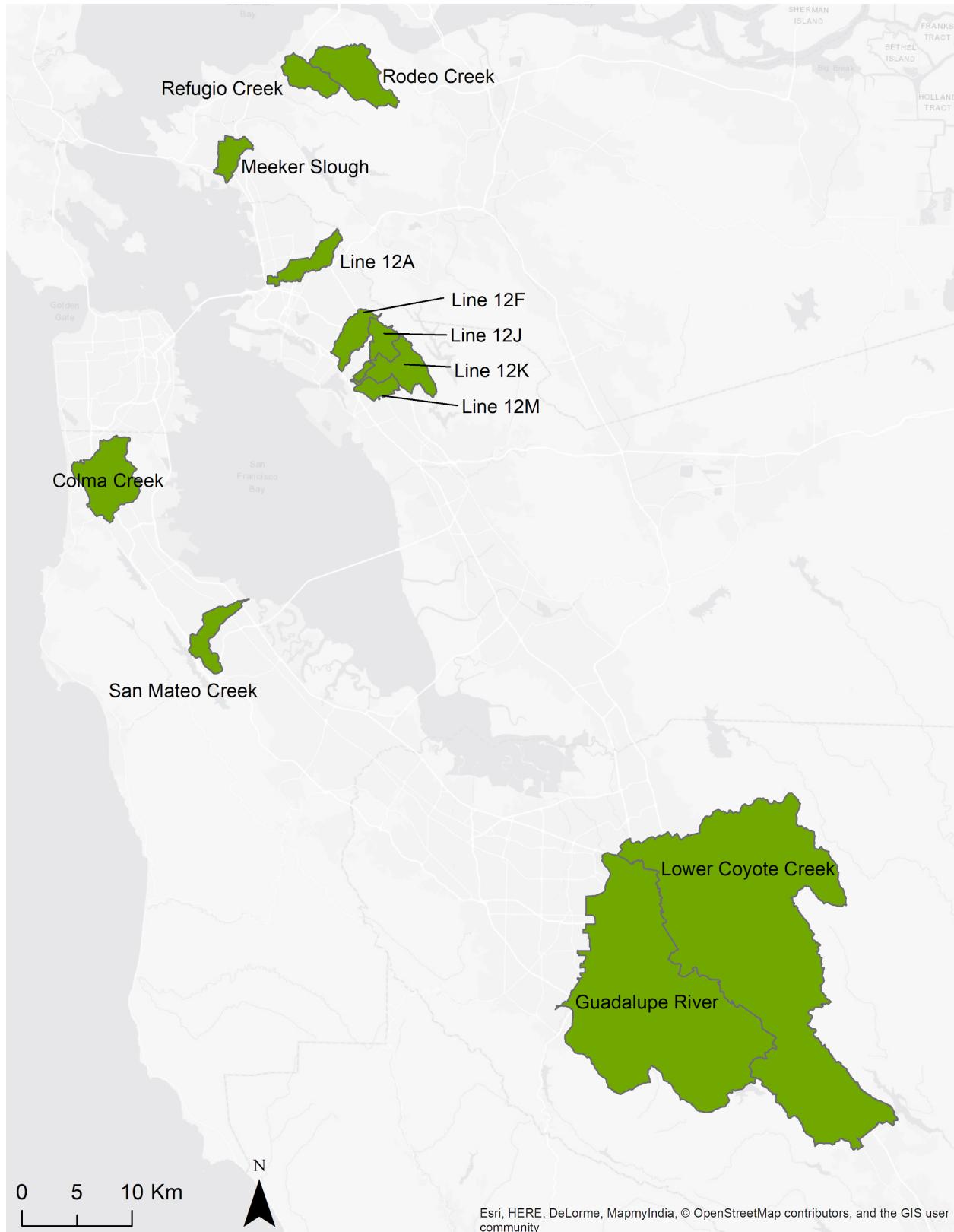


Figure 2.1. Watersheds sampled for microparticles and microplastics in stormwater.

Table 2.1. Attributes of stormwater sampling locations.

RMP Site Name	Location	Sampling Date	Latitude	Longitude	Size of Watershed (km²)	Rationale for Site Selection	# Liters Sieved
Line 12A at Shellmound St Pedestrian Bridge	Central Bay	1/8/18	37.834	-122.293	10.5	Urban (Commercial / Residential)	115
Line 12F below PG&E station	Central Bay	12/15/16	37.762	-122.214	10.2	Urban (Commercial / Residential)	25
Line 12J at mouth to 12K	Central Bay	12/15/16	37.755	-122.201	8.81	Near bay and includes commercial, residential and industrial	67
Line 12K at Coliseum Entrance	Central Bay	2/9/17	37.754	-122.204	16.4	Near bay and includes commercial, residential and industrial	295
Line 12M at Coliseum Way	Central Bay	11/28/18	37.747	-122.201	5.3	Near bay and includes commercial, residential and industrial	68
Meeker Slough at Regatta Blvd	Central Bay	1/8/18	37.918	-122.338	7.34	Mixed residential, Drains into inner harbor in Richmond	67

RMP Site Name	Location	Sampling Date	Latitude	Longitude	Size of Watershed (km²)	Rationale for Site Selection	# Liters Sieved
Colma Ck at Linden	South Bay	2/7/17	37.65	-122.412	27.5	303d listed for trash, Part of Tracking CA Trash Project, Major Tributary	197
San Mateo Creek	South Bay	1/8/18	37.573	-122.311	11.4 (below reservoir)	303(d) listed for trash, Part of Tracking Trash Project, major tributary	114
Guadalupe River	Lower SB	1/8/17	37.374	-121.933	233 (below reservoirs, 8th largest tributary to the Bay)	Near Highway 101	138
Coyote Creek	South Bay	4/6/18	37.385832	-121.910	323 (below reservoirs, 4th largest tributary to the Bay)	Major stormwater and wastewater influenced tributary to Lower South Bay, Part of Tracking CA Trash Project	114
Refugio Creek at Tsushima St	San Pablo Bay	1/18/17	38.018	-122.277	10.7	Open space	54
Rodeo Creek at Seacliff Ct Pedestrian Br	San Pablo Bay	1/18/17	38.016	-122.254	23.4	Open space	63

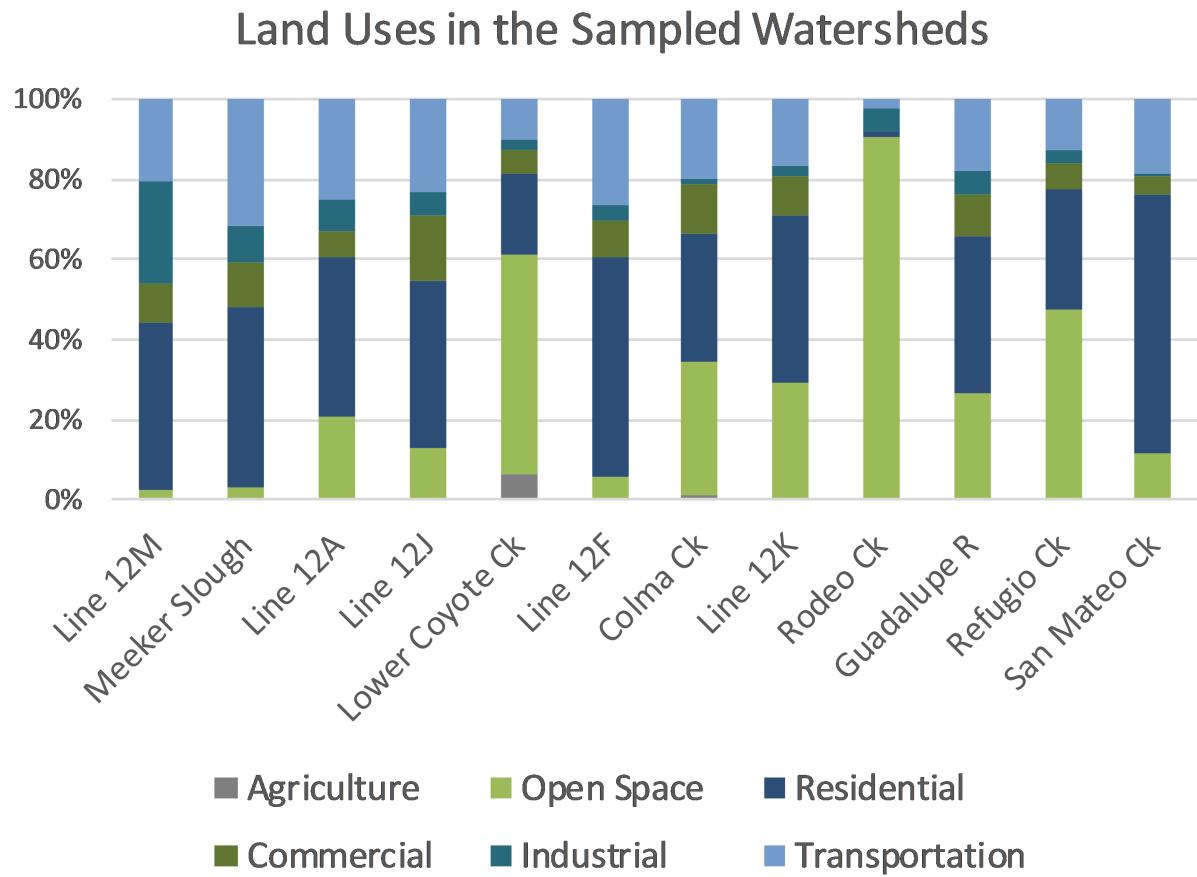


Figure 2.2. Land use distributions in the 12 sampled watersheds (excluding area upstream of reservoirs; land use source data: ABAG 2006).

Sample collection

Each site was sampled once via an aggregated sample collected throughout a storm runoff event. Because 95% of the flow in Bay Area small tributaries is the direct result of rainfall (McKee et al., 2003), we focused our sampling during rainfall events. Samples were collected between December 2016 and November 2018 during storms when more than 1.3 cm (0.50 inches) of rain was expected within six hours (BASMAA, 2016; Sedlak et al., 2017). Based on prior studies of legacy contaminants in Bay Area watersheds, this threshold typically results in storms that are sufficiently intense to mobilize small particles from the watershed (Gilbreath and McKee, 2015); storms forecasted for shorter duration and smaller magnitude often result in storms that lead to little runoff. Attempts were made to get to each sampling site and begin sampling at the start of storm-related flow. Collection typically occurred on the rising, peak, and falling portions of the hydrograph.

Due in part to limited research worldwide on microplastics in stormwater, no standardized methods to collect microparticles or microplastics in stormwater exist. At most sites in this study, samples were collected using a Teledyne ISCO portable pump sampler, with tubing from the pump attached to a 4–8 m fiberglass pole. The sampling pole was moved vertically through the water column to provide a depth-integrated sample; care was taken to avoid contacting the stream bed. A depth-integrated concentration is essential for estimating loads. At the Guadalupe River site, the drop from the height of the bridge at the sampling site to the river was too great to use an ISCO pump; instead, a water sample was collected using a stainless steel 3-gallon pail that filled almost instantly upon contact with the surface water.



The field team passed a total of 25–295 L (Table 2.1) of stormwater through stacked 355 µm and 125 µm sieves, by collecting 10–60 L “sips” multiple times during the rising and falling stages of a storm hydrograph. The number of sips was a function of the duration of the storm and varied among sites. In between sips, the field sample sieve set was covered in foil and placed in a dedicated closed box to reduce the potential for background contamination.

Once collection was complete, the sieve sets were transported to and processed in SFEI’s laboratory. Microparticles were gently rinsed off the sieves into pre-cleaned glass sample jars using distilled water. Approximately 10 mL of isopropyl alcohol was added to each sample for storage. Samples were shipped to the University of Toronto for analysis.

One field blank and one field duplicate were collected with the field samples and subjected to similar processing and analyses. A field blank was collected at one site by placing a set of sieves near the field sample for the duration of the sampling period. When the foil lid was taken off the field sample, the foil lid was also taken off the field blank to maintain the same amount of air exposure. A field duplicate was collected at Line 12M by setting up a second set of sieves

adjacent to the primary sample sieve set. For each 17 L sip that was collected across the hydrograph, the primary sample was collected first, then covered with foil, and then the duplicate sample was collected, and then covered with foil. Upon completion of each sip, both sets of sieves were returned to the dedicated closed box to await the next sip. In total, four sips were collected over the course of the storm.

Sample extraction and analysis

The method used for microparticle extraction from stormwater included a density separation method modified from Stolte et al. (2015), which served to separate the microparticles from sediments in the sample. Briefly, the samples collected in the field using a 125 µm and 355 µm sieve were recombined together and passed over 106 µm and 500 µm sieves. Particles greater than 500 µm were sorted visually under a dissection microscope by manually picking particles that appeared to be plastic based on morphology and color. The smaller size fractions were density separated to separate plastic from other materials. Briefly, each size fraction was mixed with approximately 200 mL of CaCl₂ solution in a separatory funnel and left to sit until the material settled—generally overnight. The next day, the floating portion was filtered through a 20 µm polycarbonate filter and sealed into a petri dish for visual sorting. The portion sunk to the bottom was released over a 106 µm sieve, and the particles were then further separated into size fractions using 1,000 µm, 500 µm, 355 µm, and 125 µm sieves (given the irregular shapes of microparticles, particularly fibers, these are operational size fractions based on which sieved particles did and did not pass through, rather than a more accurate representation of the microparticle sizes based on measuring the length of each side). Individual particles in each size fraction were enumerated and sorted according to color and morphology under a stereomicroscope (Leica M80 Routine Stereomicroscope, 7.5–60x zoom; microscope.com model # G42PT-L3WLED, 10–80x zoom).

A subset of the total microparticles was measured and photographed. Overall, 39% of the particles were measured and imaged using ImageJ software.

For each sample, all particles for each morphology were analyzed by FTIR or Raman spectroscopy if there were less than ten particles for each morphology. If there were more than ten particles but less than 100 within a morphological category, at least ten particles were analyzed. If there were more than 100 particles, at least 10% were analyzed by Raman/FTIR spectroscopy to determine the chemical composition of the particle using a reference spectra library. Particles analyzed by spectroscopy were selected to represent all colors for each morphology and sample. All particles selected for Raman or FTIR analysis were placed on double-sided tape and the material type of a subsample of these extracted particles was identified using either Raman spectroscopy (Horiba Scientific Xplora Plus) with LabSpec6

software, or FTIR spectroscopy with an FPA-based Alpha II FTIR setup with OPUS/3D technology (Bruker Corporation).

In total, Raman and FTIR spectroscopy was conducted on 859 (approximately 7%) of the microparticles visually identified. Many black fragments that had a distinctive rubbery texture could not be conclusively identified via spectroscopy as to their composition.

Laboratory blanks were run for every set of ten samples at a minimum; in total three laboratory blanks were analyzed. All laboratory blanks were composed of reverse-osmosis water processed using the same methods as the field samples. Laboratory glassware was cleaned with detergent and water, followed by a triple-rinse with reverse-osmosis water. Laboratory practices to avoid procedural contamination included sealing all glassware from air as much as possible, working in a clean cabinet as much as possible, using a HEPA filter in the laboratory, and wearing cotton lab coats during laboratory analysis.

Analytical method recovery evaluation

Prior to commencing analyses of field samples, a laboratory study was conducted to assess the efficacy of the extraction methods. The model stormwater matrix consisted of water treated with reverse osmosis, to which was added plant material that was blended down to a fine size to create an organic model matrix. Replicates 1 and 2 were sieved first using a 500 µm sieve, then subjected to density separation, while Replicates 3 and 4 were subjected to density separation without pre-sieving.

Statistical analysis and treatment of blanks

Simple linear regression was used to evaluate the correlation between land-use types and particle concentrations in each watershed. Significance was determined at $p < 0.05$.

Laboratory and field blank results are reported alongside field sample results. Field samples were not blank corrected (i.e., blanks were not subtracted from the field samples) due to the non-uniform nature of the background field and laboratory contamination observed. Field data were qualified when particle counts (by morphology) were less than the average of the field and laboratory blanks plus two times the standard deviation. The field and laboratory blank data are reported so individual readers can make their own inferences regarding the data.

Loading estimate methods

Microparticle and microplastic loads to San Francisco Bay from the small tributaries were estimated using the Regional Watershed Spreadsheet Model (RWSM; Wu et al., 2017), a calibrated model previously developed by the RMP to estimate average annual volumes of water via stormwater and loads of mercury and polychlorinated biphenyls (PCBs) conveyed to

the Bay. Briefly, the RWSM uses characteristics of rainfall, land use, slope, and soil type to estimate annual flow volume for any given area at a parcel scale. The model then multiplies that flow estimate by the concentration of a contaminant of interest (in this case, microparticle concentration) assigned to the parcel's land use, resulting in an estimate of annual load for a pollutant of interest. The RWSM has been calibrated for hydrology and PCBs (Wu, et al., 2017). In this study, we calibrated the model for microparticles.

Estimated mean concentration factors for each land use were determined through manual calibration to the depth-integrated field sample observations. Depth-integrated sampling is superior to surface sampling for loads estimation because the samples represent microparticles distributed throughout the water column rather than simply the most buoyant particles at the surface. The manual calibration was assessed by mean bias, the root mean squared error, and the Nash-Sutcliffe model efficiency coefficient (NSE). The NSE is used to

assess the predictive power of hydrological and water quality models (Moriasi et al., 2012). The coefficient can range from -infinity to one, with one being a perfect model simulation matched to observed data. An NSE of zero indicates the observed mean is as good of a predictor as the model. Threshold values that indicate a model of sufficient quality have been suggested to be between 0.5 and 0.65.

Using the RWSM, we applied the derived concentration factors to each parcel based on its land use, and multiplied the factor by the runoff estimates around the region to estimate loads of microparticles in stormwater from the small tributaries surrounding the Bay. Microplastic loads were also estimated using the proportion of different particles identified as plastic by Raman or FTIR

spectroscopy multiplied by the total load into the Bay.



Results

Quality assurance results

ANALYTICAL METHOD RECOVERY EVALUATION

Recoveries of spiked particles in lab-prepared model stormwater are shown in Table 2.2. Recoveries were excellent for polyethylene terephthalate fragments, cellulose acetate beads, and polyethylene beads. Polystyrene fragments and polyester fibers were more challenging to extract and had lower recoveries (mean recovery 55% and 40%, respectively). Given the dark color of the matrix, picking out the brown polystyrene fragments was difficult, and given the shape of fibers, they can be hard to identify. These results suggest that counts for polystyrene fragments and polyester fibers may be biased low.

BACKGROUND CONTAMINATION: FIELD AND LABORATORY BLANKS

The three laboratory blanks had 52, 53, and 5 particles. Of these particles, 95% were fibers, predominantly white, blue, or clear. Two pieces of film and four fragments were also identified in one of the blanks. Based on discussions with laboratory personnel, many of the fibers found in the blanks may be due to contamination from laboratory materials such as Kimwipes. In total, seven particles were identified in the field blank—six fibers and one fragment. Fifty-three particles (the maximum found in the blanks collectively) is 35% of the lowest abundance found in the field samples (152 particles).

To assess whether blank contamination of the field samples was significant, a threshold for data qualification, the average of the laboratory and field blanks plus two times the standard deviation, was calculated for each particle morphology (i.e., 79 for fibers, 3 for films, and 5 for fragments). All field fiber concentrations were above the threshold of 79. For fragments, one sample had a low fragment count just below the threshold (count of 3, threshold of 5), and the vast majority of samples had counts that were substantially above the threshold. Very low counts of films were identified in the quality assurance blank samples, and four field samples had counts lower than the threshold (counts of 2, threshold of 3). The film particle counts should therefore be viewed with caution. Thresholds for foams and spheres were not developed because they were not detected in any of the blanks.

Table 2.2. Recovery of spiked microplastics in model stormwater sample.

Microplastic Type	Particle Size	Replicate 1 Recovery	Replicate 2 Recovery	Replicate 3 Recovery	Replicate 4 Recovery	Mean Recovery
Polyethylene terephthalate fragment (clear/white)	1 mm	4 (40%)	8 (80%)	9 (90%)	10 (100%)	78%
Polystyrene fragment (brown)	2 mm	5 (50%)	7 (70%)	4 (40%)	6 (60%)	55%
Cellulose acetate bead (red)	1 mm	3 (100%)	3 (100%)	3 (100%)	3 (100%)	100%
Polyethylene bead (green)	250-300 µm	10 (100%)	10 (100%)	5 (50%)	9 (90%)	85%
Polyester fiber (red)	3 mm in length	6 (60%)	6 (60%)	1 (10%)	3 (30%)	40%

PRECISION AND VARIABILITY: FIELD DUPLICATES

The total microparticle counts in the primary and duplicate field samples were 2,346 and 1,744 (relative percent difference (RPD) 30%), which suggests that for overall particle counts, there is moderately good precision and reproducibility in the method despite the heterogeneous nature of stormwater. Inspection of the particle morphology in the duplicate samples indicated that almost all of the variation was due to fragments (data not shown).

Particle occurrence and morphology

Microparticles were identified in stormwater from all 12 sites (Figure 2.3a). In total, 12,352 microparticles were enumerated from the 12 field samples and one field duplicate. Concentrations across the sites ranged from 1.3 microparticles/L to 30 microparticles/L (mean: 9.2 microparticles/L; median 6.7 microparticles/L), and are much greater than concentrations observed in wastewater effluent, where the mean was 0.063 microparticles/L (see Chapter 3 Wastewater).

The most abundant particle type was fragments (59%), followed by fibers (39%), and film (1%) (Figure 2.3b). Foam and spheres combined were less than 1% of the total. A majority of microparticles were black in color, except at Rodeo and Refugio creeks, where clear microparticles dominated the samples (Figure 2.3c). There was not a clear pattern of abundance, shape, or color with respect to land-use distributions in the watersheds (Figure 2.3d).

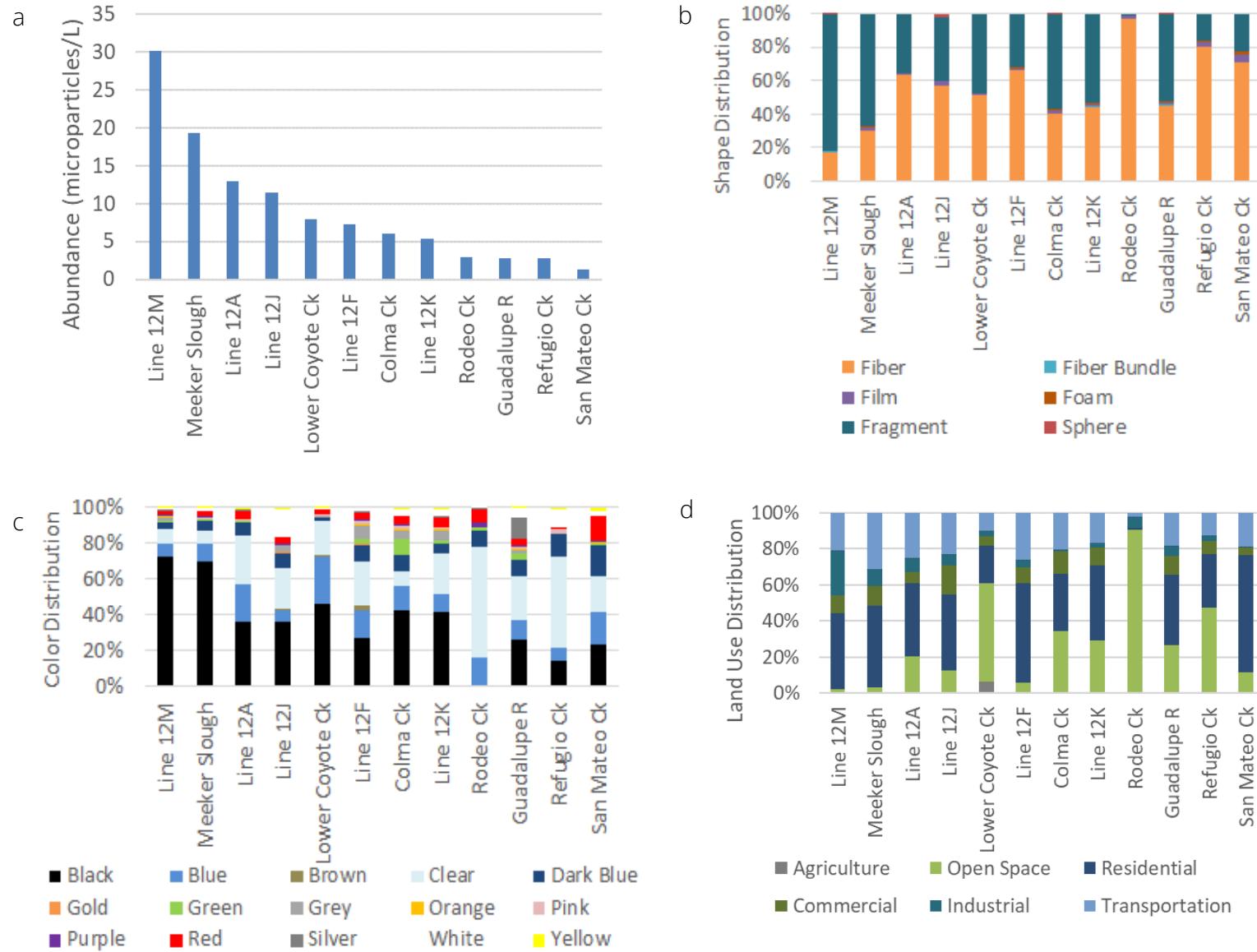


Figure 2.3. (a) Abundance, (b) shape, and (c) color distribution of microparticles collected at 12 locations in the San Francisco Bay Area (the field and duplicate sample at Line 12M are averaged); (d) land-use distribution within the 12 watersheds.

Generally, the smallest sieve fraction measured (125–355 µm) had the highest particle count, with microparticles becoming less abundant as the size fraction increased (Figure 2.4).

Fragments were the most abundant morphology in the 125–355 µm size fraction. Fragments in this size fraction accounted for 84% of the fragment total, and 49% of the entire sample set across all size fractions. The 125–355 µm group also had the greatest number of fibers, accounting for 43% of all fibers.

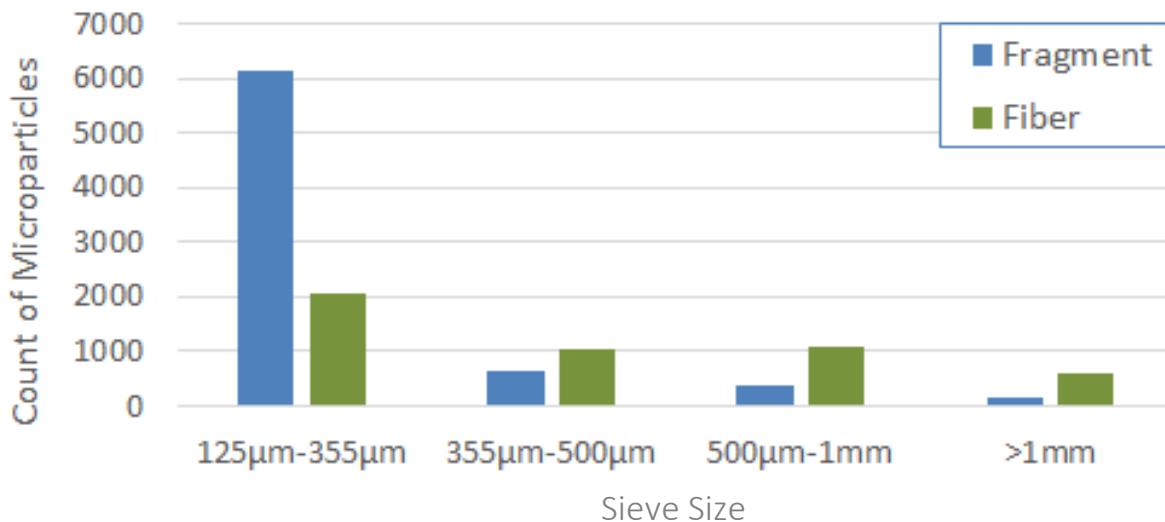


Figure 2.4. Particle morphology by size fraction (only fragments and fibers included). Note: sieve size bins are not evenly distributed.

Composition

A total of 859 microparticles (7% of the total) were further characterized using spectroscopy (Raman or FTIR; Figure 2.5), 804 of which were from field samples. Forty-three percent (346 microparticles) were identified as plastic.

Black fragments that had a distinctive rubbery texture when handled with tweezers were abundant in stormwater samples. However, these particles were challenging to analyze using Raman and FTIR spectroscopy because it was difficult to obtain a good spectrum and, therefore, polymer identification. The Raman spectra for these particles often matched to carbon black or materials with similar spectra (e.g., diamond-like carbon, ivory black, Van Dyke brown, vine black, amido black, lamp black). Based on these spectra, it was not possible to conclusively identify these particles as plastic polymers. Only one particle had a spectrum that matched tire rubber, and was therefore classified as rubber.

However, secondary characteristics such as compressibility, color, and texture suggested that these particles were similar to other black, rubbery particles that have been positively identified as rubber using FTIR spectroscopy. Other studies have also identified abundant

presence of black fragments in the environment that may potentially come from car tires (Bråte et al., 2018; Gray et al., 2018; Unice et al., 2013).

Therefore, the black, rubbery fragments that had these spectral matches to carbon black and similar materials were classified as unknown potentially rubber. Of note, carbon black is a major ingredient in tire rubber, so spectral matches to this material are not inconsistent with this classification (Edil, 2008). The remaining black fragments with “rubbery” texture were classified as other materials based on more specific spectral matches.

A total of 171 microparticles were classified as unknown potentially rubber, which represents 84% of black, rubbery fragments that were analyzed by spectroscopy, or 44% of all fragments analyzed (21% of all particles analyzed). Only one site, Rodeo Creek (a mostly rural site), had zero rubbery fragments.

The next most abundant category was fibers identified as anthropogenic unknown (11%). Anthropogenic unknown indicates a fiber that is dyed with a dye or coloring agent, but for which the underlying fiber composition could not be identified (i.e., it may or may not be plastic). Following anthropogenic unknown fibers in abundance were polyester fibers (7.2%), cellulose acetate fibers (4.6%), polypropylene fragments (4.0%), and polyethylene fragments (3.9%) (Figure 2.5; concentrations of composition categories at each site are shown in Appendix Figure A-2.2). Polyethylene terephthalate, which is widely used in bottles for water, beverages, and cleaners was identified in seven fragments (0.9%). The remaining 47% of particles analyzed were distributed among various polymer-shape categories.

Of the fibers that were characterized, 15% were natural (e.g., cellulosic or protein-based), while 35% could not be characterized because of interference in the spectral data from dyes in the fiber.

One percent of the whole dataset was identified as film (113 pieces), and 63% were further identified by spectroscopy. The most abundant film type was polyethylene film (3.0% of microparticles that were examined spectroscopically), followed by unknown film (2.0%), and anthropogenic unknown film (0.5%).

Forty-six pieces of foam were identified in the dataset, and 38 of them underwent spectroscopy. All foams except for unknown foams (2%) contributed less than 3% to the total samples that underwent spectroscopy. The second highest foam category was anthropogenic

unknown foam (0.7% of total). Only three pieces of polyurethane foam (0.4%) and one piece of polystyrene foam (0.1%) were identified. A total of twelve polystyrene microplastics (0.9%) were identified (including fragments (8), a fiber (1), a film (1), a foam (1), and a sphere (1)).

Twenty-five spheres were identified in the dataset, and 80% of them were clear. Seventeen spheres (68% of spheres) underwent spectroscopy and most of them (11 of 17) were identified as glass, while three were identified as polyethylene in the 250–500 µm size class, which suggests they could be microbeads.

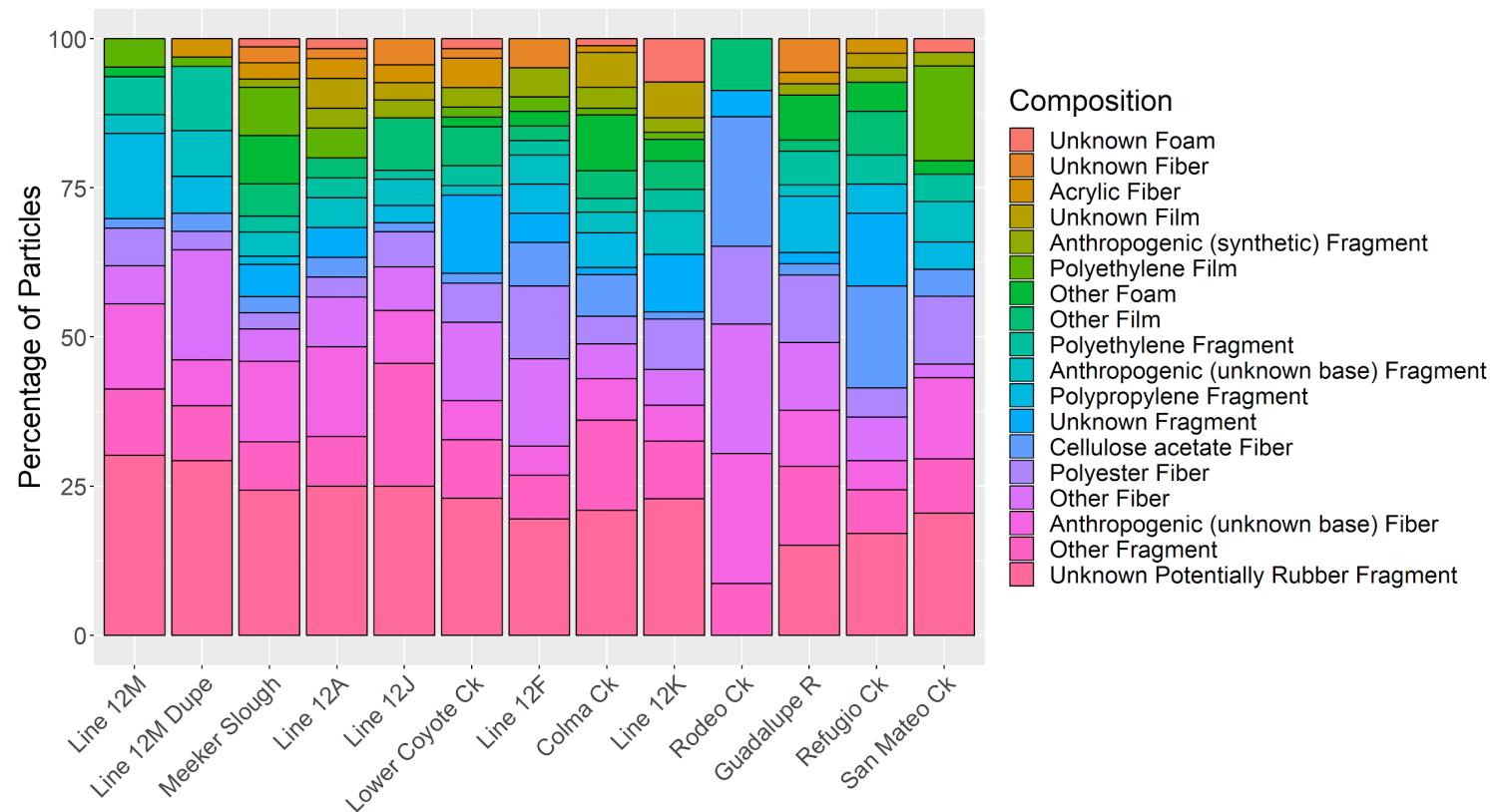


Figure 2.5. Polymer type distribution for microparticles. Polyethylene, polypropylene, cellulose acetate, polyester, and rubber are considered plastic. The most abundant 14 categories of particles are listed, while the abundances of all other particles are combined into the categories labeled “Other.”

Watershed attributes in relation to particle counts and morphology

Relationships among individual land uses and combined urban and non-urban land uses were evaluated relative to total microparticle concentrations using simple linear regression (Figure 2.6). Particle concentrations were significantly and positively correlated with percent industrial area. This correlation was significant even with the very elevated concentration from Line 12M removed ($p = 0.012$). All other correlations evaluated were not statistically significant at $p < 0.05$, but a suggestive positive correlation was observed with urban area ($p = 0.086$), while suggestive negative correlations were observed with open space ($p = 0.082$) and non-urban area ($p = 0.086$). Of note, land uses are not the only watershed “characteristic” that can be related to microparticle or microplastic concentrations. For example, unlike many other pollutants, trash and litter are also generated in the channels themselves via illegal dumping and homeless encampments, and are considered a source of secondary microplastics.

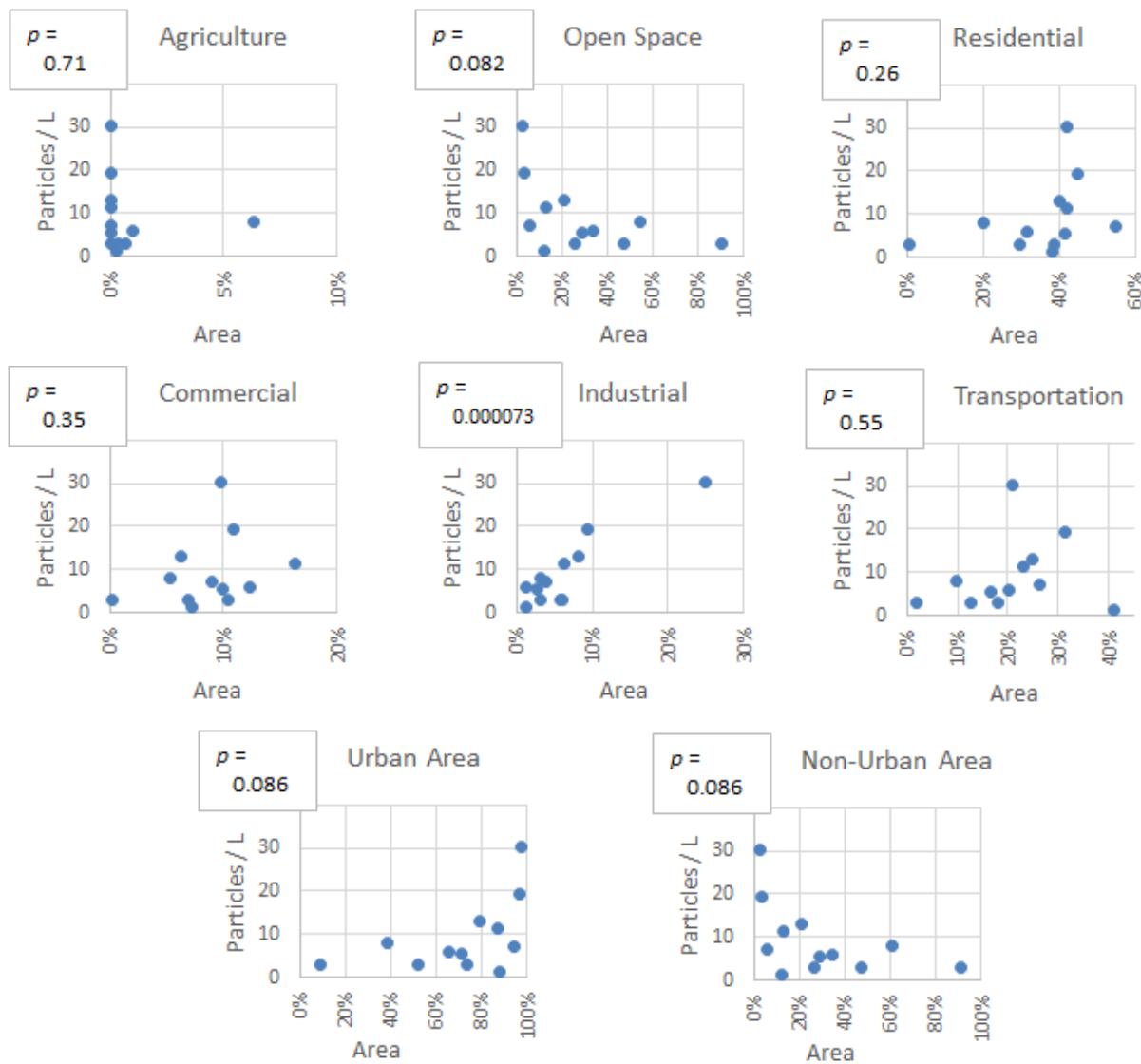


Figure 2.6. Simple linear regression relationships between microparticle concentrations and individual land uses, and combined urban and non-urban land uses.

Regional loads estimates

We developed two model calibrations using a standard manual process based on the “best fit” of simulated concentration to measured concentration for each watershed. Concentration factors for each land use were initially guided by our conceptual models of microplastic loads from different land uses in the environment, and then refined based on evaluation metrics. Only a single, limited dataset of mixed-use watersheds was used to derive and refine these concentration factors. As such, the results of this modeling effort should be considered preliminary.

The first model fit the calibration coefficients to achieve the best model evaluation metrics comparing the simulated concentrations to the depth-integrated field sample concentrations (root mean square error, percent bias, and Nash-Sutcliffe Model Efficiency Coefficient (NSE); Table 2.3). Industrial area in the first model calibration was determined to be a strong driver of microparticle transport to the Bay; however, there is not yet an established rationale for why industrial areas should have yields significantly greater than other highly urban land-use categories such as commercial and transportation. Therefore, a second model calibration was developed, forcing the industrial coefficient to be equal to the next highest land-use category—transportation (Table 2.3). This calibration led to a model that was less than half the NSE of the first model, indicating the simulated and measured data were not as closely matched in the second calibration (based on the NSE). The simulated vs. measured concentration graph for the first model is shown in Figure 2.7 and for the second model is shown in the Appendix (Figure A-2.1).

In the first model (Model 1), although the calibrated concentrations simulated for each watershed closely matched the empirically measured concentrations for most watersheds (Figure 2.7), one third of the watersheds have lower measured than simulated concentrations. One of the low outliers is Guadalupe River, where our sampling team used an alternative method (the bucket) for sampling. Another low outlier is Rodeo Creek, which has an industrial area within the watershed, which raises the simulated concentration; however, that industrial area is not directly hydrologically connected to the creek via impervious surface. An additional location, Refugio Creek, is notable as a more recently developed area, which is not accounted for in the land use data.

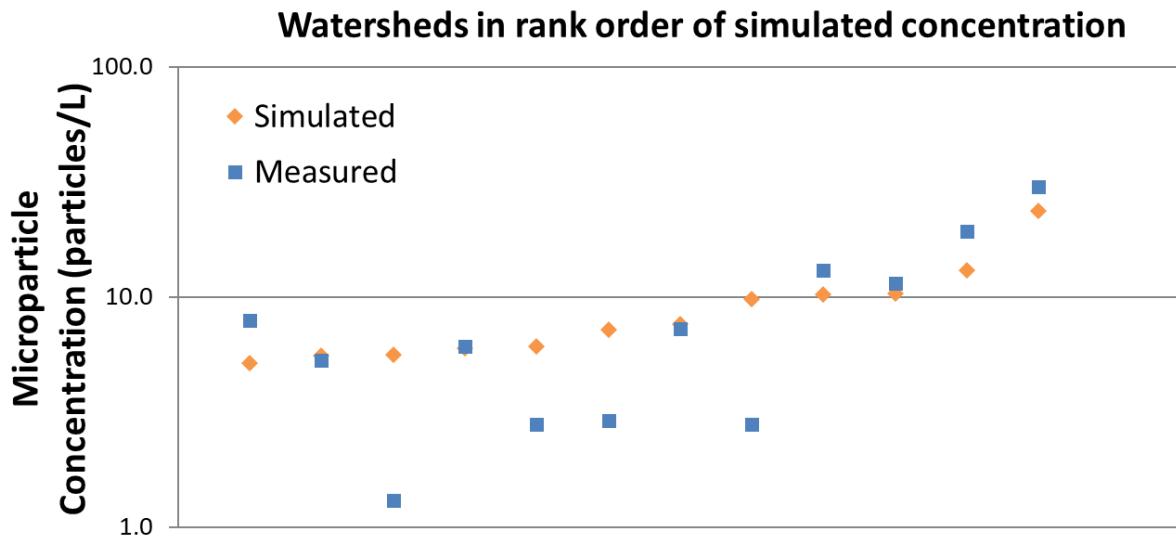


Figure 2.7. Simulated vs. measured microparticle concentration results for the first and strongest model calibration. Watersheds in rank order of simulated concentration.

Table 2.3. Microparticle concentration coefficients and model evaluation metrics for each of the two model calibrations presented.

<i>Metrics; coefficients in microparticles/L</i>	<i>Model 1</i>	<i>Model 2</i>
Industrial Coefficient	62	24
Transportation Coefficient	10	24
Commercial Coefficient	5	1
Residential Coefficient	1	0.5
Agriculture and Open Space Coefficient	0.1	0.1
Root-mean-squared Error	4.0	6.3
Percent Bias	0.0	-0.3
Nash-Sutcliffe Model Efficiency Coefficient	0.76	0.42

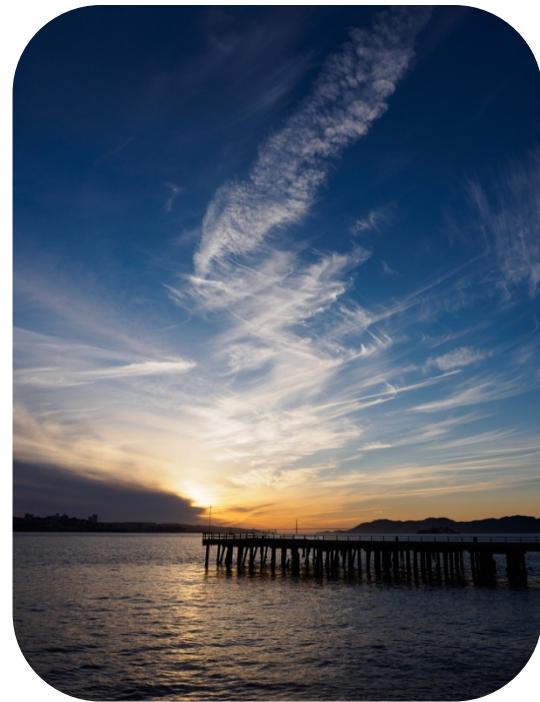
Based on the first and strongest model (Model 1), the regional load estimate of microparticles to San Francisco Bay from the small tributaries is 10.9 trillion microparticles per year (Table 2.4). Note that the small tributaries freshwater input to the Bay represents 6% of the total freshwater input (McKee et al., 2013; Oram et al., 2008; Wu et al., 2017); other inputs include inflow from the large Sacramento-San Joaquin watershed that flows through the Delta, as well as direct precipitation and wastewater inputs.

It is estimated that 74% of the microparticles from the small tributaries are flowing from the industrial areas around the Bay, which comprise just 6% of the land area (Table 2.4; Figure 2.8). Conversely, agriculture and open spaces comprise 61% of the land area draining from the small tributaries into the Bay, yet yield just 1% of the total load. These estimates should be treated with caution given the paucity of the data upon which the model calibration depends, and the RPD (30%) of the field duplicate. Further data collection, model calibration, and model validation is recommended.

Based on the particles in field samples that underwent spectroscopy, 43% of the particles were plastic and 47% were identified as anthropogenic unknown, unknown potentially rubber, or unknown, with the rest considered non-synthetic or natural in origin. This suggests that, based on the model load estimates and depending on what proportion of the anthropogenic unknown or unknown particles are plastic (0–50%), between 4.7 and 7.2 trillion microplastics enter San Francisco Bay via the small tributaries annually.

Table 2.4. Land use and microparticle model loadings estimate summary from small tributaries to the San Francisco Bay.

	Total	Ag/Open	Industrial	Transportation	Commercial	Residential
Area (km²)	6725	4091	379	546	363	1346
Proportion of Land Area	100%	61%	6%	8%	5%	20%
Runoff Volume (Mm³)	1731	1033	129	194	104	271
Proportion of Runoff Volume	100%	60%	7%	11%	6%	16%
Load (10¹² particles)	10.9	0.10	8.0	1.9	0.52	0.27
Proportion of Load	100%	1%	74%	18%	5%	2%
Yield (10⁶ particles/km²)	1615	25	21164	3547	1432	201



Discussion

Comparison to observations in other regions

These results can be compared to a limited number of available studies on microparticles and microplastics in creeks and rivers (Table 2.5), but caution should be used. Of these existing studies, the collection techniques used vary substantially, making direct comparison challenging (Dris et al., 2018; GESAMP, 2016). Major differences in collection methods include different sieve or net mesh sizes used to capture microparticles, sampling at different depth locations in the water column, and sampling during dry as opposed to storm conditions.

The range of microparticle concentrations we measured across the 12 watersheds varied from 1.3 to 30 microparticles/L. These concentrations are comparable to measurements reported in several studies from China³ (Lin et al., 2018; Wang et al., 2017; Yan et al., 2019), the Los Angeles River (Moore et al., 2011), and El Cerrito (located in the San Francisco Bay Area; Gilbreath et al., 2019). Measured concentrations were significantly higher than those reported in Europe (Dris et al., 2015; Lechner et al., 2014), Japan (Kataoka et al., 2019), and an additional study from China (Xiong et al., 2019).

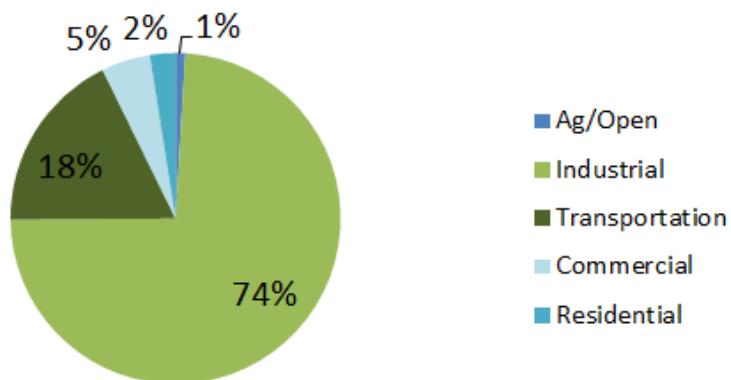


Figure 2.8. Regional land use distribution of microparticle contributions to small tributaries of San Francisco Bay based on Model 1, which suggests industrial land uses may be associated with higher levels of microparticles.

³ Of note, the Pearl River Delta has been identified as a plastic trash hotspot, and is one of the top ten rivers carrying plastic loads to the sea (Schmidt et al., 2017).

Table 2.5. Studies of microparticles and microplastics in freshwater streams reported in the literature, including key collection method differences.

Location	Sampling Condition (dry, wet, unknown)	Surface or depth-integrated sampling	Plastic ID Method	Smallest sieve size (µm)	Abundance (# particles/L)	Reference
Danube River, Austria	Unknown	Surface water	Visual	500	0.00032	Lechner et al., 2014
29 rivers in Japan	Dry	Surface water	Visual, FTIR on subset	335	0.0016 (mean of 29 sites)	Kataoka et al., 2019
Yangtze River, China	dry	Surface water	Visual, Raman / FTIR on all	333	0.0009	Xiong et al., 2019
Los Angeles River, USA	Wet	Surface, mid, and near-bottom water	Visual	333	13	Moore et al., 2011
Los Angeles River, USA	Dry	Surface, mid, and near-bottom water	Visual	333	0.022	Moore et al., 2011
San Gabriel River, USA	Wet	Surface, mid, and near-bottom water	Visual	333	0.34	Moore et al., 2011
San Gabriel River, USA	Dry	Surface, mid, and near-bottom water	Visual	333	0	Moore et al., 2011
Coyote Creek, USA	Wet	Surface, mid, and near-bottom water	Visual	333	0.074	Moore et al., 2011
Coyote Creek, USA	Dry	Surface, mid, and near-bottom water	Visual	333	0.007	Moore et al., 2011

Location	Sampling Condition (dry, wet, unknown)	Surface or depth-integrated sampling	Plastic ID Method	Smallest sieve size (μm)	Abundance (# particles/L)	Reference
11 sites on the Rhine River, Switzerland, Germany, Netherlands	Unknown	Surface water	Visual, FTIR on subset	300	0.0056 (mean of 11 sites)	Mani et al., 2015
El Cerrito rain garden catchment, USA	Wet	Depth-Integrated	Visual, Raman / FTIR on subset	125	1.6	Gilbreath et al., 2019
12 small tributaries to San Francisco Bay, USA	Wet	Depth-integrated	Visual, Raman / FTIR on subset	125	9.2 (mean of 12 sites)	This study
River Seine, Paris, France	Unknown	Surface water	Visual	80	0.030	Dris et al., 2015
Yangtze River, China	Unknown	unknown	Visual, FTIR on subset	50	2.5	Wang et al., 2017
Hanjiang River, China	Unknown	unknown	Visual, FTIR on subset	50	2.9	Wang et al., 2017
26 sites on the Pearl River along Guangzhou City, China	Unknown	Surface water	Visual, Raman / FTIR on subset	50	20	Yan et al., 2019
14 sites on the Pearl River along Guangzhou City, China	Unknown	Surface water	Visual	20	2.7	Lin et al., 2018

An initial review of the literature suggests mesh size is one of the more important variables affecting the measured concentrations of microparticles (Covernton et al., 2019). Different mesh sizes were used to capture microparticles, varying between 20 µm (Lin et al., 2018) up to 500 µm (Lechner et al., 2014). Many of the studies reviewed for this report used mesh sizes between 300–335 µm. Studies using these larger mesh sizes generally reported lower concentrations, often less than 0.1 particles/L (Kataoka et al., 2019; Lechner et al., 2014; Mani et al., 2016; Xiong et al., 2019). In contrast, studies employing smaller mesh sizes (20–50 µm) generally reported concentrations of microparticles more similar to our observations (Lin et al., 2018; Wang et al., 2017; Yan et al., 2019). One exception is a study of the River Seine in Paris, which used an 80 µm mesh size but only reported 0.030 particles/L (Dris et al., 2015).

Smaller mesh sizes are able to trap a greater proportion of microparticles. In this study, 68% of the total particle count was found in the 125–355 µm size fraction. It is unknown how many more microparticles we would have captured if we had used an even smaller mesh or sieve size. Other studies show trends indicating that the smallest fractions make up the majority of the samples (Lin et al., 2018; Wang et al., 2017; Xiong et al., 2019).

Storm conditions during sample collection may also be a highly influential factor in measured concentrations of microparticles. This project monitored creeks and tributaries during wet weather events because studies in the Bay Area indicate that 95% of flow into the Bay from the small tributaries is rainfall-driven (McKee et al., 2003). Several studies have sampled during dry weather (Kataoka et al., 2019; Xiong et al., 2019; Yonkos et al., 2014), on a monthly time interval that disregards the flow condition (Dris et al., 2018), or do not report the flow condition in the study (Dris et al., 2015; Lechner et al., 2014; Lin et al., 2018; Mani et al., 2015; Wang et al., 2017; Yan et al., 2019). Our conceptual model for microplastics is similar to many other pollutants; microplastics are entrained from the landscape in stormwater runoff and have greater concentrations during storm-driven flows. Moore et al. (2011) found a significant difference between the maximum concentrations measured on the Los Angeles River after a storm event (13 microparticles/L) vs. the concentration measured at the same location during dry weather (0.022 microparticles/L). Note, Moore et al. (2011) used a net mesh size of 333 µm, which was larger than our mesh size but was consistent between the two sampling periods. Although most studies report much lower concentrations than we measured in this study, it may be the result of the flow condition during sampling. Therefore, caution should be used in comparing between samples collected during differing flow conditions.



An additional variable that complicates regional comparisons is the depth at which microparticle samples are collected. Microparticles are often sampled using a mesh net to capture particles at and just below the surface. This method will bias toward collecting more buoyant microparticles, and therefore may also bias the types and distribution of polymers detected. It will also provide concentration data that is not appropriate for estimating loads. In this study, we collected a vertical depth-integrated sample by slowly moving the sampling tube up and down throughout the water column during sample collection. While this technique provides a more appropriate sample for calculating loads, it may instead be missing a representative portion of the most buoyant particles.

For this study, it was not feasible to profile horizontally across the stream channel. One study that evaluated microparticle concentrations across a 67 m wide channel during low flow showed significant differences between the banks and the middle of the channel (Dris et al., 2018). We hypothesized that during storm flow, in our relatively small channels (all less than 40 m wide and most less than 10 m wide), turbulence in the flow path would disperse microparticles relatively evenly across the channel. Further study would be necessary to evaluate this hypothesis.

Taken together, all of these varying factors of study design make it difficult to assess whether San Francisco Bay Area stormwater had higher, similar, or lower concentrations than other areas studied to-date.

Insights on microplastics sources

While there is no clear trend as to which types of particles dominate stream and stormwater flows worldwide, in some cases it is possible to make inferences as to sources. For example, in a study on the Danube River, industrial raw materials comprised of pellets, flakes, and spheres dominated the load (Lechner et al., 2014), and at 11 sites on the Rhine River, 60% of the microplastics were spheres believed to originate from industrial processes (Mani et al., 2015). In contrast, on the Pearl River, Lin et al. (2018) found that fibers dominated the total number of particles (81%).

In this study, fragments comprised 59% of the total and fibers comprised 39%. Similar to our study, Baldwin et al. (2016) found that fibers and fragments were the most abundant morphologies in 29 Great Lakes tributaries. Insights on the sources of Bay Area fragments and fibers are presented below.

RUBBER FRAGMENTS ARE LIKELY A MAJOR SOURCE OF MICROPLASTICS TO THE BAY

In this study, the laboratory was able to identify black fragments that had a distinctive rubbery texture when held by tweezers. In total, these black fragments with rubbery texture comprised 48% of the entire dataset. Not all of these particles were analyzed by spectroscopy; 84% of the black rubbery fragments that were further analyzed by spectroscopy were classified as unknown potentially rubber based on spectra that were inconclusive. Because these particles are particularly challenging to analyze by spectroscopy due to their small size, rubbery texture, and fluorescence from other compounds, only one of these particles could be confirmed as rubber using Raman spectroscopy. Although potential sources of rubber to stormwater are diverse, these rubber fragments could be derived from tire wear (Boucher and Friot, 2017; Gray et al., 2018; Kole et al., 2017).

Kole et al. (2017) estimated global emissions of microplastics from tire wear were, on average, 0.81 kg/year/capita worldwide. In the U.S., where there is a high number of cars per capita, as well as longer average commutes, the estimate was much higher at 4.7 kg/year/capita. Given the Bay Area population of 7.7 million people, and using 4.7 kg/year/capita, the estimate for microplastics generation from tire wear is 36.2 million kg/year. Not all of that will reach the Bay through small tributaries.⁴ Blok (2005) estimated that 70% of tire wear debris that remains on the road surface (as opposed to depositing next to the road on a less pervious surface) may run off with rainfall. Based on this estimate, as well as the polymer, color, and morphology of

⁴ Considering the estimated rubber fragment load to the Bay, and accepting the Kole et al. (2017) estimate of 4.7 kg/year/capita for our population of 7.7 million people, if all rubber fragments in stormwater were from tire wear, then 24% of all the rubber fragments from tires generated in the landscape is ultimately transported via stormwater to the Bay.

particles from the current study, the weight of evidence suggests that microplastics from rubber, which is potentially from tire wear, are a significant source of microplastics to the Bay.

Not all watersheds had rubbery fragments. Line 12M contributed 49% of the total count of rubbery fragments based on visual identification and texture, whereas Rodeo Creek (a mostly rural site) had no rubbery fragments at all. Although there are road surfaces in the Rodeo Creek watershed and, therefore, there is likely at least some tire wear, the drainage area is mostly rural and the roads are more likely to be disconnected hydrologically from the creek (we have not field-verified this).

Line 12M, on the other hand, is almost entirely urban (98%), and therefore the roads and paved surfaces are more directly hydrologically connected to the drainage channel via the storm drainage system. This connection allows rubbery fragments that are entrained in stormwater to be directed to storm channels and consequently to the Bay. Other potentially important factors include multiple used auto parts operations and a high traffic volume freeway. Although the freeway is downstream of our sampling site, the site is tidal and therefore microparticles could move upstream during flood tide. Fragments smaller than 300 μm can also be carried atmospherically (Allen et al., 2019), increasing the possibility that the nearby freeway may contribute tire particles to the drainage channel.

COMPOSITION AND SOURCES OF FIBERS VARY WIDELY

Fibers, which are ubiquitous throughout the environment (Dris et al., 2018; GESAMP, 2016), accounted for 39% of the total microparticle data set. The composition of fibers varied from plastic to cellulose to unknown. Abraded fibers from textiles and clothing likely make up a large percentage of fibers in all environmental compartments. In stormwater, there are a number of potential sources of fibers including: the use of geotextiles in engineering, industrial laundromats and residential dryers expelling fibers into the air, abrasion of fibers from textiles and clothing in the outdoor environment (GESAMP, 2016), atmospheric fallout (particularly associated with rainfall events), and degradation of cigarette filters, food packaging, and food containers.

No single polymer comprised the majority of fibers, but polyester was the largest category, contributing 20% of all fibers analyzed by spectroscopy. Polyester fibers may come from textiles or industrial applications; polyester fibers are used in car tires for stability, fabrics for conveyor belts, safety belts, coated fabrics, or other applications when resiliency is a desirable characteristic. The second largest identified category of fiber was cellulose acetate (13% of all analyzed fibers); cellulose acetate fibers are present in cigarette filters, as well as textiles and high absorbency products (e.g., diapers).

LIMITED CONNECTIONS TO SINGLE-USE ITEMS

Microplastics that may be linked to common single-use items, a focus of current plastic pollution policy actions and consumer education campaigns, were a measurable but limited portion of the microplastics identified. After unknown potentially rubber, polyethylene (8% of fragments) and polypropylene (8%) were the most significant polymers identified in the fragments analyzed. Only a small number of polyethylene terephthalate (PET) fragments (2%) were identified. Polyethylene, polypropylene, and PET are used in a wide variety of items, but notably, are also used to make many single-use products, including food and beverage containers, plastic straws, six-pack rings, and disposable cutlery. These single-use items are also among the most common beach litter items and feature prominently on the Better Alternatives Now (B.A.N.) list as products to phase out of use and production (Allen et al., 2017).

Other types of polymers and morphologies may be linked to single-use items. Polyethylene film, which represented 3.0% of total microparticles identified, is commonly used in plastic bags and wraps. Expanded polystyrene foam, used in a variety of food and other packaging, can fragment into foam particles in the environment. A small number of polystyrene particles were identified (3%) in stormwater, including nine fragments, one foam, one fiber, one film, and two spheres. As noted previously, cellulose acetate, which represented 5% of the identified microparticles, is used in a wide variety of products, including cigarette filters.

One hypothesis for why particles potentially related to these common single-use litter items are only a limited portion of the microplastics identified in stormwater is that significant actions to prevent or intercept trash (defined as particles greater than 5 mm) have been put into place in the Bay Area over the last ten years. Another hypothesis is that larger plastic debris may still be trash sized (greater than 5 mm) within the pathway. Longer exposure to sunlight and mechanical action may be necessary before such items are transformed to microplastic size. A notable proportion of particles potentially linked to single-use items was observed in the surface waters of the Bay (Chapter 4 Surface Water), a finding that supports this hypothesis. Therefore, while it is not possible to identify the original product that microparticles originated from, reducing plastic litter is expected to help address an important source of microplastics to the Bay.

Industrial land use and microparticle discharges

Given the results of this study, it appears that industrial areas may be important contributors of microplastics to the Bay. Industrial land use was particularly well-correlated with microparticle concentrations, and although the loads modeling effort is built upon data from just 12 watersheds sampled just once each, this initial model calibration suggests that loads of microparticles from the industrial area of the San Francisco Bay watershed (6% of the total

area) could export more than 70% of the total microparticle load to San Francisco Bay (Figure 2.8). It remains unclear whether there is actually more microplastic generated in industrial land areas, or if the transport mechanism is more efficient due to the high levels of imperviousness in these areas. It is also unclear whether other land uses (e.g., transportation) are as important as industrial areas, but are somehow masked due to a lack of understanding of the transport mechanisms, outdated land use data layers, or other factors.

In modeling stormwater-related contaminants, land use categories are considered a proxy for commonly occurring activities associated with contaminant discharges. The conceptual understanding of microplastic discharge has not typically placed such a strong emphasis on industrial activities and land use (Boucher and Friot, 2017; Sutton and Sedlak, 2017). It would also be appropriate to explore alternative models to interpret measurements and predict loads. For example, given the large proportion of potentially tire-derived microplastics, an analysis that focuses more exclusively on correlating concentrations relative to proximity to roadways might provide insights. Additionally, microparticle concentrations may be better associated with imperviousness, a characteristic that promotes rapid movement of contaminants from land into stormwater channels.

A number of caveats should be considered when reviewing the output of this model. First, microparticle concentrations are likely to vary within and between years. We have only a single storm composite to characterize the concentration from each watershed. Further, we used data from 12 watersheds to calibrate the model, and the individual land uses in these watersheds may not vary sufficiently from watershed to watershed to enable a robust calibration for some land uses. In addition, the RWSM hydrological model has an error range of $\pm 30\%$ for flow. Nonetheless, this first attempt at estimating microparticle loads represents a starting point for characterizing loads from watersheds in the region.

Conclusions

This study reports microparticle and microplastic concentrations in stormwater runoff from 12 small tributaries to San Francisco Bay. Concentrations at each site ranged from 1.3 microparticles/L to 30 microparticles/L, which is greater than concentrations reported in many other studies, likely because our study targeted high flow stormwater runoff conditions and used a smaller sieve size than most other studies (125 µm vs. greater than 300 µm).

The most abundant particle type was fragments (59%), with almost half the entire microparticle count being black rubbery fragments that were likely rubber particles, with vehicle tires as one likely source. Industrial land use was particularly well-correlated with microparticle concentrations, and a simple calibrated loadings model was used to calculate an estimated annual load to San Francisco Bay of 10.9 trillion microparticles; we estimate that 43–66% of this load is plastic, suggesting an annual discharge to the Bay of 4.7 to 7.2 trillion microplastics from stormwater.

The present study has helped address major data gaps on concentrations of microplastics in Bay tributaries, and presents an estimate of the total stormwater microplastics load to San Francisco Bay. It also highlights the potential importance of tire wear as a likely major contributor of microplastics into the Bay. Although other specific polymers may comprise a smaller percent of the total, because stormwater loads are so much greater than wastewater loads, even a polymer making up a very small percentage of the total would still be very significant relative to the entire wastewater load (e.g., a plastic polymer that is 1% of the stormwater microplastic load would be three to five times greater than the entire wastewater microplastic load).

Data interpretation suggests that the industrial land-use category merits further exploration as having the potential to discharge higher levels of microplastics via stormwater. Identifying land uses, attributes, or activities associated with higher discharges can inform prioritization of stormwater management of microplastics.

References

- Allen, K., Cohen, D., Culver, A., Cummins, A., Curtis, S., Eriksen, M., Gordon, M., Howe, A., Lapis, N., Prindiville, M., Thorpe, B., Wilson, S., 2017. Better Alternatives Now B.A.N. List 2.0. 5 Gyres, Algalita, Californians Against Waste, Clean Production Action, Plastic Pollution Coalition, Responsible Purchasing Network Story of Stuff, Surfrider Foundation, UPSTREAM.
- Allen, S., Allen, D., Phoenix, V.R., Le Roux, G., Durández Jiménez, P., Simonneau, A., Binet, S., and Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. *Nature Geoscience*, 12: 339-344. <https://doi.org/10.1038/s41561-019-0335-5>.
- Association of Bay Area Government (ABAG), 2006. Existing Land Use in 2005: Data for Bay Area Counties. 44 pg report and CD with GIS data layers.
<https://store.abag.ca.gov/projections.asp#elu>
- Baldwin, A.K., Corsi, S.R., Mason, S.A., 2016. Plastic debris in 29 Great Lakes tributaries: Relations to watershed attributes and hydrology. *Environmental Science & Technology* 50, 10377–10385. <https://doi.org/10.1021/acs.est.6b02917>
- Bay Area Stormwater Management Agencies Associations (BASMAA), 2014a. Tracking California's Trash (TCT) Literature Review. State Water Resources Control Board Grant Agreement No. 12, 420-550. Prepared by EOA, Inc and 5 Gyres.
- Bay Area Stormwater Management Agencies Associations (BASMAA), 2014b. Tracking California's Trash (TCT) Sampling and Analysis Plan, State Water Resources Control Board Grant Agreement No. 12, 420-550, Prepared by Geosyntec Consultants, 5 Gyres, and EOA, Inc.
- Bay Area Stormwater Management Agencies Associations (BASMAA), 2016. Tracking California's Trash (TCT) Testing Trash "Flux" Monitoring Methods in Flowing Water Bodies, State Water Resources Control Board Grant Agreement No. 12-420-550, Prepared by 5 Gyres.
- Blok, J. 2005. Environmental exposure of road borders to zinc. *Science of the Total Environment*. 348 (Sept). 173-190. <https://doi.org/10.1016/j.scitotenv.2004.12.073>
- Boucher, J. and Friot D., 2017. Primary Microplastics in the Oceans: A Global Evaluation of Sources. Gland, Switzerland: IUCN. 43pp.
- Bråte, I.L.N., Hurley, R., Iversen, K., Beyer, J., Thomas, K.V., Steindal, C.C., Green, N.W., Olsen, M., Lusher, A., 2018. *Mytilus spp.* as sentinels for monitoring microplastic pollution in Norwegian

coastal waters: A qualitative and quantitative study. Environmental Pollution 243, 383–393. <https://doi.org/10.1016/j.envpol.2018.08.077>

California Regional Water Quality Control Board San Francisco Bay (CRWQCB SF Bay), 2015. RESOLUTION No. R2-2015-0024. May 15, 2015.

City of San Jose, 2016. Direct Discharge Trash Control Program. Supplemental Submission. Submitted in accordance with provision Section C.10.e.ii of NPDES Permit No. CAS612008. May 27, 2016.

Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., and Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling techniques in seawater studies with implications for ecological risk assessment. Science of the Total Environment 667 (June): 124–32. <https://doi.org/10.1016/j.scitotenv.2019.02.346>.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., and Tassin, B., 2015. Microplastic contamination in an urban area: A case study in greater Paris. Environmental Chemistry 12(5): 592. <https://doi.org/10.1071/EN14167>.

Dris, R., Gasperi, J., Rocher, V., and Tassin, B., 2018. Synthetic and non-synthetic anthropogenic fibers in a river under the impact of Paris megacity: Sampling methodological aspects and flux estimations. Science of the Total Environment 618: 157–164. <https://doi.org/10.1016/j.scitotenv.2017.11.009>.

Edil, T.B., 2008. A review of environmental impacts and environmental applications of shredded scrap tires, in: Hazarika, H., Yasuhara, K. (Eds.), Scrap Tire Derived Geomaterials - Opportunities and Challenges. Taylor & Francis Group, London, pp. 3–18.

EOA, Inc., 2014. San Francisco Bay Area Stormwater Trash Generation Rates – Final Technical Report. Prepared for Bay Area Stormwater Management Agencies Association (BASMAA).

GESAMP, 2016. Sources, fate and effects of microplastics in the marine environment: Part two of a global assessment (Kershaw, P.J., and Rochman, C.M., eds). (IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/ UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection). Rep. Stud. GESAMP No. 93, 220 p.

Gilbreath, A.N. and McKee, L.J., 2015. Concentrations and loads of PCBs, dioxins, PAHs, PBDEs, OC pesticides and pyrethroids during storm and low flow conditions in a small urban semi-arid watershed. Science of the Total Environment 526: 251-261.

- Gilbreath, A.N., McKee, L.J., Shimabuku, I., Lin, D., Werbowski, L.M., Zhu, X., Grbic, J., and Rochman, C., 2019. Multiyear water quality performance and mass accumulation of PCBs, mercury, methylmercury, copper and microplastics in a bioretention rain garden. *Journal of Sustainable Water in the Built Environment* 5(4): 04019004.
- Gray, A.D., Wertz, H., Leads, R.R., and Weinstein, J.E., 2018. Microplastic in two South Carolina estuaries: Occurrence, distribution, and composition. *Marine Pollution Bulletin* 128: 223-233. <https://doi.org/10.1016/j.marpolbul.2018.01.030>
- Kataoka, T., Nihei, Y., Kudou, K., and Hinata, H., 2019. Assessment of the sources and inflow processes of microplastics in the river environments of Japan. *Environmental Pollution* 244: 958–65. <https://doi.org/10.1016/j.envpol.2018.10.111>.
- Kole, P.J., Löhr, A.J., Van Belleghem, F., and Ragas, A., 2017. Wear and tear of tyres: A stealthy source of microplastics in the environment. *International Journal of Environmental Research and Public Health* 14(10): 1265. <https://doi.org/10.3390/ijerph14101265>.
- Lechner, A., Keckeis, H., Lumesberger-Loisl, F., Zens, B., Krusch, R., Tritthart, M., Glas, M., and Schludermann, E., 2014. The Danube so colourful: A potpourri of plastic litter outnumbers fish larvae in Europe's second largest river. *Environmental Pollution* 188: 177–81. <https://doi.org/10.1016/j.envpol.2014.02.006>.
- Lin, L., Zuo, L-Z., Peng, J-P., Cai, L-Q., Fok, L., Yan, Y., Li, H-X., and Xu, X-R., 2018. Occurrence and distribution of microplastics in an urban river: A case study in the Pearl River along Guangzhou City, China. *Science of the Total Environment* 644: 375–81. <https://doi.org/10.1016/j.scitotenv.2018.06.327>.
- Mani, T., Hauk, A., Walter, U., and Burkhardt-Holm, P., 2016. Microplastics profile along the Rhine River. *Scientific Reports* 5(1). <https://doi.org/10.1038/srep17988>.
- McKee, L.J., Leatherbarrow, J., Pearce, S., and Davis, J. 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the Sources, Pathways and Loading Workgroup of the Regional Monitoring Program for Trace Substances. SFEI Contribution No. 66. San Francisco Estuary Institute, Oakland, Ca. <http://www.sfei.org/documents/review-urban-runoff-processes-bay-area-existing-knowledge-conceptual-models-and-monitoring>
- McKee, L.J., Lewicki, M., Schoellhamer, D.H., Ganju, N.K. 2013. Comparison of sediment supply to San Francisco Bay from watersheds draining the Bay Area and the Central Valley of California. *Marine Geology* 345: 47–62.

Chapter 2—Stormwater

Moore, C.J., G.L. Lattin, and Zellers, A.F., 2011. Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. *Revista de Gestão Costeira Integrada* 11 (1): 65–73. <https://doi.org/10.5894/rgci194>.

Moriasi, D.N., Wilson, B.N., Douglas-Mankin, K.R., Arnold, J.G., Gowda, P.H., 2012. Hydrologic and water quality models: Use, calibration, and validation. *Transactions of the ASABE* 55, 1241–1247. <https://doi.org/10.13031/2013.42265>.

Oram, J.J., McKee, L.J., Werme, C.E., Connor, M.S., Oros, D.R., Grace, R., Rodigari, F., 2008. A mass budget of polybrominated diphenyl ethers in San Francisco Bay, CA. *Environment International* 34 (2008) 1137-1147.

Schmidt, C., Krauth, T., and Wagner, S., 2017. Export of plastic debris by rivers into the sea. *Environmental Science and Technology* 51(21): 12246-12253.

Sedlak, M., Sutton, R., Box, C., Sun, J., Lin, D., 2017. FINAL Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries. SFEI Contribution No. 819. San Francisco Estuary Institute and 5 Gyres, Richmond, CA.

Sutton R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. *Marine Pollution Bulletin* 109: 230-235.

Sutton, R. and Sedlak, M., 2017. Microplastic Monitoring and Science Strategy. SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA..

Stolte, A., Forster, S., Gerdts, G., Schubert, H., 2015. Microplastic concentrations in beach sediments along the German Baltic coast. *Marine Pollution Bulletin* 99(1): 216-229.

Unice, K.M., Kreider, M.L., Panko, J.M., 2013. Comparison of tire and road wear particle concentrations in sediment for watersheds in France, Japan, and the United States by quantitative pyrolysis GC/MS analysis. *Environmental Science and Technology* 130710100101002. <https://doi.org/10.1021/es400871j>

Wang, W., Wairimu Ndungu, A., Li, Z., and Wang, J., 2017. Microplastics pollution in inland freshwaters of China: A case study in urban surface waters of Wuhan, China. *Science of the Total Environment* 575: 1369–74. <https://doi.org/10.1016/j.scitotenv.2016.09.213>.

Wu, J., Gilbreath, A., and McKee, L., 2017. Regional Watershed Spreadsheet Model (RWSM): Year 6 Progress Report. A Technical Report Prepared for the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP), Sources, Pathways and Loadings Workgroup

(SPLWG), Small Tributaries Loading Strategy (STLS). SFEI Contribution No. 811. San Francisco Estuary Institute, Richmond, CA. <http://www.sfei.org/documents/regional-watershed-spreadsheet-model-rwsm-year-6-final-report>.

Xiong, X., Wu, C., Elser, J.J., Mei, Z., and Hao, Y., 2019. Occurrence and fate of microplastic debris in middle and lower reaches of the Yangtze River – From inland to the sea. *Science of the Total Environment* 659: 66–73. <https://doi.org/10.1016/j.scitotenv.2018.12.313>.

Yan, M., Nie, H., Xu, K., He, Y., Hu, Y., Huang, Y., and Wang, J., 2019. Microplastic abundance, distribution and composition in the Pearl River along Guangzhou City and Pearl River Estuary, China. *Chemosphere* 217: 879–86. <https://doi.org/10.1016/j.chemosphere.2018.11.093>.

Yonkos, L.T., Friedel, E.A., Perez-Reyes, A.C., Ghosal, S., and Arthur, C.D., 2014. Microplastics in four estuarine rivers in the Chesapeake Bay, U.S.A. *Environmental Science & Technology* 48 (24): 14195–202. <https://doi.org/10.1021/es5036317>.

CHAPTER

3

Microparticles and Microplastics IN BAY AREA WASTEWATER : by Meg Sedlak



Highlights

- ◆ This study monitored microparticles and microplastics in effluent from eight wastewater treatment facilities that represent approximately 70% of the treated effluent flow discharged to San Francisco Bay. These facilities were geographically distributed, varied in effluent treatment capacity from 150 to 630 million liters per day, and employed a range of secondary and tertiary treatments. Microparticles captured on sequential 355 and 125 micron mesh sieves were manually counted using microscopy; approximately 30% of the microparticles were then analyzed using spectroscopy to determine whether they were microplastics.
- ◆ Microparticles were identified in effluent from all eight facilities, which discharged an average of 0.063 microparticles per liter. The concentrations of microparticles observed in this study are consistent with the range of values reported in the literature.
- ◆ Fibers, followed by fragments, were the most frequently identified shapes of microparticles, a common observation in the literature. Fibers that underwent Raman or Fourier Transform Infrared (FTIR) spectroscopy were primarily identified as anthropogenic unknown (approximately 50%). Anthropogenic unknown indicates a fiber that is dyed with a man-made chemical, but for which the underlying fiber composition cannot be identified (i.e., it may or may not be plastic). Approximately 19% of the fibers were identified as plastic, and 24% were identified as non-plastic.
- ◆ Fragments that underwent spectroscopy were primarily identified as plastic (54%), with most being polyethylene (31%). Several fragments could not be readily identified (20%).
- ◆ Facilities employing tertiary treatment, including dual media filtration had lower microparticle concentrations than secondary treatment facilities, suggesting that enhanced treatment may have multiple societal benefits, including reductions in microparticles as well as pollutants. However, it is likely far more cost-effective to prevent pollution in the first place (e.g., bans on plastic such as microbeads) or to mitigate it directly at the point of entry (e.g., providing filters for washing machines).
- ◆ In aggregate, approximately 91 million microparticles per day were discharged by the eight facilities. Assuming other Bay Area facilities are similar, we estimate approximately 129 million microparticles are discharged per day, or approximately 47 billion microparticles annually to the Bay from wastewater treatment facilities. This is substantially lower than the estimate developed for the annual microparticle loads from the small tributaries surrounding the Bay (10.9 trillion microparticles to the Bay per year; Chapter 2 Stormwater).
- ◆ Not all microparticles are plastic. Of the 91 million microparticles discharged per day by the eight facilities, based on Raman/FTIR spectroscopy for a subset of microparticles

and information in the literature, we estimate the range of microplastics discharged to the Bay by the eight facilities to be 29–45 million microplastics per day, with a plausible estimate of 32 million microplastics per day. This would translate to 46 million microplastics per day (or 17 billion microplastics per year) for the Bay Area municipal wastewater pathway as a whole.

Objectives

The purpose of this element of the San Francisco Bay Microplastics Project was to characterize the morphology and concentrations of microparticles and microplastics in municipal wastewater effluent. Wastewater effluent is a potentially important pathway of microplastics to the environment (European Chemicals Agency, 2019). Microplastics may be introduced into wastewater through processes including the washing of textiles; the fragmentation of plastic pieces used in homes and industry; and the discharge of microbeads from personal care products (e.g., facial scrubs and toothpastes), controlled-release medications, and industrial processes (e.g., abrasive blasting and painting).

In this report, we have distinguished between microparticles, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and microplastics, which have been confirmed to be plastic through Raman or FTIR spectroscopy.

Microparticle and microplastic morphologies and concentrations were characterized in final effluent samples collected from eight wastewater treatment facilities that discharge to San Francisco Bay.

Through this assessment, we sought to achieve the following objectives.

1. **Quantify the abundance of microparticles and microplastics in effluent.** Understanding the abundance of microparticles and microplastics in effluent is important for evaluating the sources, pathways, loadings, and processes [Management Question (MQ) 3] leading to microplastics in the Bay [(MQ1); see the Microplastic Monitoring and Science Strategy report (Sutton and Sedlak, 2017) for further detail on the MQs].
2. **Characterize types of microparticles and microplastics found in effluent and their chemical composition.** Understanding the types of microparticles and microplastics found in effluent will help determine the sources of microplastics to wastewater. This could help inform future decisions about management measures (MQ5) that could contribute to possible future reductions of microplastics transported to the Bay via effluent.
3. **Evaluate whether treatment type affects the abundance of microparticles released.** The study design included four secondary treatment facilities and four tertiary treatment facilities to allow evaluation of the effect of treatment on microparticle abundance. All tertiary facilities employed additional filtration at the end of the treatment train. Our previous screening study monitoring Bay Area secondary and tertiary facilities found no statistically significant difference in effluent microparticle concentrations based on treatment (Sutton et al., 2016). Indications of treatment efficacy for microparticle and

microplastic removal may inform future management decisions relevant to wastewater agencies (MQ5).

4. **Evaluate effluent microparticle concentrations in the San Francisco Bay Area relative to other effluent studies.** The results from this study will be placed in context with the literature. The prior screening study suggested that average effluent concentrations of microparticles in Bay Area wastewater were higher than those observed in other parts of the U.S., though few studies of wastewater were available at that time (Mason et al., 2016).
5. **Refine sample collection and analysis methods for effluent samples.** A key step in quantifying the abundance of microparticles and microplastics (MQ1) is establishing appropriate field and laboratory methods for measurement. We refined existing methods for this study and examined their performance using standard quality assurance measures.

As presented in the Sampling and Analysis Plan (Sedlak et al., 2017), we hypothesized:

- ◆ Concentrations of microplastics in wastewater effluent will be independent of treatment trains (i.e., secondary vs. tertiary treatment).
- ◆ Concentrations of microplastics in stormwater and wastewater will be comparable; however, the composition of the microplastics will be different. (See Chapter 8 Synthesis for further exploration of this hypothesis.)

This study builds upon a prior screening study of San Francisco Bay Area effluent (Sutton et al., 2016) by sampling facilities over 24 hours to obtain more representative samples, and by conducting repeat sampling to assess variation. In addition, spectroscopy was conducted on a subset of particles to identify the chemical composition, in particular to confirm whether the particles were plastic.

Methods

Wastewater facility selection

Eight facilities voluntarily participated in the wastewater study; site selection was coordinated with other study elements such as stormwater and sediment (Table 3.1; Figure 3.1). These facilities were geographically distributed around the Bay, varied in effluent treatment capacity from approximately 90 to 630 million liters per day (24 to 167 MGD; San Francisco Regional Water Quality Control Board - Region 2, 2019), and employed a range of secondary and tertiary treatments (Table 3.1). These eight facilities represent approximately 70% of the total flow of effluent to San Francisco Bay (San Francisco Bay Regional Water Quality Control Board - Region 2, 2019). Secondary treatment included biological treatment followed by sedimentation. Tertiary treatment included a variety of additional treatments, but all facilities used dual media filtration (sand and anthracite) as a finishing step.

Sample collection

Effluent from eight wastewater treatment facilities was collected in the summer and fall of 2017 (Table 3.2). A standard collection method was used for all eight facilities; however, minor modifications were necessary to accommodate individual facilities or issues that occurred during sample collection (e.g., clogging of the sieves). The method is described below and modifications are noted in Table 3.2. With the one exception of a set of samples collected during the wet season, we believe the variations in collection are minor and unlikely to affect the results.

Each facility was sampled twice; samples were collected Tuesday through Friday to avoid the potential influence of different consumer behaviors over the weekend. Based on a Bay Area Clean Water Agencies study of the EBMUD facility (Dyachenko et al., 2017), effluent was collected over a 24-hour period to obtain a more representative sample relative to the previous screening study (Sutton et al., 2016), which sampled over two hours during peak flow. For two events, samples were collected over a modified time period (Table 3.2). All particle counts were normalized for the total volume passed through the sampling sieves to calculate the concentration of microparticles per liter (microparticles/L) for comparative purposes.

Table 3.1. Wastewater treatment facilities sampled.

Location	Site¹	Treatment	Design Flow (MGD)	Sampling Dates	Adjacent Bay Sediment Site
North Bay	Central Contra Costa Sanitary District (CCCSD)	Secondary	~50	9/7/17; 12/6/17	SUB53
North Bay	Fairfield-Suisun Sewer District (FSSD)	Tertiary	23.7	8/23/17; 9/7/17	
Central Bay	San Francisco Public Utilities Commission, Southeast Plant (SFPUC)	Secondary	86	11/6/17; 11/7/17	
Central Bay	East Bay Municipal Utility District (EBMUD)	Secondary	~120	8/21/17; 9/26/17; 10/20/17	
South Bay	East Bay Dischargers Authority (EBDA) ²	Secondary	108	8/31/17; 9/26/17	
Lower South Bay	Sunnyvale Water Pollution Control Plant	Tertiary	29.5	9/19/17; 10/17/17	SOSL16
Lower South Bay	Palo Alto Regional Water Quality Control Plant	Tertiary	39	7/20/17; 8/1/17	
Lower South Bay	San Jose-Santa Clara Regional Wastewater Facility	Tertiary	167	8/10/17; 9/19/17	SOSL40

¹ Field blanks were collected at the SFPUC treatment plant. Field duplicates were collected at the Palo Alto treatment plant.

² EBDA represents the aggregated effluent from six wastewater treatment plants that is discharged to the Bay (i.e., San Leandro, Oro Loma, Hayward, Union, Dublin San Ramon, and Livermore).

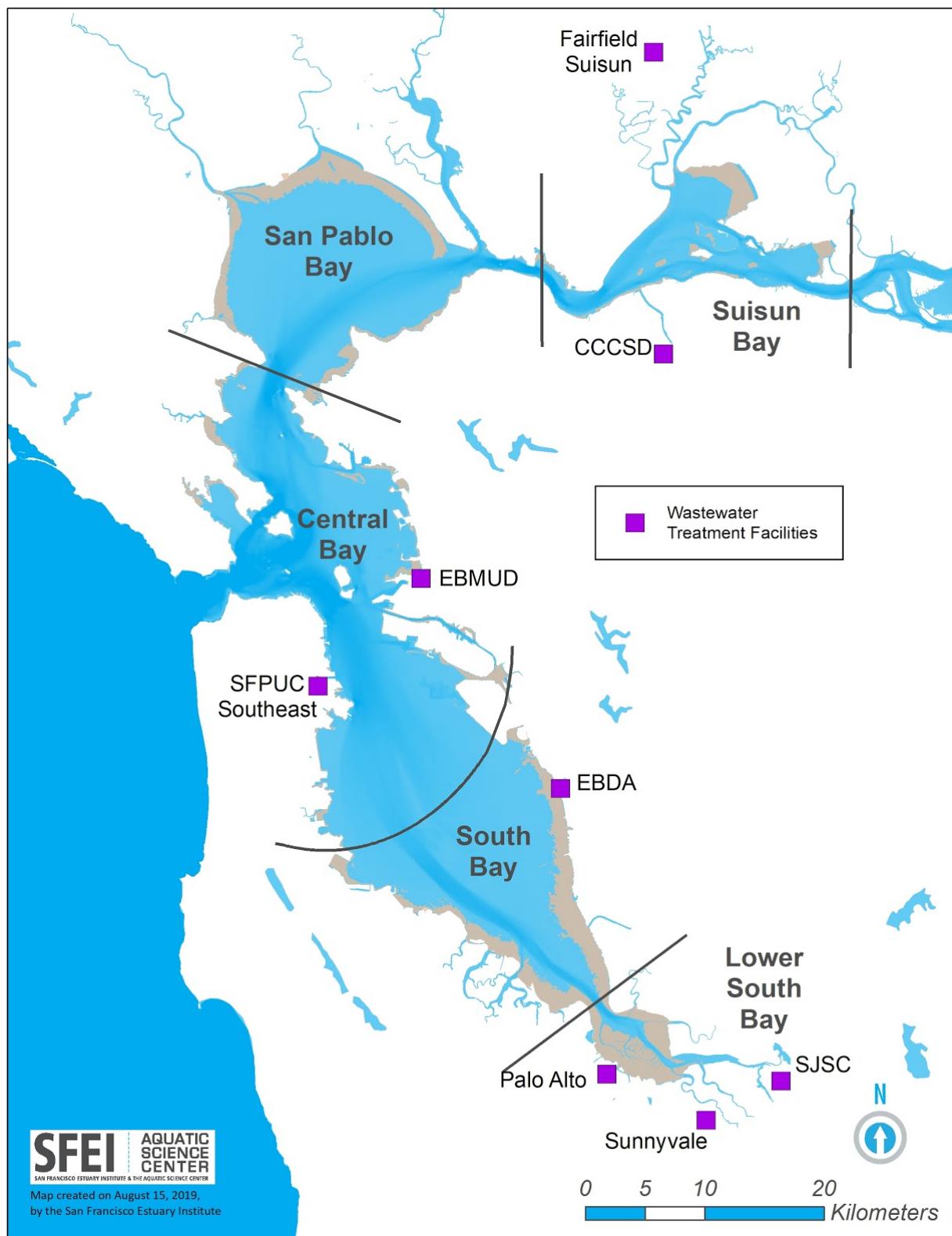


Figure 3.1. Location of wastewater treatment facilities.

Dry weather sampling was targeted to reduce the potential variation that might be associated with wet weather infiltration and inflow; however, due to logistical challenges, it was not possible to collect all samples in the dry season (Table 3.2). Two samples from SFPUC, the only combined sewer operation in the study, were collected approximately one month after a small rainfall event (0.76 cm) and one CCCSD sample was collected one week after a similarly-sized event in December. A heavy rainfall event (4.0 cm), considered to be a “first flush,” occurred in late November in the Bay Area; this occurred after sampling SFPUC and prior to collecting the second CCCSD sample.

Based on procedures employed in other wastewater studies (Dyachenko et al., 2017; Mason et al., 2016; Sutton et al., 2016; Talvitie et al., 2017), effluent was passed through 20.3 cm (standard 8-inch) diameter, stacked Tyler sieves with 355 µm and 125 µm stainless steel mesh. During the sampling period, the sieves were placed underneath an upside down bucket to prevent the deposition of airborne particles (Figure 3.2). The base of the bucket was fitted with a pipe to convey the effluent from the sampling port to the sieves.

At most facilities, the sample volume was measured using a Recordall® Disc Meter; effluent from the sampling port was piped to the meter, where it was then discharged through the stacked sieves (Figure 3.2; Table 3.2). The total volume of wastewater effluent that flowed through the system was shown on the dial, and the meter reading at the beginning and end of the sampling period was recorded. At three facilities, the measurement of effluent volume was modified because it was not feasible to use the flow meter (Table 3.2). The range of flow rates varied between 1.1–8.7 liters per minute, with higher flow rates used at sites known to have lower suspended solids in the final effluent.

The sample flow rate was adjusted at each facility to avoid clogging of the sieves. The total volume of effluent subsampled over the 24-hour period from the facilities ranged from approximately 900 to 12,300 liters (Table 3.2; average volume 4,900 L). At one site, one sample event was shorter (12 hours) due to clogging of the sieves, resulting in a smaller volume of effluent (916 L) passing through the sieve set.

A field blank was collected at the SFPUC site by placing a secondary sieve set adjacent to the primary field sieve set. The field blank sieve set was placed under an upside down bucket with no perforation; the adjacent field sample sieve set was connected to the effluent sampling port via a pipe. Both sieve sets remained in place for the duration of the 24-hour sampling event. The field blank was processed with the field samples. A field duplicate was collected at the Palo Alto site by using a Y-splitter on the sampling port to divide the effluent into two streams to flow through two sieve sets (Figure 3.2).

After collection, the sieve sets were covered in foil, placed in a cooler, and transported to SFEI to be processed in the laboratory. Microparticles were gently rinsed off the sieves into pre-cleaned glass sample jars using distilled water prior to shipping to the University of Toronto for analysis.



Figure 3.2. Wastewater effluent sample collection apparatus; duplicate sampling event showing parallel collection trains. Samples were collected by connecting a flow meter to the effluent sampling valve. Effluent was directed into a sieve set that was placed under an upside-down bucket. Sampling train designed by City of Palo Alto employees.

Table 3.2. Microparticles in treated wastewater and estimates of discharge per liter and per day.

Wastewater Treatment Plant	Level of Treatment	Ave TSS (mg/L)	Flow (MLD) ^a	Sample Event ⁱ	Size Category (mm)	No. Microparticles by Type					Sampled Volume (L) ^h	Per Liter	No. Microparticles Total No. per Day
						Fragment	Sphere	Fiber	Film	Foam ^j			
Central Contra Costa Sanitation District ^{b,c}	Secondary	7.5	119	A	125	19	7	43	0	10	79	3244	0.024
					355	77	18	85	16	24	220	3244	0.068
					Total	96	25	128	16	34	299	3244	0.092
					125	1	0	33	0	0	34	916	0.037
	Secondary	20	129	B ^b	355	3	0	26	3	1	33	916	0.036
					Total	4*	0	59	3	1	67	916	0.073
					125	90	9	232	1	11	343	2859	0.12
					Total	127	12	285	23	56	503	2859	0.18
SFPUC Southeast Treatment Plant ^{c,e}	Secondary	20	216	A	355	37	3	53	22	45	160	2859	0.056
					Total	127	12	285	23	56	503	2859	0.18
					125	43	4	34	7	14	102	2263	0.045
					Total	162	12	192	57	31	454	2263	0.2
	Secondary	13	227	B	355	119	8	158	50	17	352	2263	0.16
					Total	162	12	192	57	31	454	2263	0.2
					125	49	5	18	0	229	301	3483	0.086
					Total	122	5	111	6	240	484	3483	0.14
East Bay Municipal Utilities District (EBMUD) ^{d,e,f}	Secondary	13	174	A	355	73	0	93	6	11	183	3483	0.053
					Total	122	5	111	6	240	484	3483	0.14
					125	37	2	79	3	27	148	3405	0.043
					Total	45	3	159	6	38	251	3405	0.074
	Secondary	9	182	B	355	8	1	80	3	11	103	3405	0.03
					Total	45	3	159	6	38	251	3405	0.074
					125	9	3	63	0	0	75	3914	0.019
					Total	16	4	284	1	0	305	3914	0.078
East Bay Dischargers Association (EBDA) ^e	Secondary	9	194	A	355	7	1	221	1	0	230	3914	0.059
					Total	16	4	284	1	0	305	3914	0.078
					125	9	3	63	0	0	75	3914	0.019

Wastewater Treatment Plant	Level of Treatment	Ave TSS (mg/L)	Flow (MLD) ^a	Sample Event ⁱ	Size Category (mm)	No. Microparticles by Type						Sampled Volume (L) ^h	Per Liter	No. Microparticles Total No. per Day
						Fragment	Sphere	Fiber	Film	Foam ^j	Total			
East Bay Dischargers Association (EBDA) ^e	Secondary	9	169	B	125	8	0	43	0	5	56	7885	0.0071	
					355	20	4	41	1	18	84	7885	0.011	
					Total	28	4	84	1	23	140	7885	0.018	3,045,411
					125	0	0	33	0	0	33	5954	0.0055	
					355	2	0	14	2	0	18	5954	0.003	
	Tertiary (Dual media filtration)	<0.6	39	A	Total	2*	0	47	2	0	51	5954	0.0086	332,020
					125	16	0	12	4	5	37	8150	0.0045	
					355	7	0	33	1	0	41	8150	0.005	
					Total	23	0	45	5	5	78	8150	0.0096	424,768
					125	7	0	13	0	0	20	1440	0.014	
Fairfield-Suisun	Tertiary (Dual media filtration)	<0.6	44	B	355	0	0	38	0	0	38	1440	0.026	
					Total	7*	0	51	0	0	58	1440	0.04	985,614
					125	6	0	15	0	1	22	2880	0.0076	
					355	5	0	35	0	0	40	2880	0.014	
					Total	11*	0	50	0	1	62	2880	0.022	1,090,837
	Tertiary (Dual media filtration)	11	32	A	125	3	0	114	0	0	117	9887	0.012	
					355	1	0	3	3	0	7	9887	0.0007	
					Total	4*	0	117	3	0	124	9887	0.013	954,200
					125	7	0	33	0	0	40	9887	0.004	
					355	1	0	38	0	0	39	9887	0.0039	
Palo Alto	Tertiary (Dual media filtration)	<0.6	73	A	Total	8*	0	71	0	0	79	9887	0.008	526,400
					125	7	0	33	0	0	40	9887	0.004	
					355	1	0	38	0	0	39	9887	0.0039	
					Total	8*	0	71	0	0	79	9887	0.008	526,400

Wastewater Treatment Plant	Level of Treatment	Ave TSS (mg/L)	Flow (MLD) ^a	Sample Event ^b	Size Category (mm)	No. Microparticles by Type						Sampled Volume (L) ^c	Per Liter	No. Microparticles Total No. per Day
						Fragment	Sphere	Fiber	Film	Foam ^d	Total			
Palo Alto	Tertiary (Dual media filtration)	<0.6	73	Field dup	125	19	0	54	2	1	76	12313	0.0062	-
					355	11	0	17	1	0	29	12313	0.0024	-
					Total	30	0	71	3	1	105	12313	0.0085	-
					125	49	40	20	13	6	128	7586	0.017	-
San José-Santa Clara	Tertiary (Dual media filtration)	1.3	303	A	355	7	2	15	3	15	42	7586	0.0055	-
					Total	56	42	35	16	21	170	7586	0.022	6,672,425
					125	29	2	36	1	5	73	4868	0.015	-
			307	B	355	14	2	53	4	10	83	4868	0.017	-
					Total	43	4	89	5	15	156	4868	0.032	9,833,733
					Count Total	784	111	1878	147	466	3386			90,709,152
Percent Total^e						23	3	55	4	14	100			Total particles/day

^a Flow reported on day of sample collection.^b Sample Event B at CCCSD was 12 hours instead of 24.^c Samples from SFPUC were collected one month after a rainfall event of 0.29 inch (10/19/2017); Sample B from CCCSD was collected approximately one week after a 0.29 inch rainfall event (11/27/2017).^d Due to clogging of the sieve, Sample B was split into two collection events - the first event was for 21 hours; the second event was for 3 hours over the last 24-hr period. The results are combined and reported.^e These samples were collected prior to de-chlorination.^f Due to clogging of the sieve, a 1-mm sieve was added to the 355 and 125 micron sieve stack. The 1-mm material was included in the 355 count.^g Percentage Total calculation does not include Field duplicate from Palo Alto.^h Flow was measured using a flow meter except at Sunnyvale, where an ISCO sampler was used and flow was calculated using a container of known volume and a stopwatch.ⁱ A indicates first sample collected at site; B indicates second sample collected at the site.^j Foam count of Sample A CCCSD from the 125 fraction is an undercount (noted as 10, however, over 100 more were identified but not quantified by the lab).^{*} The fragment counts for CCCSD (4), FFSS (2) Sunnyvale (7 and 11) and Palo Alto (4 and 8) should be treated with caution as they are less than the lab/field blank average plus 2 standard deviation of blank (11.9 fragments threshold).

Sample extraction and analysis

In the analytical laboratory, wastewater samples were dewatered and then processed using a digestion step to remove organic material. The samples were passed through a 110 µm sieve to remove water. They were then reconstituted in a pre-filtered 20% potassium hydroxide (KOH) solution, as recommended by Munno et al. (2018), at room temperature for a one-week period to remove organic matter. Potassium hydroxide is the preferred base for digestion (Dehaut et al., 2016; Lusher et al., 2017). At the end of the one-week period, samples were sieved through the 110 µm sieve and rinsed with reverse-osmosis-treated (RO) water. Samples were then transferred into a clean glass jar for microplastic analyses.

Once extracted, particles were visually identified as potentially plastic (microparticles) and classified by morphology and color using a dissecting microscope. For a subset of microparticles, photographs were taken and the length and width of each particle measured. For each morphology and color, the first ten microparticles identified were analyzed by Raman/FTIR spectroscopy to determine the chemical composition of the particle using a reference spectral library. In total, Raman/FTIR spectroscopy was conducted on approximately 40% of the visually identified microparticles.

To evaluate the potential of background contamination, one laboratory blank was run for every ten samples processed; in total, five laboratory blanks were run. All laboratory blanks were composed of RO water processed using the same methods as the field samples. To prevent contamination in the laboratory, laboratory surfaces were wiped down daily, all glassware and metal tools were rinsed with RO water three times between each sample, and researchers wore cotton laboratory coats and worked in a clean cabinet when possible.

Analytical method recovery evaluation

Prior to commencing analyses of field samples, a laboratory study was conducted to assess the efficacy of the extraction methods. Matrix spikes were prepared using a synthetic effluent consisting of algal material that was blended down to a fine size to create an organic model matrix. This synthetic effluent was spiked with particles of a range of sizes, morphologies, and polymers. The spikes consisted of known numbers of polyethylene terephthalate fragments (clear/white), polystyrene beads (white), cellulose acetate beads (red), polyethylene beads (green) and polypropylene fibers (blue), ranging in particle size from 250 µm to 3 mm.

Statistical analysis and treatment of blanks

Statistical analyses were conducted in R (R Core 2018) using the Wilcoxon rank-sum test and two-tailed distributions.



Laboratory and field blank results are reported alongside field sample results. Field samples were not blank corrected (i.e., blank counts were not subtracted from the field sample counts) due to the non-uniform nature of the background field and laboratory contamination observed. The field and laboratory blanks were used to develop threshold values for data qualification using standard data validation practices to help evaluate the results as discussed below. The field and laboratory blank data, as well as the threshold values, are reported so individual readers can make their own assessment.

Field duplicate results are reported in Table 3.2. Statistical analyses to assess the influence of treatment were conducted using the primary field sample and did not include the field duplicate to avoid biasing the analyses.

Results

Quality assurance results

ANALYTICAL METHOD RECOVERY EVALUATION

Analysis of the three replicate spiked matrix samples indicated quantitative recovery (i.e., approximately 100%) for polyethylene terephthalate, polystyrene, and cellulose acetate particles (Table 3.3). Polyethylene beads and polypropylene fibers had slightly lower recoveries, on average 76% and 90%, respectively. Results from spiked matrix samples indicated the possibility for sample extraction procedures to result in fragmentation of certain particles, which could lead to recoveries greater than 100% if each fragment is enumerated. Specifically, one of the 30 spiked polyethylene terephthalate particles fragmented. In addition, all nine cellulose acetate particles had visible cracks, suggesting the potential for fragmentation.

Table 3.3. Recovery of spiked microplastics in model wastewater effluent (matrix) sample. Seaweed was blended down to a fine size to create the model matrix.

Particle and Plastic Type	Particle Size	Replicate 1 Recovery	Replicate 2 Recovery	Replicate 3 Recovery	Notes
Polyethylene terephthalate fragment (clear/white)	1 mm	10 (100%)	10 (100%)	10 (100%)	A particle in Replicate 3 broke into pieces, but was counted as a single fragment.
Polystyrene bead (white)	1 mm	10 (100%)	10 (100%)	10 (100%)	
Cellulose acetate bead (red)	1 mm	3 (100%)	3 (100%)	3 (100%)	Cracks evident.
Polyethylene bead (green)	250–300 µm	7 (70%)	8 (80%)	8 (80%)	
Polypropylene fiber (blue)	3 mm in length	9 (90%)	9 (90%)	0 (0%)	Replicate 3 was likely not spiked with fibers due to lab error.

BACKGROUND CONTAMINATION: FIELD AND LABORATORY BLANKS

Microparticles were detected in the laboratory and field blanks at low levels (Figure 3.3). Based on the Raman or FTIR spectroscopy conducted on 86% of the blank particles, 26% were identified as plastic, 40% as anthropogenic unknown (may or may not be plastic), and 27% as non-plastic material (e.g., cotton, anthropogenic cellulosic, or natural materials). The remaining particles could not be identified (i.e., could not be matched to spectra in the library).

Fibers were detected in all blanks (ranging from 4 to 19 fibers), with the highest total count observed in the field blank (14 in the 125 µm fraction and 17 in the 355 µm fraction, for a total of 31; Figure 3.3). The lengths of the fibers appeared normally distributed in both lab and field blanks, with approximately 30% of the fibers being 1–2 mm in length. Clear and white fibers were most frequently observed in the laboratory blanks (range = 1 to 11). For approximately 50% of these clear and white fibers, Raman spectroscopy indicated they were predominantly cotton, anthropogenic cellulosic, and polyester fibers; another 40% of these fibers could not readily be identified. Based on discussions with the analytical laboratory, the source of these clear and white fibers is not known and likely quite diverse (e.g., cotton lab coats, Kimwipes, paper). The field blank fractions contained fewer white/clear fibers (3 and 4, respectively) and more blue and dark blue fibers (7 and 9 fibers, respectively); more than half of these were identified as anthropogenic unknown. The source of these fibers is not known.

Other particle types were either detected infrequently at low levels or not detected. Foam and sphere particles were not detected in any of the blanks (Figure 3.3). Films were detected in one laboratory blank (5 pieces). Fragments were detected in several laboratory and field blanks, ranging from 2 to 11 particles.

Fibers and fragments were detected predominantly in the field samples. To assess whether blank contamination of the field samples was significant, a threshold for data qualification of the average of the laboratory and field blanks plus two times the standard deviation was calculated for each particle morphology. All field fiber concentrations were above the threshold of 32.6 per sample; for fragments, several samples were below the fragment threshold of 11.9 per sample and are discussed below (Table 3.2). Thresholds were not developed for foam, sphere, or film morphologies because they were either not detected or detected in only one blank.

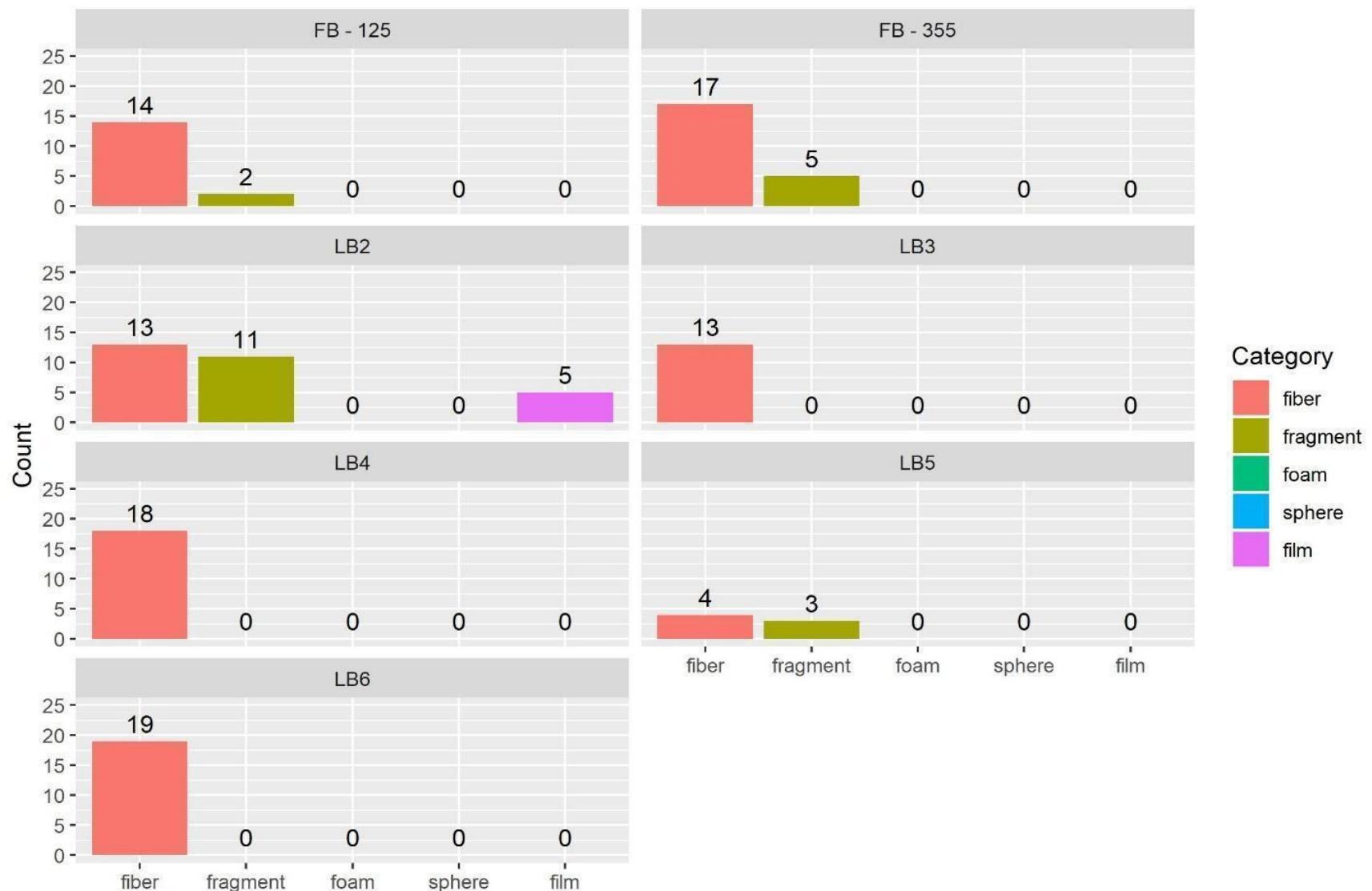


Figure 3.3. Microparticles detected in field blanks (FB) and laboratory blanks (LB). The 125 and 355 notation indicate the screen size (μm) used to sieve the field sample.

PRECISION AND VARIABILITY: FIELD DUPLICATES

The total microparticle counts in the duplicate field samples were 124 and 105 (relative percent difference (RPD) 16.6%), which suggests good precision and reproducibility in the method despite the heterogeneous nature of effluent. Inspection of the morphology of the particles in the paired field-duplicate samples indicated that almost all of the variation was due to counts of fibers (117 vs. 71) and fragments (4 vs. 30). Given the structure of fibers (long and narrow), the orientation of the fiber is expected to appreciably affect whether it is captured by sieves.

Occurrence of microparticles in effluent and variation with treatment

The microparticle concentrations (counts normalized by effluent volume) ranged from 0.008 microparticles/L (Palo Alto) to 0.2 microparticles/L (SFPUC) (Table 3.2). The eight wastewater treatment facilities discharged an average of 0.063 microparticles/L. The lowest count per liter was observed at a facility with tertiary treatment (Palo Alto); the highest count was observed at a facility with secondary treatment that receives combined inflows from stormwater and sanitary sewers (SFPUC). Overall, there was good agreement between the particle counts for the two sampling events conducted at each facility (Figure 3.4).

Treatment level (i.e., secondary or tertiary) was significantly related to microparticle concentration (Figure 3.4). The microparticle concentrations from tertiary treatment facilities were statistically significantly lower than those from secondary treatment facilities (Wilcoxon rank sum test $W = 4$, $p < 0.001$, two-tailed). The average total suspended solids (TSS) concentrations from these tertiary treatment facilities was approximately half that of the secondary facilities (5.8 vs. 12.2 mg/L), which may help to explain the lower microparticle concentrations. In addition, effluent from the tertiary treatment plants tended to show less variation in sample counts than effluent from secondary treatment (average RPD of 24% vs. 69%, respectively).

Treatment appeared to affect the number of fibers observed. Mean concentrations in the samples from secondary ($n = 8$) and tertiary ($n = 8$ facilities and 1 field duplicate) treatment groups were 0.056 and 0.013 fibers/L, respectively; the distributions of the two groups were significantly different (Wilcoxon rank-sum test $W = 5$, $p = 0.0016$, two-tailed). The common language effect size was 93%, which means that out of all possible comparisons between any secondary and tertiary sample in the study, the secondary treatment sample had a higher value 93% (67 of 72) of the time.

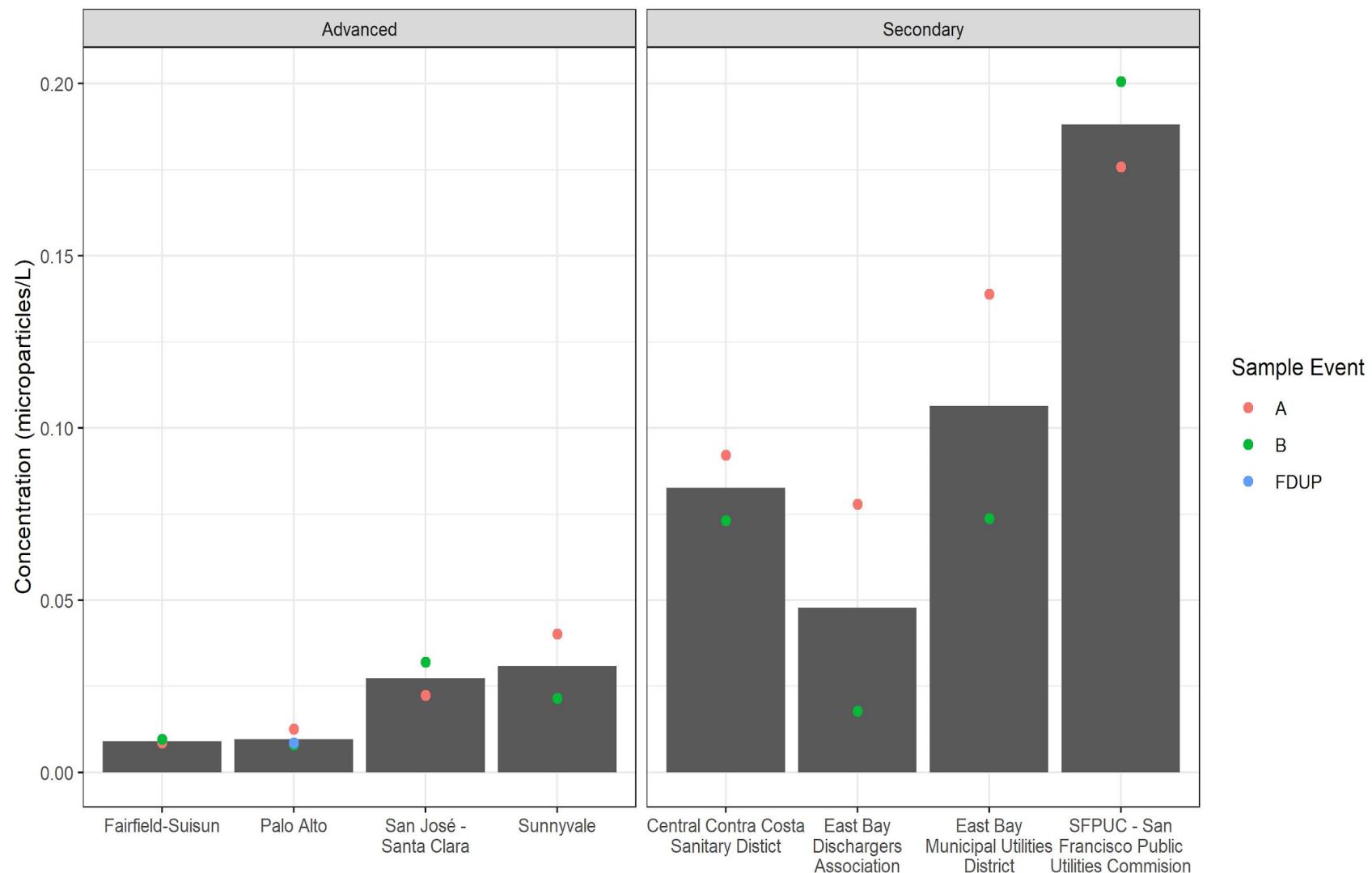


Figure 3.4. Microparticle concentrations by facility, with tertiary facilities on the left and secondary facilities on the right. Fragment counts for the following facilities were within the range of the average of field and laboratory blank counts plus 2 times the standard deviation: CCCSD (1 sample); FFSD (1 sample), Sunnyvale (both samples), and Palo Alto (both samples) and should be treated with caution. Legend indicates the results from the two different sample events (A and B) as well as the result of the field duplicate (FDUP) collected at the Palo Alto site.

Morphology and composition of microparticles

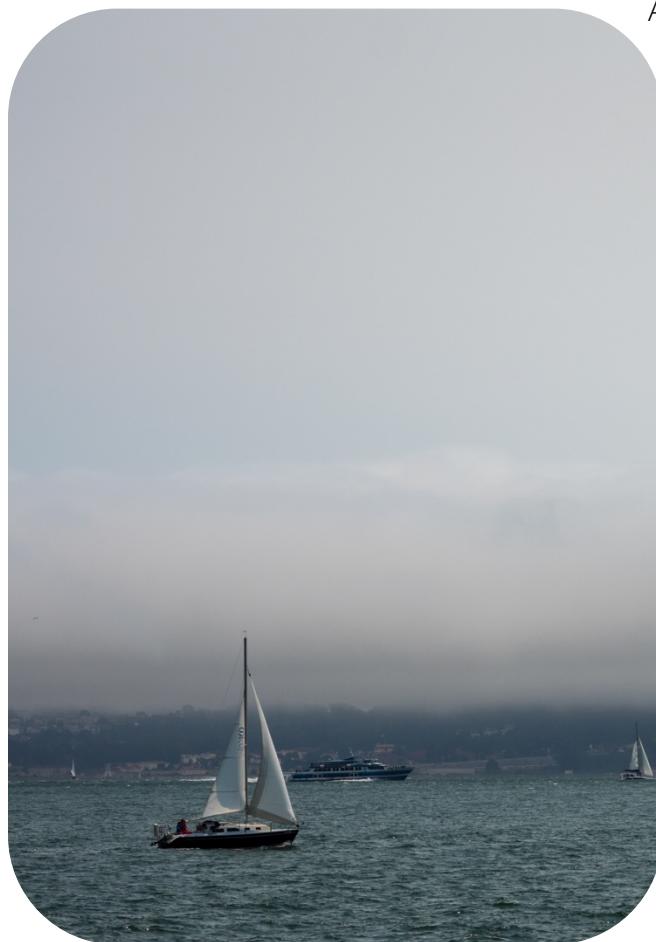
Overall, fibers were the major type of particle observed in effluent, accounting for an average of 55% of all particles observed across all eight plants, followed by fragments (23%), foams (14%), films (4%), and spheres (3%). Although fibers were a significant fraction identified at all facilities (Figure 3.5), the relative distributions of other morphologies varied by facility and are likely a reflection of treatment processes, the composition of influent to the facility, and other factors.



Figure 3.5. Average microparticle concentrations by morphology at individual treatment facilities. Fragment counts for the following facilities were within the range of the average of field and laboratory blanks plus 2 times the standard deviation: CCCSD (1 sample); FFSD (1 sample), Sunnyvale (both samples), and Palo Alto (both samples) and should be treated with caution. SFPUC: San Francisco Public Utilities Commission; EBMUD: East Bay Municipal Utility District; EBDA: East Bay Dischargers Authority; CCCSD: Central Contra Costa Sanitary District.

FIBER CONCENTRATION AND COMPOSITION

Fibers were the most frequently identified morphology in effluent (Figure 3.5). Fiber concentrations for effluent from tertiary treatment facilities ranged from 0.006 to 0.04 fibers/L and were statistically lower than fiber concentrations in effluent from secondary treatment facilities, which ranged from 0.01 to 0.1 fibers/L.



Approximately 33% of the fibers were analyzed by Raman/FTIR spectroscopy, and roughly 50% (317 of 628; Figure 3.6) were identified as anthropogenic unknown. Anthropogenic unknown indicates a fiber that is dyed with a coloring agent, but for which the underlying fiber composition cannot be identified (i.e., it may or may not be plastic). This occurs because the color or spectrum of the dye interferes with obtaining an identifiable spectrum from the underlying material. The second most frequently detected fiber type was anthropogenic cellulosic (80 fibers; 13%), which are manufactured, dyed fibers with spectra that indicate they are derived from natural cellulosic materials (such as cotton, rayon, modal, or Lyocell), though more specific material identification in these cases was not possible; these particles are not plastic. The remaining fiber types included cotton (57 fibers), polyester (41),

unknown (41), and a myriad of other compositions, representing less than 15% of the overall count. Overall, approximately 19% of the fibers that underwent spectroscopy were identified as plastic. The lengths of the fibers appeared to show a normal distribution, with the most frequently reported length in the 1 mm to 2 mm range (Figure 3.7).

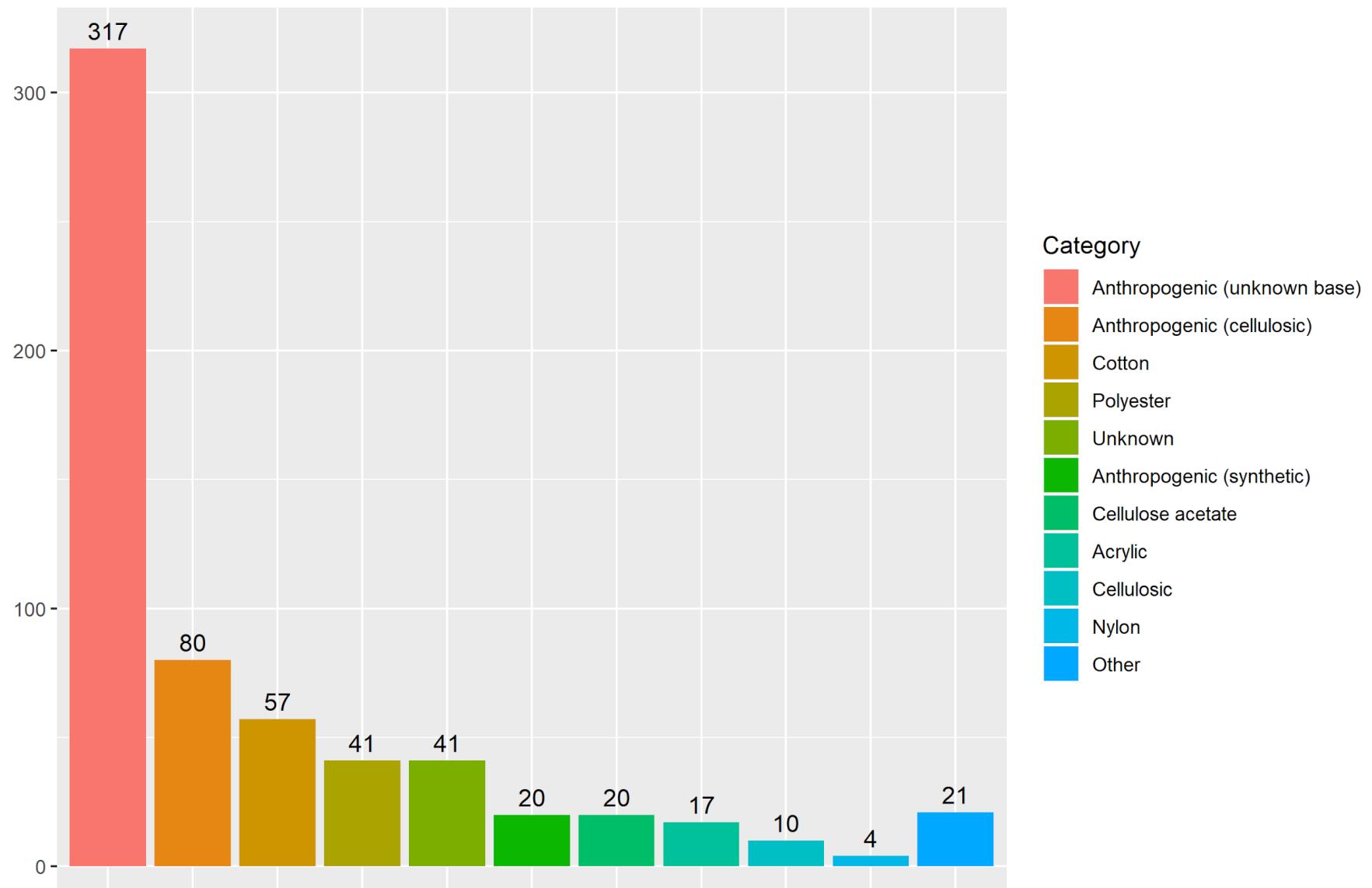


Figure 3.6. Count and composition of fibers analyzed by spectroscopy. Polyester, cellulose acetate, anthropogenic (synthetic), and acrylic are considered to be plastic

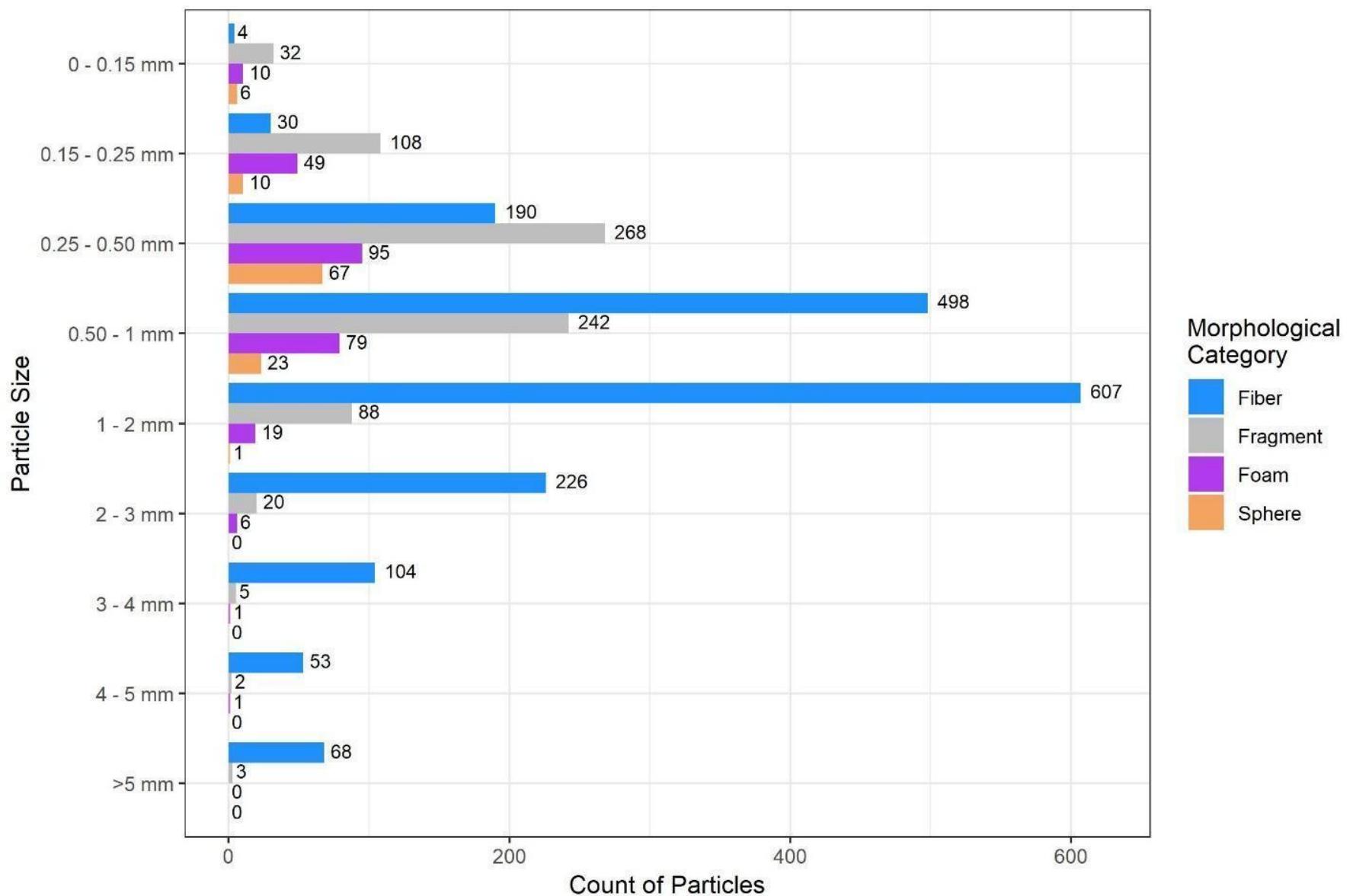


Figure 3.7. Distribution of particle size by morphological category. Note: size bins are not evenly distributed.

FRAGMENT CONCENTRATIONS AND COMPOSITION

After fibers, fragments were the next most frequently detected shape (Figure 3.5). Normalized to flow, the concentrations followed a similar trend with SFPUC having the highest concentrations (0.04 and 0.07 microparticles/L). SFPUC is the only facility that receives combined flows from sanitary and stormwater sewers, which may account for the relatively high level of fragments observed (see Chapter 2 Stormwater for further details on stormwater composition). Most fragments ranged in length between 250 µm and 1000 µm (Figure 3.7).

Approximately 40% of the fragments (312 of 784) were analyzed using spectroscopy; the majority were identified as polyethylene (96 fragments, 31%), followed by fragments of unknown composition (61 fragments), and anthropogenic unknown (47 fragments; Figure 3.8). Overall, 54% of the fragments that underwent spectroscopy were identified as plastic.

At Sunnyvale (both samples), Palo Alto (both samples), FSSD (one sample), and CCCSD (one sample), the fragment counts (total, not normalized by effluent volume) were less than the average of the field and laboratory blanks plus two times the standard deviation (a threshold of 11.9), so these results should be treated with caution (Table 3.2).

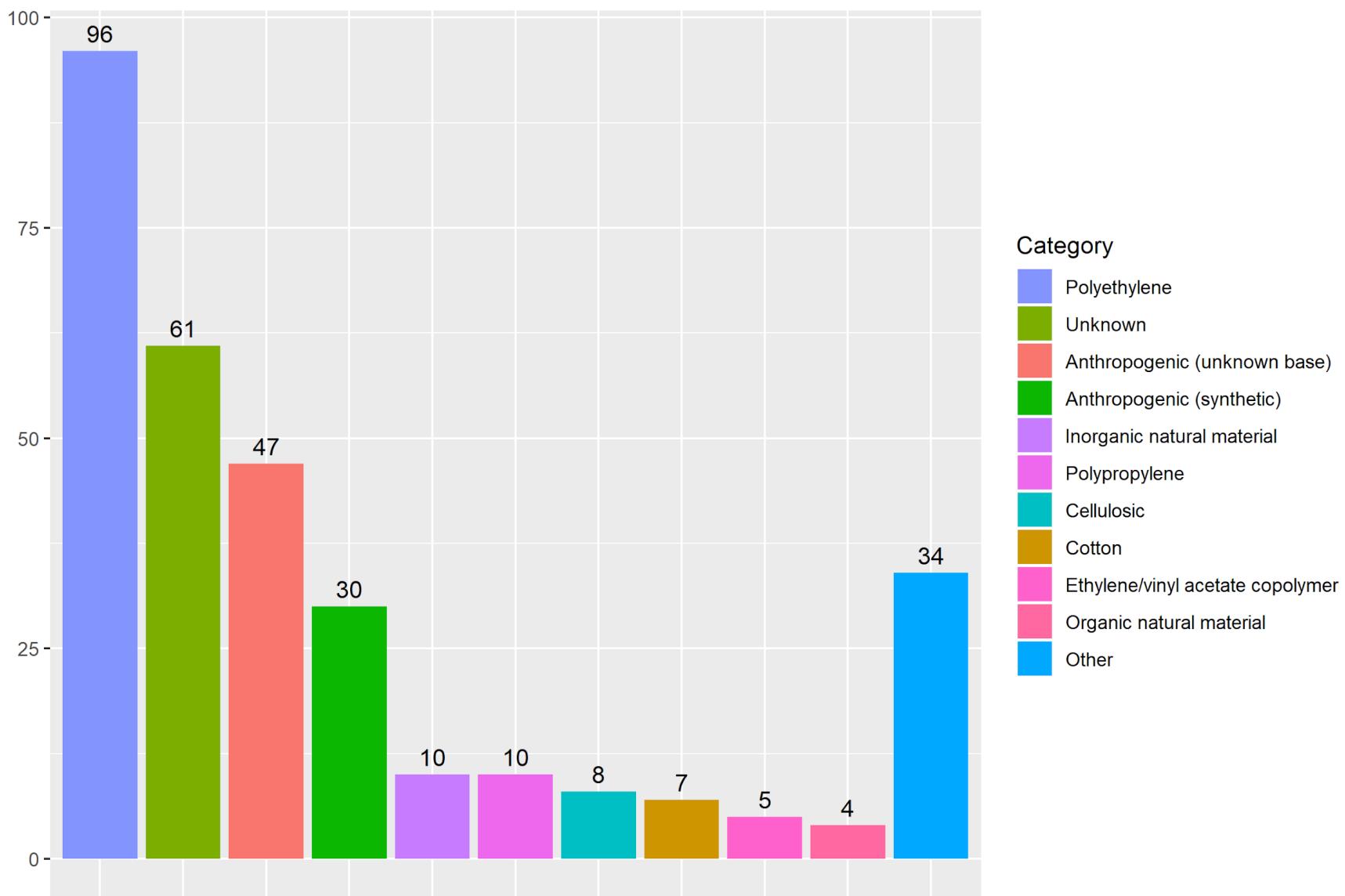


Figure 3.8. Count and composition of fragments analyzed by spectroscopy. Polyethylene, anthropogenic (synthetic), polypropylene, and ethylene/vinyl acetate copolymer are considered to be plastic.

FOAM CONCENTRATIONS AND COMPOSITION

Foam was identified at all secondary treatment facilities (i.e., CCCSD, EBMUD, SFPUC, and EBDA) as well as at one tertiary treatment facility (San Jose; Figure 3.5). Foam concentrations ranged in the secondary facilities from 0.001 to 0.069 foam microparticles/L; in contrast, the concentrations from San Jose were lower, around 0.003 foam microparticles/L. Foam counts in Sample A from CCCSD are a low-biased estimate, as over 100 more small foam pieces were identified in this sample but were not quantified and are therefore not included in the concentration.

The particle size distribution for foams was similar to that of fragments, with most particles in the 250 µm to 500 µm range. Most of the foam was white. Approximately 40% of the foam particles were analyzed by spectroscopy. The vast majority (73%) were classified as mixtures of stearates, lubricants, and waxes, which are not plastics. Stearates, a component of soap scum, are readily identified by FTIR analyses. Analysis of lubricants is more challenging, but can be done by Raman spectroscopy. Long-chained lubricant polymers were identified, with the analyst noting that these materials are not plastic based on secondary characteristics such as their fragile texture. Only one particle was identified as polystyrene foam. Overall, approximately 17% of the foam particles that underwent spectroscopy were identified as plastic.

At the EBDA facility, fewer foam particles were observed and the composition was somewhat unique. Of the foams that underwent spectroscopy, approximately a third were very similar in shape and color and contained titanium dioxide, a common ingredient in white paint, sunscreens, and food coloring.

SPHERE CONCENTRATIONS AND COMPOSITION

Spheres were identified in all of the samples from secondary facilities in relatively low amounts (concentrations ranging from 0.0005 to 0.008 spheres/L; Figure 3.5). The diameter of most of the spheres was between 250 µm and 500 µm (Figure 3.7). Approximately 70% of the spheres (75 of 111) were analyzed by spectroscopy; 90% of these spheres were identified as polyethylene.

Effluent from just one tertiary treatment facility, San Jose, contained spheres (42 particles); however, the high counts only occurred for one of the two sampling events and seemed to consist largely of clear spheres. Clear spheres that underwent spectroscopy were identified as polyethylene, ethylene/vinyl acetate copolymer, anthropogenic unknown, and organic material. In contrast, in the second sampling event, only four spheres were identified, none of which were clear. Spectroscopy on two of these spheres determined they were polyethylene.

FILM CONCENTRATIONS AND COMPOSITION

Films were not detected frequently; four samples contained between 16 and 57 particles (concentrations ranging from 0.005 to 0.025 films/L; Figure 3.5). For the remaining samples, film counts ranged from zero to six pieces. Approximately 57% of the particles (84 of 147) underwent spectroscopy. Almost half were identified as polyethylene film, 18% were identified as anthropogenic unknown or unknown, and 8% were identified as anthropogenic synthetic. Approximately 75% of the film particles identified were plastic.

Discussion

Standardizing methods to minimize and account for background contamination

The lack of standardized methods for the collection, extraction, and analysis of microplastics has been widely acknowledged as a significant challenge (European Chemicals Agency, 2019; Lares et al., 2018; Science Advice for Policy by European Academies, 2019; Simon et al., 2018; Wolff et al., 2019). Early studies often did not include field blanks, field duplicates, or other QA/QC samples. More recent studies tend to include some QA/QC samples (e.g., Dris et al., 2015; Lares et al., 2018; Leslie et al., 2017; Mason et al., 2016; Mintenig et al., 2017; Simon et al., 2018; Wolff et al., 2019); however, reported results suggest the significance of background contamination varies widely among studies. In part, this is attributable to the lack of standardization among the myriad methods used to extract and identify microparticles, the volumes sampled, and the size fractions targeted. Overall, the levels of microparticles in blanks collected for this study were on the low end of reported blank values, suggesting relatively minimal contamination derived from sample collection and analysis (Figure 3.9).

Some studies of microparticles in wastewater identify low to no background contamination based on field and laboratory blanks. Leslie et al. (2017) identified a range of zero to three fibers in their laboratory blanks; no other morphology was detected and information regarding the preparation of the blanks was not provided. Lares et al. (2018) conducted repeat sampling of a wastewater treatment plant and collected multiple field blanks using distilled water; on average, four microparticles were identified in their blanks.

In contrast, other studies reported blank concentrations higher than those observed in this study. In a study of effluent from 12 German wastewater facilities (Mintenig et al., 2017), three laboratory blanks consisting of tap water filtered through a 3 µm filter were processed with the field samples. Particles and fibers were identified in the blanks, with an average of 21 particles and 130 fibers in a 150 L blank sample. Similarly, in a study of a German secondary wastewater treatment facility, the laboratory extraction blank had 66 microparticles, approximately 40% of

which were polytetrafluoroethylene (PTFE) and were attributed to laboratory equipment and tools (Wolff et al., 2019).

Simon et al. (2018) reported an average of 2,111 microparticles in three laboratory blanks. They targeted a significantly lower microparticle size fraction than this study (10 µm vs. 125 µm). Several researchers have postulated that microparticle counts increase with decreasing size (Covernton et al., 2019). In addition, Simon et al. (2018) employed subsampling methods using vortexes to homogenize the sample, which may have caused additional fragmentation.

Fibers were the major particle type observed in field and laboratory blanks, which attests to their ubiquitous presence in the environment. Airborne deposition has been shown to be a significant pathway, particularly for fibers (Cai et al., 2017; Dris et al., 2016, 2015). Sample contamination by airborne fibers has been identified as a serious issue (Science Advice for Policy by European Academies, 2019; Silva et al., 2018). Development of standard methods for the collection and analysis of microplastics that mitigate the contamination of samples with airborne fibers during collection and analysis are needed.

While collection of field and laboratory blanks is now more common, there is no standard method for assessing field samples relative to measured levels of blank contamination. In some instances, the blank sample counts are subtracted from the field sample counts (Brander et al., in review; Leslie et al., 2017). More often, the blank data are reported alongside field samples to allow the reader to make an independent assessment (Dris et al., 2015; Lares et al., 2018; Simon et al., 2018; Wolff et al., 2019). We have chosen to present the blank data with the field data.

Bay Area effluent concentrations of microparticles are consistent with prior studies

The concentrations of microparticles reported in this study are consistent with the prior screening study of Bay wastewater treatment plant effluent (Sutton et al., 2016), which reported an average of 0.086 microparticles/L. Slightly lower counts in this study may be attributed in part to the longer sampling period (24-hours vs. 2-hour peak flow), as well as to the possibility of increased dilution in this study compared to the prior study, which occurred during drought conditions when water conservation measures were widely practiced. In a study of EBMUD effluent, Dyachenko et al. (2017) observed lower counts in 24-hour composite samples (0.024 microparticles/L) vs. two-hour peak flow samples (0.16 microparticles/L).

In addition, a different analytical laboratory and extraction method was used in the prior screening study, which may contribute to variation in results. Lastly, variation may arise from differences in facility selection. Six of the eight facilities were included in both studies; however,

the remaining two were different (San Mateo and San Francisco Airport (prior study) vs. Sunnyvale and SFPUC (this study)).

The range of microparticle concentrations observed in this study is on the low end of values reported in the literature (Figure 3.9), although the use of different size fractions examined, extraction methods, and analysis methods make comparisons among studies challenging. For example, Swedish researchers found 0.008 microparticles/L in effluent from a small Swedish wastewater treatment plant (Magnusson and Norén, 2014). However, the researchers used visual identification techniques and as a result, focused on larger particles above 300 µm. The reported particle concentrations were lower than this study, but this might be expected given our sample collection method captures many particles in the 125 to 300 µm range (Figure 3.7). Conversely, several studies (Simon et al., 2018; Wolff et al., 2019) that included a lower size fraction, down to 10 µm, reported higher particle counts than this study (Figure 3.9).



The extraction method can also affect the particle counts. EBMUD researchers found 0.02 microparticles/L in final effluent (Dyachenko et al. 2016), lower than the concentrations reported for the same facility in this study (0.14 and 0.07 microparticles/L). In the Dyachenko et al. (2016) study, fatty acids and cellulose fibers were found to be major interferences in the identification of microplastics. As a result, the laboratory developed and employed a more aggressive method to extract microplastics, involving hexane rinses and repeat peroxide oxidation steps, which may have degraded some of the plastics, resulting in lower counts.

In summary, the range of observations reported for microparticles in effluent spans four orders of magnitude (Figure 3.9); much of this variation is likely due to factors related to different methods of sample collection and analysis. Approximately half of these studies report concentrations less than 0.1 microparticles/L, which are consistent with the results of this study. In addition, the current study results are consistent with the prior Bay Area screening study (Sutton et al., 2016).

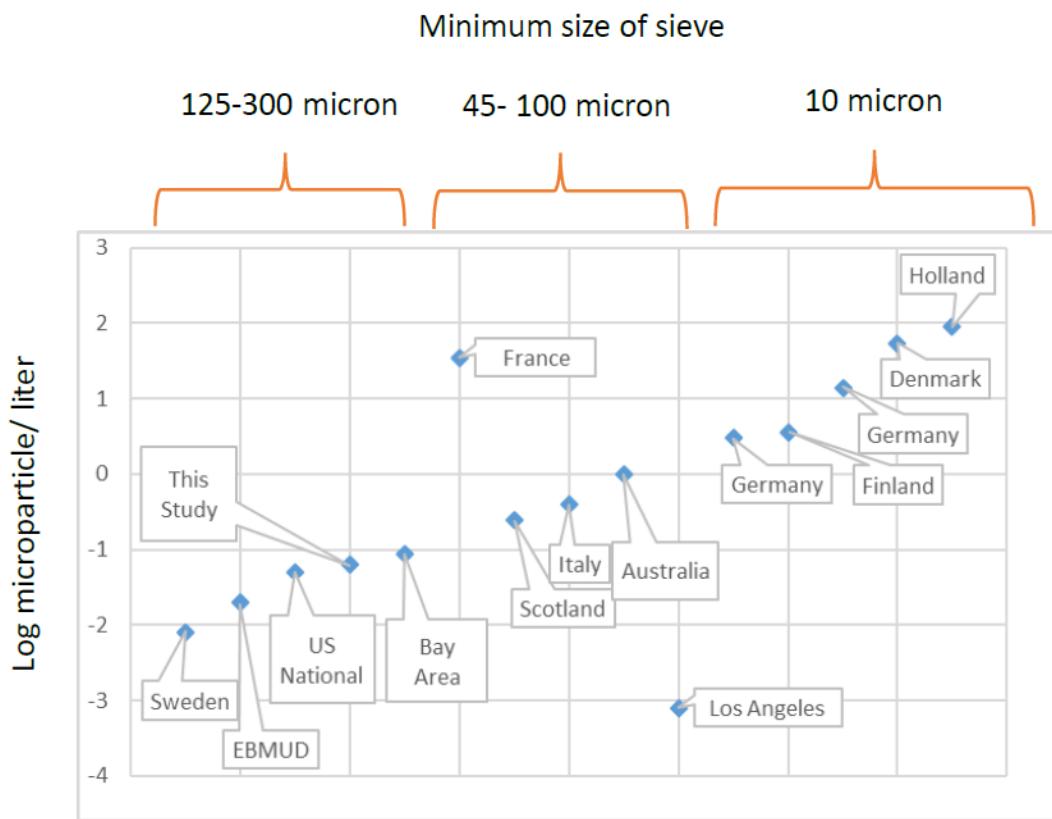


Figure 3.9. Microparticle concentrations reported in the literature (see Appendix Table 3A.1 for references). Note x-axis goes from largest minimum sieve size on left to the smallest on right.

Calculation of microparticle and microplastic loads to San Francisco Bay

In aggregate, the eight facilities discharged approximately 91 million microparticles per day (Table 3.2; Figure 3.10). The aggregated microparticle load to the Bay was estimated by calculating the total number of microparticles discharged for each facility during each sampling event using the total 24-hour flow reported on the day of sampling multiplied by that day's measured microparticle concentration (i.e., particle count divided by sample collection volume). The two sampling events for each facility were averaged, and daily discharge rates for all sampled facilities were summed to calculate the combined total discharge per day at the eight sampled facilities.

The individual daily load estimates (Table 3.2) varied between 0.3 to 45 million particles per facility. This range is consistent with the average load estimated by Mason et al. (2016) of 4 million microparticles per facility per day (based on a national survey of 17 facilities), but is significantly less than the estimate of 160 million microplastics from an Italian wastewater treatment plant (Magni et al., 2019).

If it is assumed that the discharges of these eight facilities are representative of all Bay Area facilities, then it is possible to develop an estimate of microparticles discharged per year by all Bay Area wastewater treatment plants by taking the total daily load estimate of 91 million microparticles per day for the eight facilities, multiplied by 365 days per year and then divided by 70% (the percentage of the total flow that these eight facilities represent). The estimated annual discharge to the San Francisco Bay is 47 billion microparticles per year. This is substantially less than the estimated 10.9 trillion microparticles discharged from stormwater (see Chapter 2 Stormwater).

Based on the spectroscopic data, not all microparticles are plastic. The number of particles that are definitively plastic is not known because it was not feasible to conduct spectroscopy on all particles collected. Spectroscopy was conducted on approximately 40% of the particles, with 35% of the microparticles examined confirmed to be plastic. Another 30% were classified as anthropogenic unknown (meaning that they may or may not be plastic).

This information was used to estimate the amount of microplastics discharged into the Bay. As a first approximation, if 35% of all particles are assumed to be plastic based on the spectroscopy results, then 32 million microplastics are discharged per day by these eight facilities.

However, it is possible that some fraction of the particles classified as anthropogenic unknown are plastic as well. The majority of such particles in this study were fibers (80%), and approximately 60% of textiles today are made from nylon (polyamide) and polyester (Almroth et al., 2018). As an upper bound, if we assume that 60% of the anthropogenic unknown fibers are plastic (approximately 24% of total particles examined with spectroscopy), then the total amount of microplastics discharged from these eight facilities is 45 million microplastic particles per day. To calculate a lower bound, we assume that all of the microparticles detected in the blanks are plastic, and subtract by morphology the average of the laboratory and field blanks from the samples (i.e., average number of fibers in the blanks was 16, average fragments in blanks was 4, and average film in blanks was 1). The blank subtraction is conducted using the combined values of the 125 and 355 μm sieves, as fibers may or may not be caught on a particular sieve size, as it is a function of the orientation of the fiber. Following this blank subtraction, the estimate of microplastics discharged is 29 million microplastics per day. The estimate of discharge from these eight facilities to the Bay therefore ranges from 29 to 45 million microplastics per day, with a well-supported estimate of 32 million microplastics per day. Assuming similar compositions for the remaining wastewater treatment plants, this estimate translates to 46 million microplastics per day, or 17 billion microplastics per year.

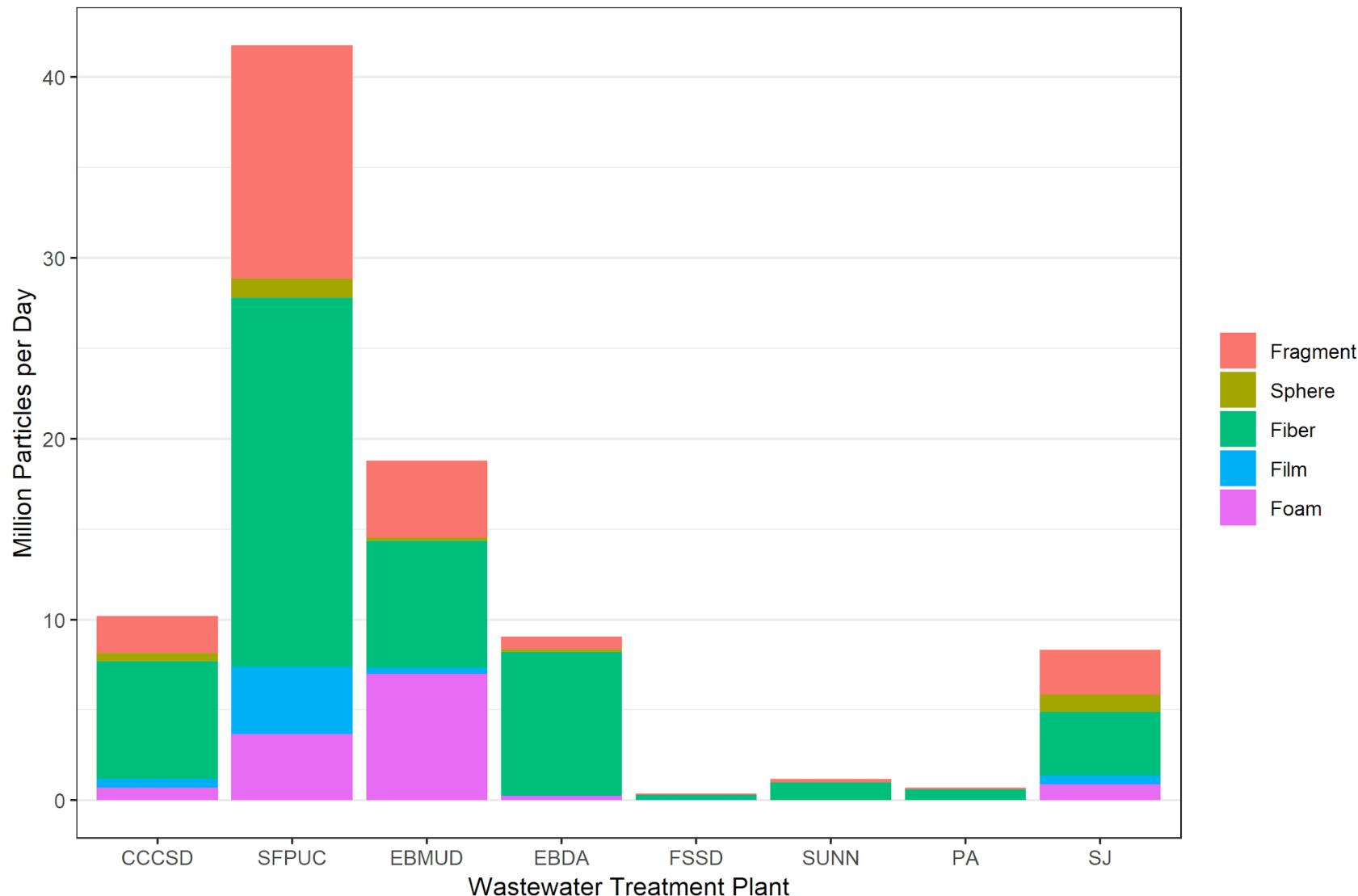


Figure 3.10. Estimated daily loading of microparticles by morphology for each facility. Fragment counts for the following facilities were within the range of the average of field and laboratory blanks plus 2 times the standard deviation: CCCSD (1 sample); FFSD (1 sample), Sunnyvale both samples, and Palo Alto (both samples) and should be treated with caution.

Textiles as sources of fibers to wastewater

Overall, fibers were the most numerous type of particle observed in this study, accounting for an average of 55% of all particles across all eight plants. This result is consistent with the literature (Dris et al., 2015; Magnusson and Norén, 2014; Murphy et al., 2016; Sutton et al., 2016; Talvitie et al., 2017; Ziajahromi et al., 2017). Textiles, including clothing, carpets, and other household furnishings, are hypothesized to be a major source of these fibers, and can be released during washing (Almroth et al., 2018; Hartline et al., 2016; Jönsson et al., 2018). A single garment can produce more than 1,900 fibers per wash (Browne et al., 2011).

For synthetic clothing, the amount of fibers released during washing is a function of fabric type, weave, age of fabric, and washing machine (Almroth et al., 2018; Hartline et al., 2016). For example, polyester fleece sheds nearly an order of magnitude more fibers than polyester knit fabric (7,360 fibers/m²/L vs. 87 fibers/m²/L; Almroth et al. 2018). Worn garments tend to shed more fibers than new garments (Almroth et al., 2018; Hartline et al., 2016), and front-loading washing machines have been shown to release seven times fewer microfibers per unit mass than top loaders (Hartline et al., 2016), with the increase in fiber loss attributed to abrasion that occurs during the agitation cycle.

Polyester is frequently identified as the major synthetic fiber polymer observed in wastewater (Almroth et al., 2018; Browne et al., 2011; Hernandez et al., 2017; Mintenig et al., 2017; Murphy et al., 2016; Wolff et al., 2019). Browne et al. (2011) found that polyester fibers were most frequently detected (67%), followed by acrylic (17%) and nylon (or polyamide; 16%). Zajahroni et al. (2017) found that the most commonly detected microplastics in Australian effluent were polyester fibers and irregularly shaped polyethylene particles, thought to originate from washing textiles and personal care products, respectively. Lares et al. (2018) found that polyester was the most frequently detected fiber composition, comprising 96% of plastic fibers. Murphy et al. (2016) identified polyester as the most common plastic type in final effluent (28%), followed by nylon (20%), polypropylene (12%), and acrylic (12%).

Synthetic fibers consisting primarily of nylon (polyamide) and polyester are approximately 60% of total global fiber production (Almroth et al., 2018) and are used widely in textiles for clothing and home furnishings as well as more industrial applications such as fibers in car tires, conveyor belts, and ropes. Based on these findings, it is likely that some of the fibers that could only be identified as anthropogenic unknown in this study are plastic. In summary, it appears likely that fibers from textiles are a major source of microplastics to the Bay and to the aquatic environment in general.

Personal care products as sources of spherical microbeads to wastewater

Recent federal legislation, the United States Microbead-Free Waters Act of 2015, has banned the use of plastic microbeads in wash-off personal care products such as facial scrubs, toothpaste, and body wash (McDevitt et al., 2017). This legislation superseded a more comprehensive California ban (AB 888, Waste management: plastic microbeads). The implementation of the federal ban has been phased in, with the use of microbeads in wash-off personal care products prohibited by July 1, 2018. Consumer use of these products purchased before the full implementation of the ban may result in some discharge of microbeads after the ban.

Microbeads are generally defined as particles manufactured in a size range of roughly 5 µm to 1 mm (Rochman et al., 2015) and intentionally added to personal care or cleaning products. In a survey of facial cleansers available in the Bay Area, microbeads ranged in size from 60 to 800 µm, with a mean of 264 µm (Chang, 2015). Microbeads are most frequently made of polyethylene, polylactic acid, polypropylene, or polystyrene (Rochman et al., 2015).

The term microbead is somewhat misleading, as clearly identifiable, bead-like spheres typically represent only about 10% of the composition of microbeads, with the vast majority of the particles comprised of fragments (Carr et al., 2016; Fendall and Sewell, 2009). Spheres were present at low concentrations relative to other particle types in this study. The highest concentration observed was 0.008 spheres/L at SFPUC. The size range and composition (i.e., polyethylene) of the spheres detected in this study suggested that many of these were likely derived from personal care products (Chang, 2015; Fendall and Sewell, 2009). While it is likely that some portion of fragments detected in this study were from microbeads as well, it is impossible to attribute the portion of fragments given the wide diversity of potential fragment sources.

If the spheres detected in effluent are assumed to be 10% of the total microbeads, then the estimated concentration in SFPUC effluent would be 0.08 microbeads/L. This estimated concentration is consistent with an earlier estimate of 0.10 microbeads/L in effluent calculated by Rochman et al. (2015), and higher than the average concentration of 0.017 microbeads/L calculated by Mason et al. (2016). Other facilities examined in this study discharged lower concentrations of spheres, or none at all, suggesting lower contributions from microbeads.

Most foam in wastewater was not plastic

Foam was identified at several facilities; however, only one particle was identified as polystyrene foam. Over 70% of these foams were identified via spectroscopy as containing stearates, lubricants, or waxes, materials that are not plastic. Calcium stearate is widely used in

soaps, lubricants, surfactants, and food; it is also the primary component of soap scum. Stearates have been identified in wastewater effluent previously (Dyachenko et al. 2017; Ziajahromi et al. 2017), with soaps a likely source.

Discussions with wastewater treatment facility personnel in this study did not identify a source of this type of foam within the plants, suggesting that this material enters the influent stream from an unidentified source.

Tertiary treatment facilities had lower microparticle concentrations

Wastewater treatment facilities are able to remove microparticles with a relatively high efficiency, in the range of 83% to 99.9% (Carr et al., 2016; Dris et al., 2015; Michielssen et al., 2016; Mintenig et al., 2017; Murphy et al., 2016; Talvitie et al., 2017). Based on studies of the treatment process, it appears that the bulk of the removal occurs during primary treatment, with one study citing removal of 78% of the microparticles during primary treatment (Murphy et al., 2016).

A major finding of the present study is that facilities using tertiary treatment had significantly lower microparticle concentrations relative to facilities using secondary treatment. Bay Area facilities employ a variety of treatments, but all tertiary plants use dual media filters for finishing purposes.

Reports in the literature to date show conflicting results with regard to treatment, and some studies have found no correlation with type of treatment. Results of the prior screening study did not indicate a difference in effluent concentrations between Bay Area tertiary and secondary facilities (Sutton et al., 2016). Similarly, a study of 17 U.S. wastewater treatment facilities, evaluating six tertiary treatment facilities and 11 secondary treatment facilities (including the prior Bay Area results), also did not identify a significant difference based on treatment, although a limited evaluation of effluent pre- and post-tertiary treatment at one facility suggested a 15% reduction in particle concentrations due to tertiary treatment (Mason et al., 2016). In contrast, Michielssen et al. (2016) reported that a single facility using tertiary treatment had approximately 40% fewer microparticles than a secondary treatment facility (2.6 vs. 5.9 microparticles/L; Michielssen et al., 2016). We originally hypothesized that treatment would not affect microparticle concentrations in effluent, consistent with the prior Bay Area screening study (Sutton et al. 2016). Our results do not support this hypothesis.

The combined sewer system had more fragments and films

The highest microparticle counts and concentrations occurred at SFPUC, a combined sewer facility that receives stormwater and wastewater. SFPUC had the highest concentration of fragments, an average of 0.058/L, approximately four times the average at the remaining facilities with concentrations significantly above the blank (0.013 fragments/ L; Figure 3.5).

SFPUC also had the highest concentrations of film particles, with half of the films identified being found at this facility. As discussed in Chapter 2 Stormwater, fragments are more prevalent in stormwater. The finding of high concentrations of fragments in SFPUC suggests that urban stormwater runoff may also contribute microplastics to the Bay (see Chapter 2 Stormwater; Gilbreath et al, 2019). Although this study design targeted dry weather sampling, due to logistical constraints, it was not possible to sample SFPUC in the dry season and samples were collected approximately one month after a small wet weather event (0.3 inches). It is possible the greater abundance of fragments is due to stormwater runoff from rains the month before or residual materials in stormwater pipes.



There are relatively few microplastic studies of combined sewers; however, one study of a combined sewer facility that received stormwater found that discharges tended to be dominated by fragments, whereas a facility that received only sanitary influent was dominated by fibers (Mason et al., 2016). Season may also be a factor. In a seasonal study of a secondary wastewater treatment facility in Germany (Wolff et al., 2019), wet season samples were observed to have almost twice the number of microparticles than dry season samples (5.9 vs. 3.0 microparticles/L). The authors attributed the difference to higher flow velocity during the wet season, which reduced particle settling during the treatment process, resulting in more microparticles in the effluent.

The colors of fragments observed in effluent from SFPUC were more diverse than other Bay Area facilities, and included gold and silver fragments that were not identified in any other effluent. However, other more definitive indicators of stormwater, such as black rubbery fragments that might be indicative of tire wear, or paint fragments and reflective glass spheres potentially derived from road markings (Gilbreath et al, 2019), were not detected in SFPUC effluent.

Conclusions

The purpose of this study was to characterize microparticles and microplastics in treated wastewater. Effluent was analyzed from eight Bay Area wastewater treatment facilities, which collectively release approximately 70% of the treated effluent flow discharged to San Francisco Bay. Microparticles were identified in effluent from all eight facilities, discharging an average of 0.063 microparticles/L. The concentrations of microparticles observed in effluent in this study were consistent with the range of values reported in the literature. Fibers, followed by fragments, were the most frequently identified morphologies, a common observation in the literature. Based on the data collected, we estimate that 91 million microparticles enter the Bay each day from municipal wastewater.

Spectroscopic examination of a subset of particles suggests that approximately 17 billion microplastic particles enter the Bay annually. This supports a conceptual model that indicates municipal wastewater is a major pathway for microplastics to enter the Bay. Facilities employing tertiary treatment had significantly lower microparticle concentrations than secondary treatment facilities, suggesting that enhanced treatment may have multiple societal benefits, including reduction in microplastic as well as nutrients and other pollutants.

However, it is likely far more cost-effective to prevent pollution in the first place (e.g., bans on sources of microplastic pollution, such as microbeads) or to control it directly at the point of entry (e.g., providing filters for washing machines). An active area of research has been the generation of fibers from textiles as a result of washing (Hartline et al., 2016; Jönsson et al., 2018). Additional research to identify effective measures to prevent the release of fibers (i.e., yarn and fabric design), as well as measures to prevent fibers from being discharged to the sewer system (e.g., washing machine filters) are needed.

A major challenge in the field of microplastics is the resource-intensive nature of spectroscopic characterization of individual particles. Developing automated techniques will be important for fully assessing fibers in samples with higher levels of contamination. Another challenge specific to those fibers that underwent spectroscopy was our inability to precisely identify the composition for more than half of the fibers due to interferences caused by dyes, resulting in their classification as anthropogenic unknown. Expanding spectral libraries specific to microparticles and microplastics, and improving our understanding of the dyes used on different textile materials, will be important for fully assessing fiber composition and potential sources.

References

- Almroth, B.M.C., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles; a source of microplastics released into the environment. *Environ. Sci. Pollut. Res.* 25, 1191–1199. <https://doi.org/10.1007/s11356-017-0528-7>
- Brander, S., Renick, V., Foley, M., Steele, C., Woo, M., Lusher, A., Carr, S.A., Helm, P.A., Box, C., Cherniak, S., Andrews, R., Rochman, C.M., in review. Sampling and QA/QC, or how many blanks do I need? A guide for scientists investigating the occurrence of microplastics across matrices.
- Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environ. Sci. Technol.* 45, 9175–9179. <https://doi.org/10.1021/es201811s>
- Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X., Chen, Q., 2017. Characteristic of microplastics in the atmospheric fallout from Dongguan City, China: Preliminary research and first evidence. *Environ. Sci. Pollut. Res.* 24, 24928–24935. <https://doi.org/10.1007/s11356-017-0116-x>
- Carr, S.A., Liu, J., Tesoro, A.G., 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Res.* 91, 174–182. <https://doi.org/10.1016/j.watres.2016.01.002>
- Chang, M., 2015. Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. *Mar. Pollut. Bull.* 101, 330–333. <https://doi.org/10.1016/j.marpolbul.2015.10.074>
- Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. *Sci. Total Environ.* 667, 124–132. <https://doi.org/10.1016/j.scitotenv.2019.02.346>
- Dehaut, A., Cassone, A.-L., Frère, L., Hermabessiere, L., Himber, C., Rinnert, E., Rivière, G., Lambert, C., Soudant, P., Huvet, A., Duflos, G., Paul-Pont, I., 2016. Microplastics in seafood: Benchmark protocol for their extraction and characterization. *Environ. Pollut.* 215, 223–233. <https://doi.org/10.1016/j.envpol.2016.05.018>

Chapter 3—Wastewater

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: A case study in Greater Paris. Environ. Chem. 12, 592.
<https://doi.org/10.1071/EN14167>

Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? Mar. Pollut. Bull. 104, 290–293.
<https://doi.org/10.1016/j.marpolbul.2016.01.006>

Dyachenko, A., Mitchell, J., Arsem, N., 2017. Extraction and identification of microplastic particles from secondary wastewater treatment plant (WWTP) effluent. Anal. Methods 9, 1412–1418. <https://doi.org/10.1039/C6AY02397E>

European Chemicals Agency, 2019. Annex XV Restriction Report – Microplastics (Version 1.1). European Chemicals Agency (ECHA).

Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face: Microplastics in facial cleansers. Mar. Pollut. Bull. 58, 1225–1228.
<https://doi.org/10.1016/j.marpolbul.2009.04.025>

Gilbreath et al, 2019. Multi-year water quality performance and mass accumulation of PCBs, mercury, methylmercury, copper and microplastics in a bioretention rain garden.

Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., Holden, P.A., 2016. Microfiber masses recovered from conventional machine washing of new or aged garments. Environ. Sci. Technol. 50, 11532–11538. <https://doi.org/10.1021/acs.est.6b03045>

Hernandez, E., Nowack, B., Mitrano, D.M., 2017. Polyester textiles as a source of microplastics from households: A mechanistic study to understand microfiber release during washing. Environ. Sci. Technol. 51, 7036–7046. <https://doi.org/10.1021/acs.est.7b01750>

Jönsson, C., Levenstam Arturin, O., Hanning, A.-C., Landin, R., Holmström, E., Roos, S., 2018. Microplastics Shedding from Textiles—Developing Analytical Method for Measurement of Shed Material Representing Release during Domestic Washing. Sustainability 10, 2457.
<https://doi.org/10.3390/su10072457>

Lares, M., Ncibi, M.C., Sillanpää, Markus, Sillanpää, Mika, 2018. Occurrence, identification and removal of microplastic particles and fibers in conventional activated sludge process and advanced MBR technology. Water Res. 133, 236–246.
<https://doi.org/10.1016/j.watres.2018.01.049>

Chapter 3—Wastewater

Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M., Vethaak, A.D., 2017. Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environ. Int.* 101, 133–142.
<https://doi.org/10.1016/j.envint.2017.01.018>

Lusher, A.L., Welden, N.A., Sobral, P., Cole, M., 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Anal. Methods* 9, 1346–1360.
<https://doi.org/10.1039/C6AY02415G>

Magni, S., Binelli, A., Pittura, L., Avio, C.G., Della Torre, C., Parenti, C.C., Gorbi, S., Regoli, F., 2019. The fate of microplastics in an Italian wastewater treatment plant. *Sci. Total Environ.* 652, 602–610. <https://doi.org/10.1016/j.scitotenv.2018.10.269>

Magnusson, K., Norén, F., 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant (No. C 55). IVL Swedish Environmental Research Institute.

Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., Fink, P., Papazissimos, D., Rogers, D.L., 2016. Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environ. Pollut.* 218, 1045–1054.
<https://doi.org/10.1016/j.envpol.2016.08.056>

McDevitt, J.P., Criddle, C.S., Morse, M., Hale, R.C., Bott, C.B., Rochman, C.M., 2017. Addressing the issue of microplastics in the wake of the Microbead-Free Waters Act—A new standard can facilitate improved policy. *Environ. Sci. Technol.* 51, 6611–6617.
<https://doi.org/10.1021/acs.est.6b05812>

Michielssen, M.R., Michielssen, E.R., Ni, J., Duhaime, M.B., 2016. Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed. *Environ. Sci. Water Res. Technol.* 2, 1064–1073.
<https://doi.org/10.1039/C6EW00207B>

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Res.* 108, 365–372.
<https://doi.org/10.1016/j.watres.2016.11.015>

Munno, K., Helm, P.A., Jackson, D.A., Rochman, C., Sims, A., 2018. Impacts of temperature and selected chemical digestion methods on microplastic particles: Impacts of temperature and digestion method on microplastics. *Environ. Toxicol. Chem.* 37, 91–98.
<https://doi.org/10.1002/etc.3935>

Chapter 3—Wastewater

Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. Environ. Sci. Technol. 50, 5800–5808.
<https://doi.org/10.1021/acs.est.5b05416>

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.

Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Scientific evidence supports a ban on microbeads. Environ. Sci. Technol. 49, 10759–10761. <https://doi.org/10.1021/acs.est.5b03909>

San Francisco Regional Water Quality Control Board - Region 2, 2019. San Francisco Basin Plan (downloaded Jan 23 2019 from website).

Science Advice for Policy by European Academies, 2019. A Scientific Perspective on Microplastics in Nature and Society (Version 2019.1.1).

Sedlak, M., Sutton, R., Box, C., Sun, J., Lin, D., 2017. FINAL Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries (SFEI Contribution No. 819). San Francisco Estuary Institute and 5 Gyres, Richmond, CA.

Silva, A.B., Bastos, A.S., Justino, C.I.L., da Costa, J.P., Duarte, A.C., Rocha-Santos, T.A.P., 2018. Microplastics in the environment: Challenges in analytical chemistry - A review. Anal. Chim. Acta 1017, 1–19. <https://doi.org/10.1016/j.aca.2018.02.043>

Simon, M., van Alst, N., Vollertsen, J., 2018. Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. Water Res. 142, 1–9.
<https://doi.org/10.1016/j.watres.2018.05.019>

Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. Mar. Pollut. Bull. 109, 230–235. <https://doi.org/10.1016/j.marpolbul.2016.05.077>

Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay (SFEI Contribution No. 798). San Francisco Estuary Institute, Richmond, CA.

Talvitie, J., Mikola, A., Setälä, O., Heinonen, M., Koistinen, A., 2017. How well is microlitter purified from wastewater? – A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. Water Res. 109, 164–172.
<https://doi.org/10.1016/j.watres.2016.11.046>

Chapter 3—Wastewater

Wolff, S., Kerpen, J., Prediger, J., Barkmann, L., Müller, L., 2019. Determination of the microplastics emission in the effluent of a municipal waste water treatment plant using Raman microspectroscopy. Water Res. X 2, 100014.
<https://doi.org/10.1016/j.wroa.2018.100014>

Ziajahromi, S., Neale, P.A., Rintoul, L., Leusch, F.D.L., 2017. Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. Water Res. 112, 93–99. <https://doi.org/10.1016/j.watres.2017.01.042>

CHAPTER

4

Microparticles and Microplastics

IN SURFACE WATER IN SAN FRANCISCO BAY AND ADJACENT NATIONAL MARINE SANCTUARIES

by Carolynn Box



Highlights

- ◆ This study measured microparticles and microplastics in surface water samples collected from San Francisco Bay and adjacent Greater Farallones, Cordell Bank, and Monterey Bay National Marine Sanctuaries during dry and wet seasons.
- ◆ Twenty-eight sites were sampled using a manta trawl designed to collect particles greater than 355 µm.
- ◆ Microparticles were identified and characterized as fragments, foams, spheres, or films. Just over half of the samples were also analyzed for fibers. Microparticle abundance was higher in Bay surface water than in the marine sanctuaries. Microparticle abundance in one of the Bay surface water samples was one of the highest observed in the world to date.
- ◆ Microparticle abundance was higher in Bay surface water samples collected during the wet season than the dry season. This result suggests that wet weather may mobilize microparticles and microplastics from the surrounding Bay Area watersheds. A statistically significant seasonal effect was not observed in the sanctuaries, at least partially due to the low abundance of microparticles observed.
- ◆ When microparticles from surface waters were quantified, the dominant morphology was fibers followed by fragments. Of the microparticles that underwent spectral identification, approximately 53% of fibers were determined to be plastic, while 87% of fragments, 68% of foam particles, 97% of spheres, and 83% of film particles were determined to be plastic.
- ◆ Polyethylene and polypropylene fragments, polystyrene foam, and polyethylene and polypropylene films made up a majority of the non-fiber microparticles that underwent spectroscopy. These polymer and particle types may be linked to the breakdown of single-use plastic items, packaging, and bags. Polyethylene beads were also identified in surface water samples, possibly linked to microbeads found in personal care and cleaning products.
- ◆ Average plastic microfiber abundance within the Bay ranged from 270,000 to 340,000 microplastic fibers/km² for the wet season and 40,000 to 59,000 microplastic fibers/km² for the dry season (upper and lower bound estimates). Average estimated microplastic abundance (excluding fibers) was 450,000 to 440,000 microplastics/km² for the wet season and ranged from 42,000 to 45,000 microplastics/km² for the dry season.
- ◆ Manta trawl sample collection is not an ideal method for capturing fibers. Sampling methods designed to collect more representative levels of fibers, as well as particles smaller than 355 µm, were deployed at some sites to test their effectiveness. Evaluation of these samples suggests the need for sampling larger volumes, and provides further evidence of the impacts of background contamination from fibers on data quality.

Objectives

The goal of this study was to quantify the abundance and composition of microparticles and microplastics in surface water samples from San Francisco Bay and the adjacent National Marine Sanctuaries (Greater Farallones, Cordell Bank, and Monterey Bay) during dry and wet seasons using multiple collection methods. For brevity, San Francisco Bay will be referred to as “the Bay” and the National Marine Sanctuaries will be referred to as “the marine sanctuaries.”

Through this assessment, we sought to achieve the following objectives.

1. **Quantify the abundance of microparticles and microplastics in surface water in the Bay and marine sanctuaries.** Characterizing a baseline data set for microparticles and microplastics in the surface waters of the Bay and the marine sanctuaries provides an answer to the first management question (MQ1) articulated in the RMP Microplastic Monitoring and Science Strategy for San Francisco Bay (Sutton and Sedlak, 2017): How much microplastic pollution is there in the Bay and the surrounding ocean? The baseline data can also be used to evaluate whether concentrations of microparticles and microplastics in the Bay increase or decrease over time (MQ4). Additionally, information from this study can be used to evaluate the effectiveness of management actions (MQ5), such as the national ban on microbeads (Microbead-Free Waters Act of 2015), which required companies to phase out the production and sales of rinse-off personal care products containing plastic microbeads. Because this study predates the phase-out deadline (July 1, 2019), the data will be useful for establishing a baseline from which this management action and others can be evaluated in the future.
2. **Characterize types of microparticles and microplastics found in surface water and their chemical composition.** Understanding the types of microparticles and microplastics found in the Bay and the marine sanctuaries will help determine the sources of microplastics (MQ3). This could help inform future management measures (MQ5) aimed at reducing microplastics in the Bay and coastal ocean.
3. **Compare microparticle and microplastic abundance, type, and composition within the Bay and the marine sanctuaries.** The transport of microparticles within the Bay and to the marine sanctuaries is influenced by a number of pathways and processes (MQ3), including stormwater and wastewater discharges, tidal flushing, circulation patterns, as well as coastal upwelling in the marine sanctuaries. For example, the North and Central Bays receive freshwater inflows from the Sacramento-San Joaquin River Delta and also experience frequent tidal flushing. In contrast, South and Lower South Bays receive less freshwater input and experience infrequent tidal flushing. These processes likely influence the abundance, type, and composition of microparticles and microplastics observed in different regions.

4. Evaluate microparticle abundance in Bay surface water relative to other studies. The results from this study are placed in context with the literature. A prior screening study suggested that surface water abundance of microparticles in the Bay was higher than those observed in other urbanized water bodies in North America (Sutton et al., 2016).
5. Refine methods for collecting and analyzing samples. A key step in quantifying the abundance of microparticles and microplastics (MQ1) is establishing appropriate field and laboratory methods for measurement. We explored multiple methods for surface water sample collection and examined performance using standard quality assurance measures.

As presented in the Microplastic Sampling and Analysis Plan (Sedlak et al., 2017), we evaluated the following hypotheses:

- ◆ Concentrations of microplastics in the Bay will be higher than in the ocean.
- ◆ Within the Bay, concentrations of microplastics will be higher in areas with limited flushing such as Lower South Bay.
- ◆ Concentrations of microplastics will be higher in samples collected during the wet season than the dry season.

In this report, we distinguish between microparticles, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and microplastics, which have been confirmed to be plastic through spectroscopy. The upper size boundary for microparticles and microplastics is typically defined as 5 mm, while the lower size boundary is operationally defined by the sample collection method. In this study, the surface water collection efforts focused on using a manta trawl and sieves with a pore size of 355 µm. Manta trawls have limitations in their ability to characterize all morphologies (Barrows et al., 2017; Covernton et al., 2019; Hidalgo-Ruz et al., 2012). Fibers are particularly challenging due to their shape (e.g., long and narrow); the orientation of the fiber can dramatically affect whether it is caught by the net or passes through.

This study builds on a previous pilot study that identified microparticles in San Francisco Bay surface water (Sutton et al., 2016). The pilot study suggested that microparticle abundance in Central and South Bay water was higher than observed in other water bodies with significant urban influence such as the Great Lakes (Eriksen et al., 2013a), Chesapeake Bay (Yonkos et al., 2014), and Puget Sound (Davis and Murphy, 2015). The present study of surface water is far more extensive, with samples collected throughout the Bay and marine sanctuaries, as well as during both wet and dry seasons. In addition, spectroscopy was conducted on a subset of microparticles to confirm whether the particles were plastic.

Methods

Summary

Between August 21, 2017, and March 30, 2018, a total of 73 field samples (i.e., 58 field samples, plus quality control samples that included eight duplicates and eight field blanks) were collected with a manta trawl. All manta trawl samples were characterized for fragment, foam, sphere, and film particle morphologies. Due to the high particle counts observed in the field samples and growing concerns that some fibers may escape from manta trawl samples, not all manta trawl samples were characterized for fibers. Instead, 34 (52%) of the 65 field samples (excluding field blanks) were analyzed for fibers as well as the other morphologies. Fibers were not counted in the remaining 31 samples. The decision to subsample was made after the first set of dry season samples were analyzed by the laboratory. This caused the distribution of fiber samples to skew slightly towards the dry season (i.e., 22 dry season samples were analyzed vs. 12 wet season samples). With the intention to better understand fiber concentration and develop additional field monitoring options, 55 one-liter grab samples (i.e., 49 field samples, plus quality control samples that included three duplicates and three field blanks) were collected at each of the monitoring sites.

Site selection

Surface water samples were collected using a manta trawl at 17 sites throughout the Bay (Figure 4.1) and 11 sites within Monterey Bay, Cordell Bank, and Greater Farallones National Marine Sanctuaries (Figure 4.2) to provide robust spatial coverage of the Bay and adjacent marine sanctuaries (Table 4.1). Surface water samples collected using a pump method and 1 L grab method are shown in Figure 4.3.

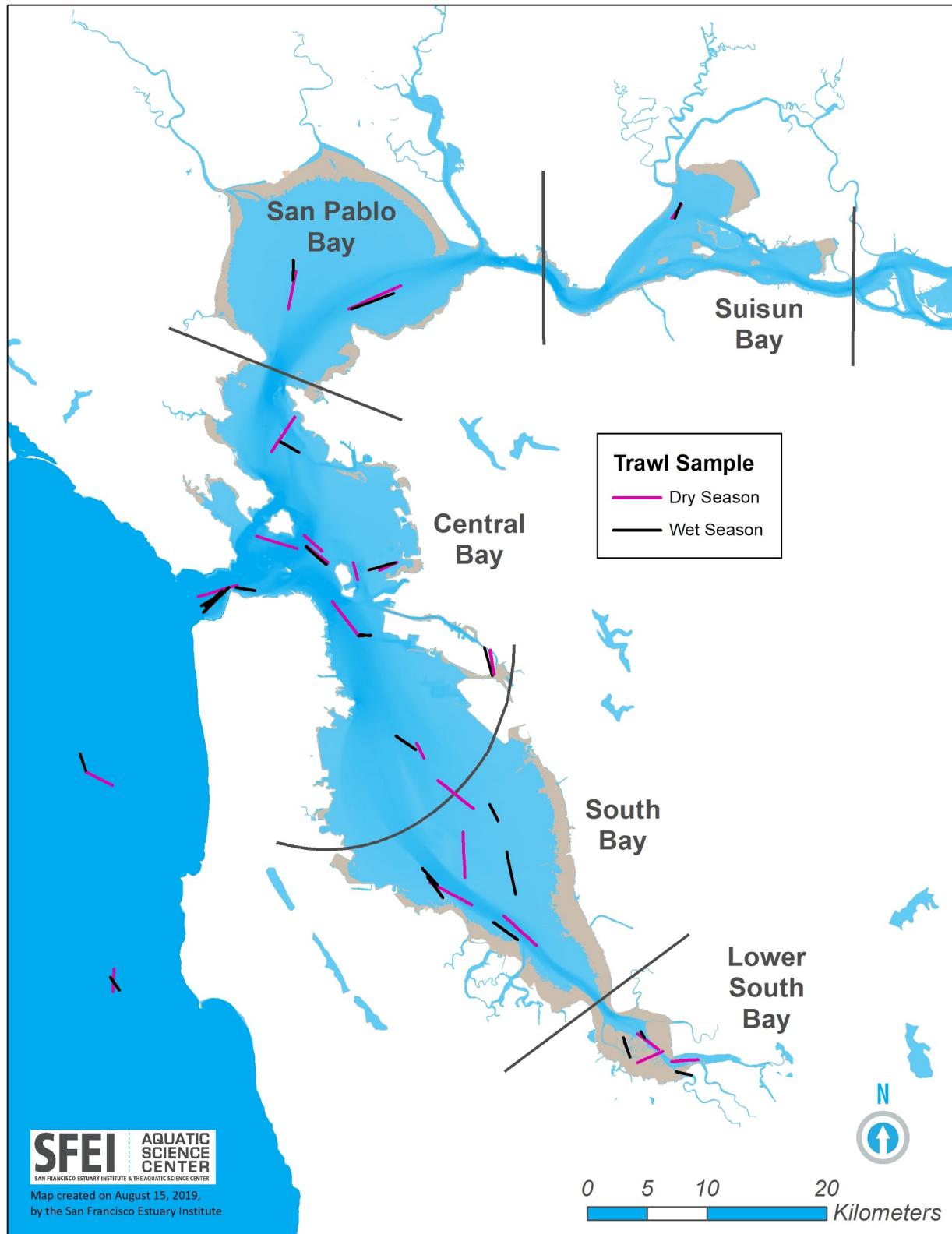


Figure 4.1. Manta trawl sample sites within San Francisco Bay. Subembayments are delineated by longer black lines, including Suisun Bay, San Pablo Bay, Central Bay, South Bay, and Lower South Bay.

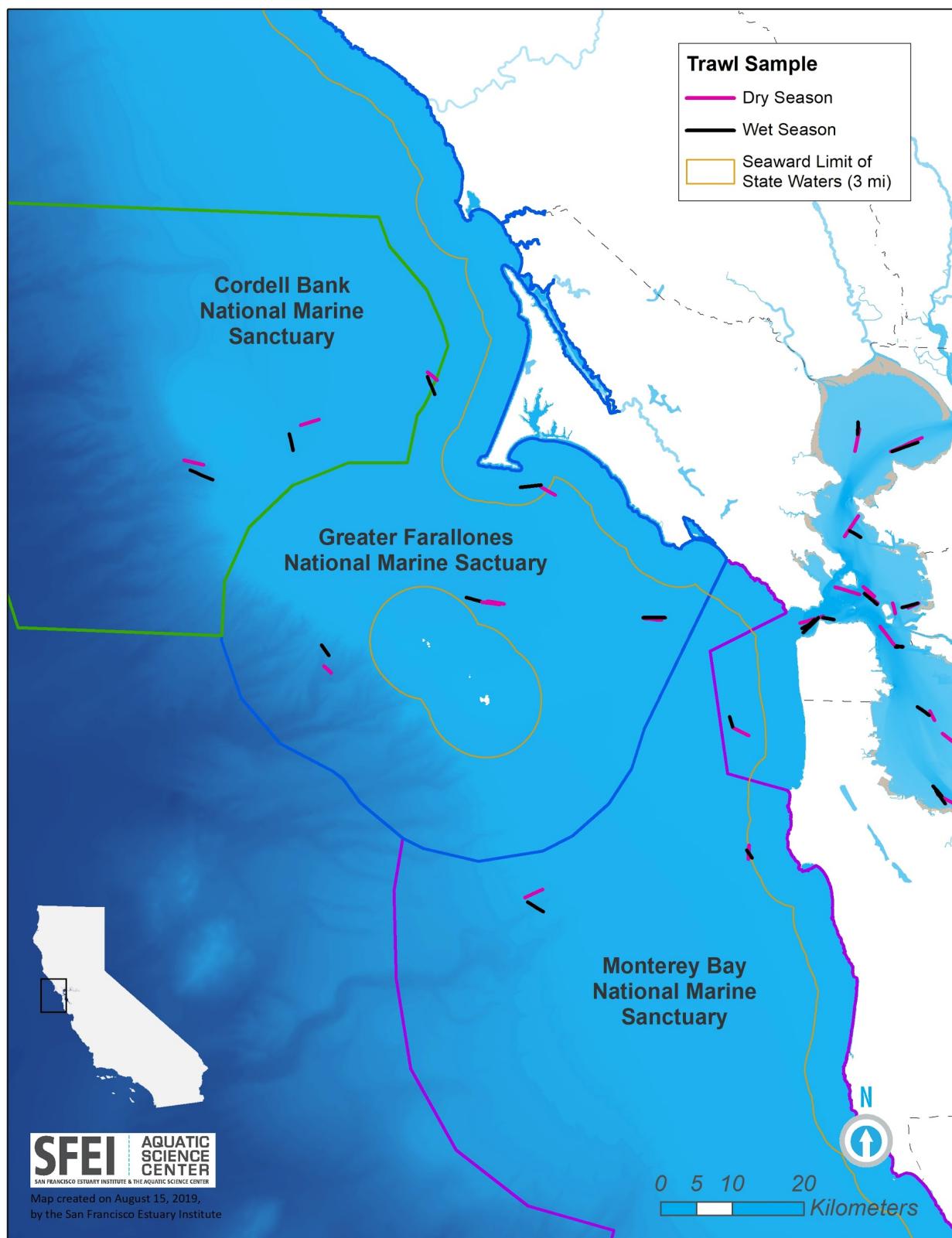


Figure 4.2. Manta trawl sampling sites within National Marine Sanctuaries.

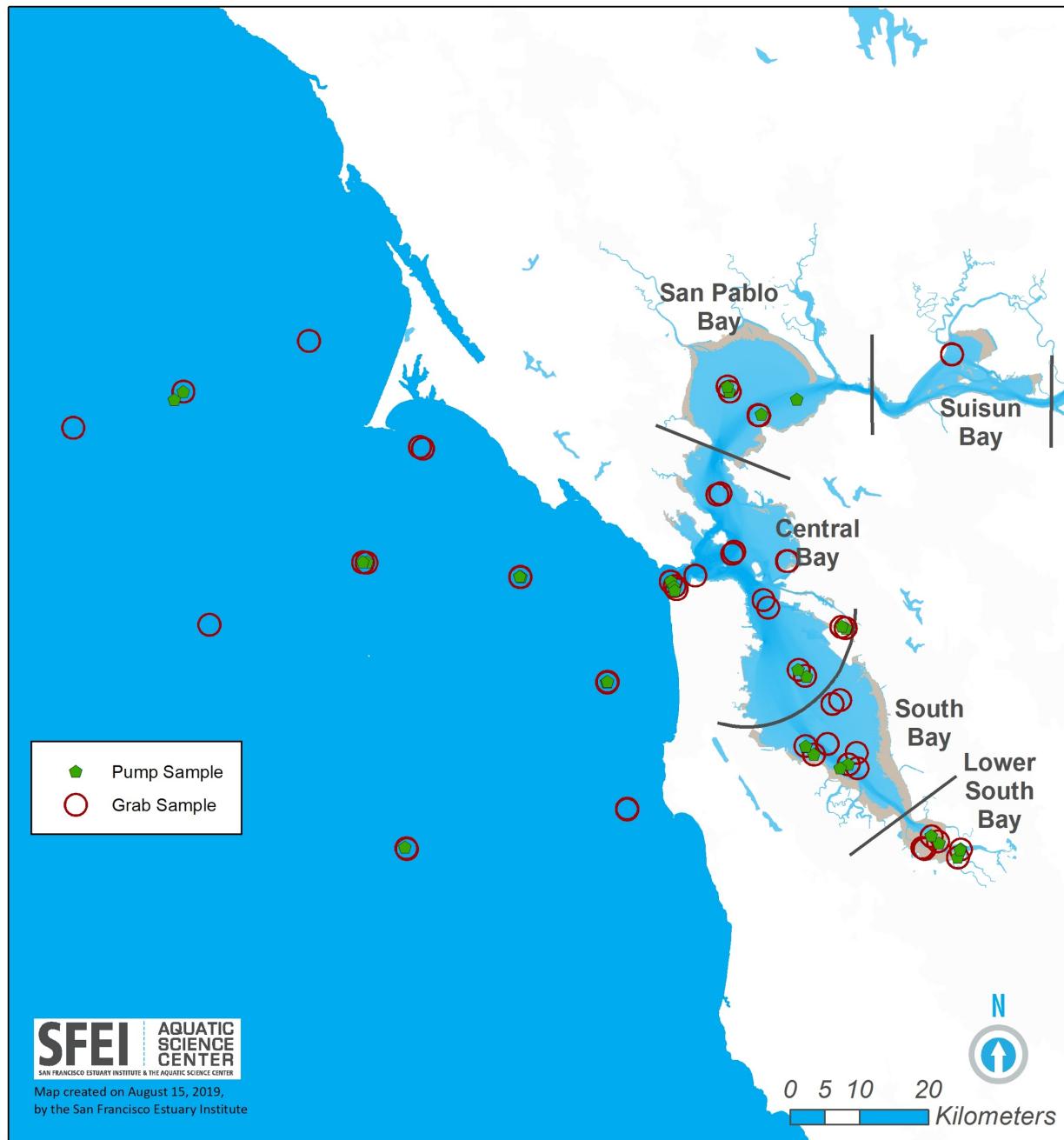


Figure 4.3. Location of pump and 1 L grab samples. Black lines indicate different sections of the Bay, including Suisun Bay, San Pablo Bay, Central Bay, South Bay, and Lower South Bay.

Table 4.1. Type of samples (field samples, field duplicates, and field blanks) collected at monitoring sites via manta trawl, 1 L grab, and pump. Fragment, foam, sphere, and film particles were characterized in all samples; fibers were characterized in all 1 L grab samples and in designated (bold) manta trawl samples only. (b) = blank; (d) = duplicate; ** = sample collected but not processed.

Monitoring site	Dry season manta trawl sample*	Wet season manta trawl sample*	Dry season 1 L grab sample	Wet season 1 L grab sample	Dry season pump sample	Wet season pump sample
SPB1	X	X	X	X		
SPB2	X	X	X	X	X	X
SPB3	X (b)	X (b)	X	X		X
CB4	X	X	X	X		
CB5	X (d)	X	X	X		
CB6	X	X	X	X		
CB7	X	X (d)	X (d)	X		
CB8	X (d)	X	X	X	X	X
CB9	X	X (b)	X	X	X	X
SB10	X	X	X	X		
SB11	X	X	X	X		
SB12	X	X (d)	**	X	X (d)	X
SB13	X (b)	X	X (b)	X (b)	X (b)	X
LSB14	X	X	**	X	X	X
LSB15	X	X (d)	**	X		
LSB16	X	X	X	X	X (d)	X
Alcatraz Site	X					
CBNMS22	X	X (b)	X			
CBNMS23	X	X	X		X (b)	X
CBNMS24	X	X	X			
GFNMS25	X (b)	X	X (b)	X		

Monitoring site	Dry season manta trawl sample*	Wet season manta trawl sample*	Dry season 1 L grab sample	Wet season 1 L grab sample	Dry season pump sample	Wet season pump sample
GFNMS26	X (d)	X	X (d)	X	X (d)	X
GFNMS27	X	X	X			
GFNMS28	X	X	X	X	X	X (d)
MBNMS29	X	X (d) (b)	X	X	X	X
MBNMS30	X	X	X	X (d)	X	X (d)
MBNMS31	X	X	X	X		
MBNMS32	X	X	X	X		X

The Bay was divided into four segments: Lower South Bay, South Bay, Central Bay, and North Bay (including Suisun Bay and San Pablo Bay; Figure 4.1). Monitoring sites were distributed within each segment; more samples were collected in Central Bay because of its large geographic area and the fact that it is a zone of convergence for northern and southern segments, as well as incoming oceanic water. Additional samples were also collected near the Golden Gate Bridge (MBNMS29) to provide data for modeling the flux of microplastics between the Bay and marine sanctuaries (Chapter 7 Transport Model).

The three marine sanctuaries included in this study encompass a combined total of 27,648 km² (i.e., Cordell Bank with 3,331 km², Greater Farallones with 8,534 km², and Monterey Bay with 15,783 km²). Three sites were sampled within Cordell Bank, four sites within Greater Farallones, and two sites within Monterey Bay (Figure 4.2), with two additional sites just west of the Golden Gate Bridge, adjacent to Monterey Bay National Marine Sanctuary (we consider these to be part of Monterey Bay).

Sample collection

At each site, we employed conventional manta trawl field methods to capture particles larger than 355 µm. In addition, we evaluated new methods for collection of smaller microparticles using a pump system and a 1 L grab; both methods were designed to collect representative samples of fibers, as well as particles smaller than 355 µm (Tables A-4.1 and A-4.2; Barrows et al., 2018, 2017; GESAMP, 2019; Miller et al., 2017). To assess seasonal variability, each site was sampled twice: once during the dry season and once during the wet season following a significant storm event. Sampling occurred at the site located south of Alcatraz on September 18, 2017, during an educational event; this site was only sampled during dry weather.

California has a Mediterranean climate in which precipitation mainly occurs between November and May. Dry season sampling for the Bay and marine sanctuaries occurred between August 21, 2017, and November 5, 2017. Wet season sampling occurred between November 16, 2017, and March 31, 2018, following storm events. A wet weather event was defined as 1.3 cm of rainfall within 24 hours, and such an event triggered field sampling. With one exception, all wet weather events met these criteria. At sites SB10, SB11, SB12, and SB13, approximately 1.6 cm of rainfall occurred over multiple days March 13–17, 2018 (i.e., not within a 24-hour period). This event was sampled because the probability of another storm event occurring was low.

Within the Bay, sampling occurred within three days of a defined wet weather event. For the marine sanctuaries, sampling occurred between five and ten days after a major storm system, to allow time for transport of microplastics out of the Bay. To determine if a rain event was significant enough to flush material out of San Francisco Bay and into the marine sanctuaries,

staff evaluated NOAA meteorological forecasts, Delta outflow, the magnitude and number of storm events leading up to the proposed wet weather sampling, and the salinity of the Bay. Details, including when the last rain event occurred and current directions and strength, were documented for each sample.

MANTA TRAWL SAMPLES (355 µm AND ABOVE)

A manta trawl (Figure 4.4) was used to collect microparticle samples from surface waters (Eriksen et al., 2013b; Free et al., 2014; Masura and Foster, 2015). The manta trawl is a modified Neuston net with a rectangular opening 16 cm high by 61 cm wide. The nylon net is 3 m long with a 335 µm mesh size. The sample is ultimately collected in a 30 x 10 cm² collection bag (referred to as a cod end) attached to the end of the net. The trawl was towed behind the vessel for 30 minutes at each site, with tow speeds below 3 knots, while the vessel maintained a consistent heading. If there was a current, sample collection was conducted against the current (i.e., the vessel was pointed into the current).

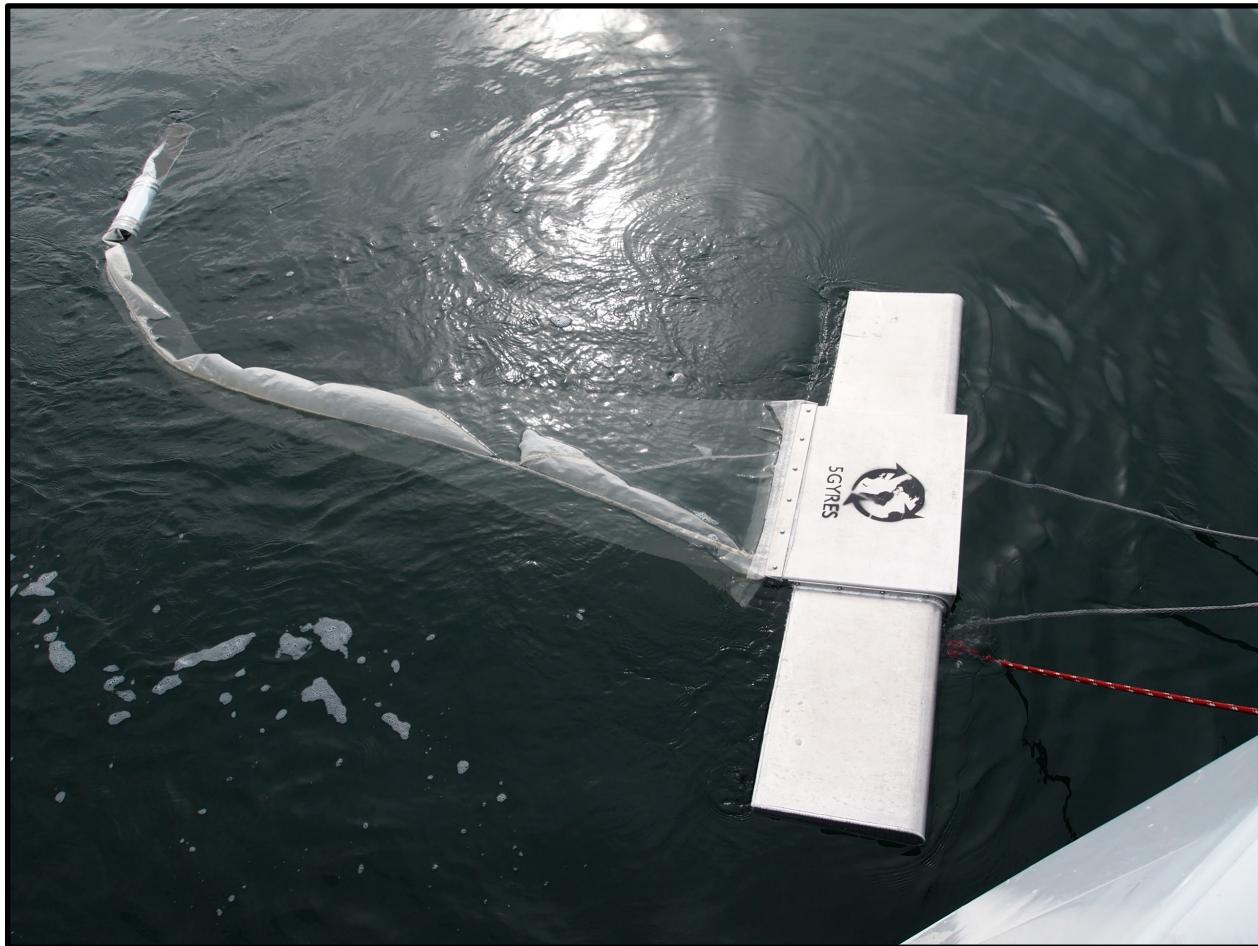


Figure 4.4. Manta trawl deployed alongside the *Derek M. Baylis* in San Francisco Bay. (Photo by Erika Delemarre.)

Depending on the site characteristics, different vessels were deployed to collect surface water samples. The *Derek M. Baylis*, a 65 ft auxiliary-powered sailing vessel, was used to sample most of the Bay sites and all of the sites in the marine sanctuaries. For shallow water sites (e.g., Lower South Bay and South Bay), it was necessary to use a smaller vessel (26 ft C-dory Tomcat motor boat owned by San Francisco Baykeeper).

The manta trawl was deployed with different techniques to avoid turbulent areas such as the boat's wake. On the *Derek M. Baylis*, the trawl was attached to the sail boom using a 45 m synthetic rope and extended over the port side of the vessel. On the Baykeeper boat, the manta trawl was deployed from the rear of the craft. Prior to deploying on both vessels, a flowmeter (General Oceanics Inc., Miami FL) with a standard rotor (2030R) was attached to the trawl to measure the amount of water passing through the trawl.



Once the tow was completed, the trawl was brought onboard with care taken to avoid the net coming into contact with surrounding objects. The net was rinsed from the exterior to flush materials toward the cod end. The cod end, held in place on the net using two marine-grade stainless steel hose clamps, was separated from the net and all material captured in the cod end was rinsed onto a 355 µm sieve with deionized water (DI). The contents of the sieve were then rinsed with DI water into a precleaned 500 mL glass sample jar (semi-VOA cleaned, Environmental Sampling Supplies (ESS), San Leandro CA). A clean metal spoon and metal tweezers were used to transfer recalcitrant material remaining on the screen to the sample jar. To prevent bacterial and algal growth, 70% isopropyl rubbing alcohol was added to each sample jar (enough to represent about 10% of the liquid in the jar). If needed, multiple sample jars were used. At one site (CB-9), 1.9 L clean jars were also used due to the high volume of material captured in the net.

Occasionally, large natural debris such as sticks and leaves were captured in the net. In these instances, the debris was rinsed thoroughly over the sieve to remove any particles and then discarded. Additionally, any small fish captured in the net were rinsed and released. Fish captured were documented on California Department of Fish and Wildlife forms and submitted to the State under the scientific collection permit obtained for this work (SCP-12364). All sample jars were placed in coolers before being transferred back to SFEI, where they were stored in a refrigerator at 2°C prior to being shipped to the University of Toronto for analysis.

Sieves were pre-cleaned in the SFEI laboratory with DI water and covered with foil to reduce contamination. Foil remained on the sieves until they were used in the field. In the field, sieves were cleaned using DI water and covered with foil between sites. The trawl was also precleaned prior to being brought onboard, and was cleaned between each sampling site using a high-pressure hose to remove particles that may have been stuck in the trawl. Cod ends were cleaned in the SFEI laboratory by rubbing the sides together to loosen any particles and rinsing with DI water. Pre-cleaned cod ends were used for each sample when possible; however, on several occasions—as a result of being at sea for multiple days—cod ends were cleaned onboard with tap water followed by a rinse with DI water prior to attaching the cod end to the manta trawl.

Eight field blanks (five collected in the Bay and three in the marine sanctuaries) and seven field duplicates (five in the Bay and two in the marine sanctuaries) were collected during manta trawl sampling to evaluate field procedural contamination and to assess the reproducibility of the sampling event. Field blanks were collected by pouring two liters of DI water through the manta trawl, followed by the same field processing procedures used for field samples. The cod end was removed and rinsed into a 355 µm sieve and all contents were rinsed into a sample jar. Field duplicates were collected by resampling along the same line as the primary sample, using the original coordinates, heading, speed, and duration.

To reduce contamination of samples in the field, crew and staff were instructed to avoid clothing that had the potential to shed (e.g., fleece) and wore natural materials where possible (e.g., cotton and wool). In addition, care was taken to keep sampling equipment in clean, sealed containers to avoid the deposition of airborne materials on to the equipment (e.g., sieves were covered with foil and placed in coolers or sealed boxes). Latex gloves were also worn when handling the samples.

ONE-LITER GRAB SAMPLES (20 µm AND ABOVE)

A total of 55 1 L grab samples, 32 in the Bay and 23 in the marine sanctuaries, were collected between August 21, 2017, and March 31, 2018. Samples were collected at 13 of the 16 Bay manta trawl sites and six of the 11 marine sanctuary manta trawl sites, along with three duplicates and three field blanks (Tables A-4.5a, A-4.5b, and A-4.6).

At each sampling site, a 1 L pre-cleaned amber glass wide-mouth bottle (ESS, San Leandro CA) was filled with surface water prior to collecting the manta trawl sample. The bottle was attached to a 2 m pole in order to submerge and fill the sample container with undisturbed surface water alongside the research vessel (Figure 4.5). Prior to collecting the sample, the bottle was rinsed with surface water three times by placing the bottle below the surface to fill it, capping the bottle and shaking the bottle before discarding the contents. After rinsing three

times, the field sample was collected, and the bottle was labeled and placed in a cooler. The sample was transported back to SFEI where it was maintained in a refrigerator at 2°C prior to being shipped to the University of Toronto for analysis. In the case where the site also included the collection of a field duplicate, the duplicate was collected immediately after the primary sample using the same techniques.

Three field blanks were collected by filling sample containers with Milli-Q water on the vessel just after a field sample was collected, using the same techniques. Field blanks were placed into coolers and sent back to SFEI with the primary field samples.

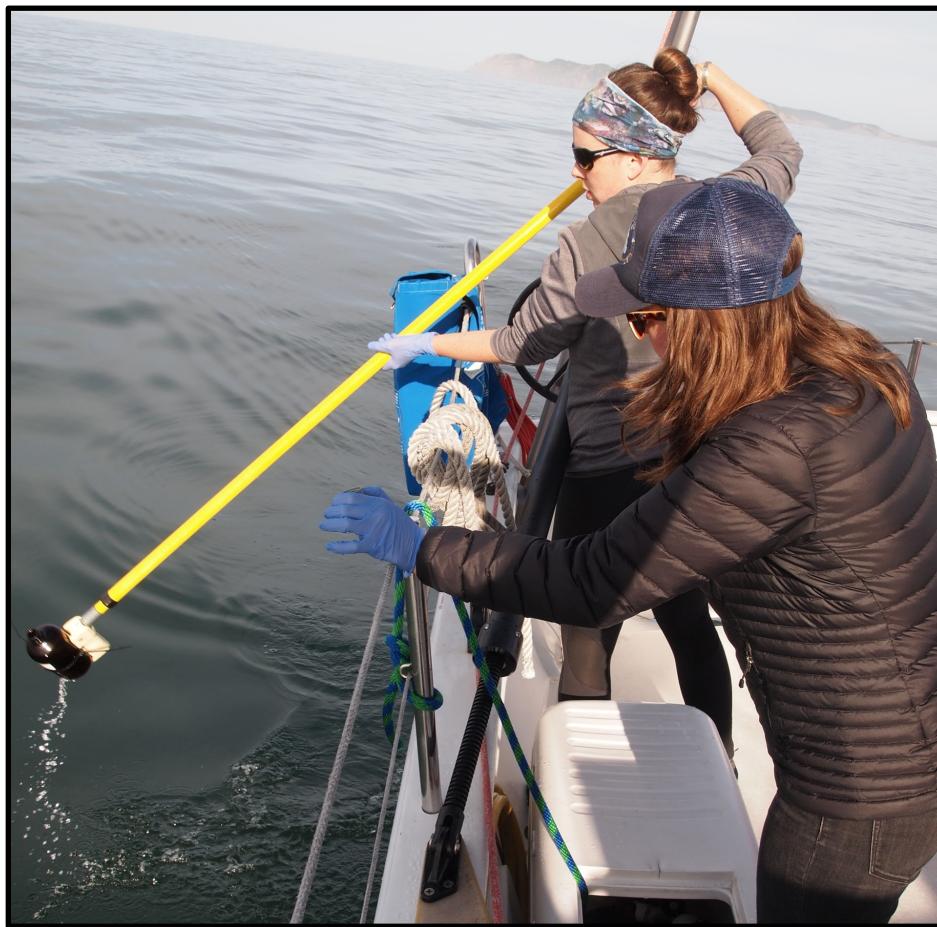


Figure 4.5. Collection of 1 L grab sample from the *Derek M. Baylis*. (Photo by Erika Delemarre.)

PUMP SAMPLES (20 µm TO 5 mm)

To test an additional collection method for smaller microparticle size fractions, a pump-filtration system was designed to collect particles 20 µm and larger at approximately half of the sites included in the project (Sedlak et al., 2017). Up to ten liters of surface water from each site were pumped through a 20 µm polycarbonate filter for analysis. Filter papers were transferred to a glass jar and shipped to University of Toronto for analysis.

Field samples collected using the pump method were compromised due to an unidentified source of contamination. Specifically, the number of fibers found in the field blanks were similar to or higher than the number of fibers counted in the field samples; therefore, the samples were not analyzed and are not reported herein.

Sample extraction and analyses

MANTA TRAWL SAMPLES

In the laboratory, manta trawl samples were dewatered and processed using a digestion step if large quantities of organic material were present. If the digestion step occurred, the samples were dewatered and reconstituted in a pre-filtered 20% potassium hydroxide (KOH) solution, as recommended by Munno et al. (2018), at room temperature for one-week (Munno et al., 2018). KOH is the preferred base for organic material digestion (Dehaut et al., 2016; Lusher et al., 2017). At the end of the one-week period, samples were filtered again through a 110 µm sieve, rinsed with reverse-osmosis-treated (RO) water and transferred into a clean glass jar for microplastic analyses.

Samples that underwent digestion and the remaining samples that did not have large quantities of organic material (that skipped the digestion step) were passed through a 212 µm sieve to capture all particles greater than the 355 µm field sieve pore size. Microparticles were rinsed off the sieve into a clean glass sample jar for further processing using RO water.

Midstream in the analyses of all matrices including surface water, the method was updated by changing the size and number of sieves to separate out particle sizes. In the revised method, a column of stacked sieves (125 µm, 355 µm, 500 µm, and 1 mm) replaced the 212 µm sieve to expedite the extraction process.

After processing, samples were sorted on a clean glass Petri dish under a dissecting microscope. The first ten particles that appeared to be plastic within each color/morphology combination (e.g., blue fragment, white sphere) from each size fraction were transferred to a clean glass petri dish and affixed with double-sided sticky tape. These individual particles were arranged in rows, labeled by particle number (Figure 4.6), imaged, and measured. The remaining particles were counted and recorded based on their size fraction, color, and morphology.

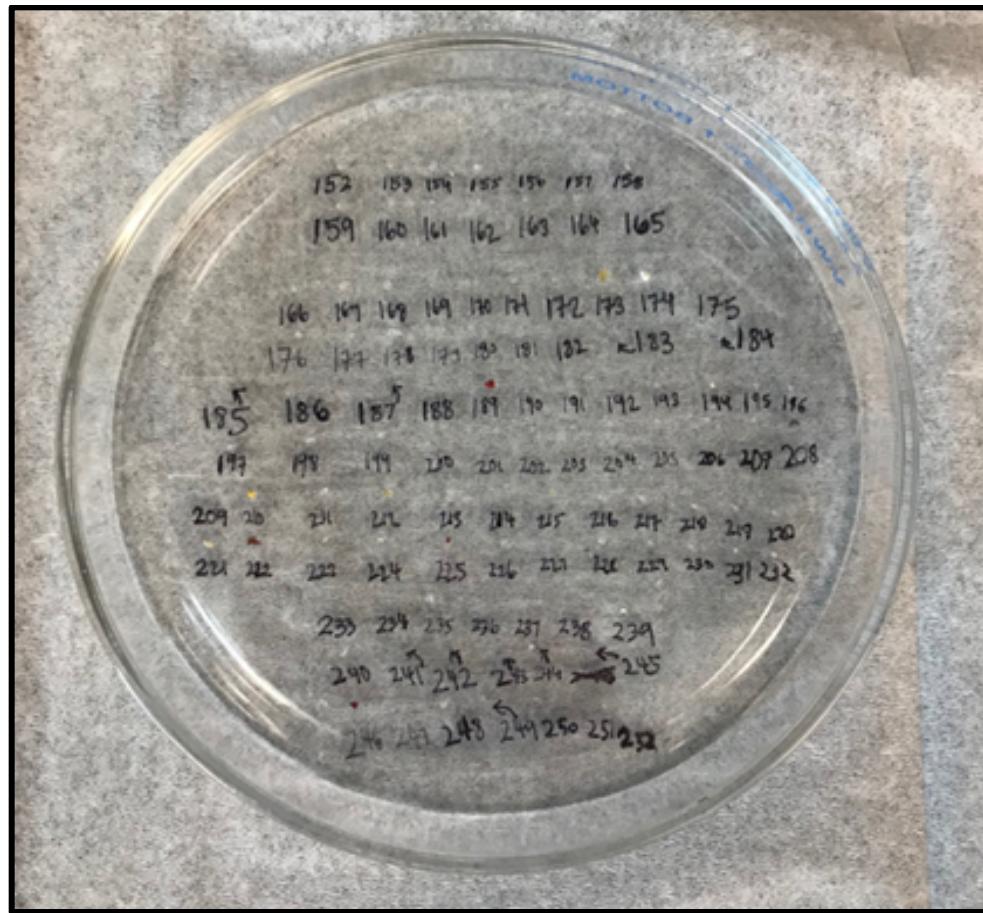


Figure 4.6. An image of a sample prepared for Raman/FTIR spectroscopy. Each number represents an individual particle that was manually extracted from the sample.

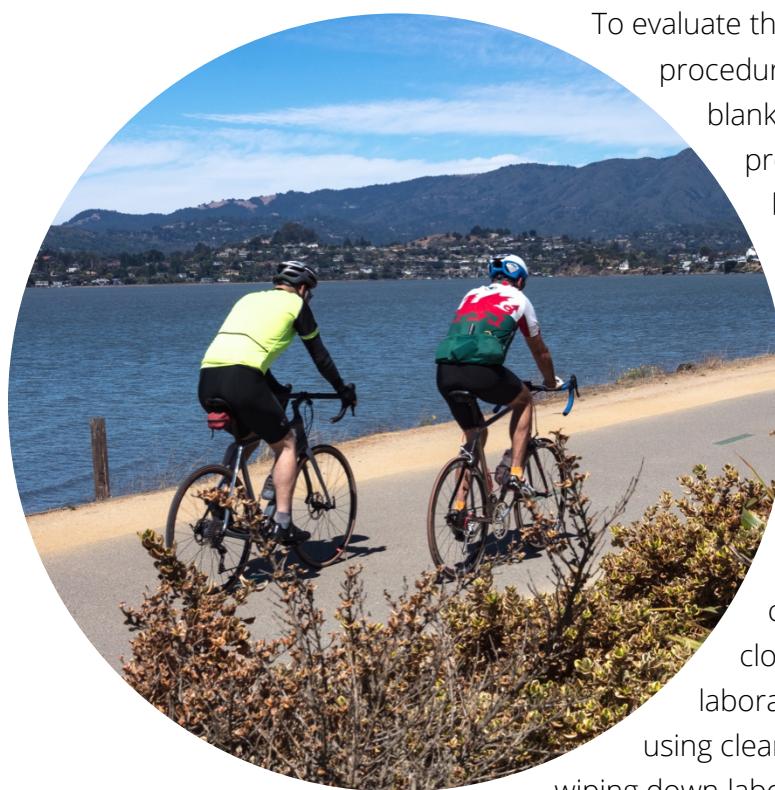
Due to the high numbers of microparticles identified in the samples, only 32% of all particles were measured for length and width. Laboratory analysts measured all particles that underwent spectroscopy. In addition, prior to implementing the expedited sieve method, ten particles of each color and category were measured for fragments, foams, spheres, and films, and 5–10 of each color fiber were measured. After implementing the expedited sieve method, only particles that underwent spectroscopy were measured. In total, 22% of fibers, 38% of fragments, 31% of foams, 72% of films, 78% of spheres, and 76% of fiber bundles (multiple fibers that cannot be disentangled), were measured.

A subset of the particles selected for imaging and measurements were analyzed by Raman/FTIR spectroscopy to determine the chemical composition of the particle. In total, Raman/FTIR spectroscopy was conducted on approximately 13% of the microparticles collected.

The high number of fibers present in the Bay samples made it logically impossible to enumerate all samples for fibers (e.g., two samples contained more than 1,400 fibers each). In

addition, because the shape of fibers (e.g., long and narrow) and the orientation of the fiber dramatically affect whether it is caught by the net or passes through, it is not clear that manta trawl sampling provides a representative sample (Barrows et al., 2017; Covernton et al., 2019). Fibers were analyzed in approximately half (52%) of the samples (Table 4.1).

A wet weather sample (CB9-Manta-11Jan18) contained more organic material and micro- and macroparticles than any other sample. During sampling, the manta trawl passed through a visible tidal front and the entire mesh bag filled with woody debris, leaves, grasses, trash, and other items. This sample filled five sample jars: three 500 mL glass sample jars and two 1.9 L glass mason jars. Four of the five jars (three 500 mL sample jars and one 1.9 L jars) were completely analyzed, but the last 1.9 L jar was only partially analyzed (one quarter of the jar). The results for the partially analyzed jar were multiplied by four.



To evaluate the potential for laboratory procedural contamination, one laboratory blank was run for every ten samples processed; in total, seven laboratory blanks were run, and three of these included analyses of fibers. All laboratory blanks were composed of RO water processed using the same methods as the field samples. Laboratory quality assurance / quality control (QA/QC) involved best practices for reducing procedural contamination (i.e., wearing cotton clothing, installing HEPA filtration in the laboratory, keeping samples covered, using clean glassware and RO water, and wiping down laboratory surfaces each day).

ONE-LITER GRAB SAMPLES

The 1 L sample was filtered through a 20 μm polycarbonate filter. The particles captured on the filter were then sorted by color/category combination (e.g., black fiber, blue fragment) using a dissecting microscope. For each color/category combination, 10 particles were randomly selected and analyzed using Raman spectroscopy. Microparticles contained in laboratory blanks and field blanks did not undergo spectroscopy.

Statistical analysis, treatment of blanks, and methods used to estimate microplastic abundance

STATISTICS

Statistical analyses were conducted in R (R Core, 2018) using a series of non-parametric tests. Non-parametric tests were used due to the presence of non-normal, skewed distributions in surface sample results that were better represented by medians as a measure of central tendency in most of the hypothesis tests. Other considerations were the presence of outliers, and in some situations, small sample sizes. Mann Whitney U tests were used for two sample comparisons when results were not paired, such as evaluating whether microparticle abundances in the Bay vs. marine sanctuaries were different. Wilcoxon signed-rank tests were used for two-sample paired comparisons, such as comparing microparticle abundances in dry season vs. wet season in the Bay and sanctuaries. If there were repeat samples within the same season at a site, the values were averaged for the statistical analyses. Additional samples were collected at two sites, CB5 (2 samples taken during the dry season) and MBNMS29 (3 samples taken during the wet season). Kruskal-Wallis tests were performed when there were multiple comparisons (typically with post-hoc Dunn's Tests for pairwise comparisons) such as evaluating whether Bay, sanctuary, and blank samples were different or whether microparticle abundances in each Bay subembayment were different. The threshold for statistical significance was $p < 0.05$. Except for paired tests, in the cases where more than one sample was collected at a site within a season (not including duplicates), the samples were treated as separate data points.

BLANKS AND BACKGROUND CONTAMINATION

In general, field samples were not blank corrected (i.e., blank counts were not subtracted from the field sample counts) for this report to allow other scientists to view raw data.

A close examination of particles in the field blanks and field samples, however, led to the identification of intermittent procedural contamination likely associated with materials onboard the research vessels we used to collect samples. Specifically, one of the field blanks collected in San Pablo Bay during the first day of sampling had a particularly high fiber count, and 36% of the fibers were orange; a similar observation was noted in a field sample collected on a different day. A likely source of this intermittent contamination was from orange personal flotation devices worn on the vessels during sample collection. Additionally, a black mat onboard the *Derek M. Baylis* was identified as the source of thick, curly black fibers that were observed in some field blanks and samples. The black mat was onboard during the first three days of field sampling (August 21–23, 2017), and was subsequently removed. All orange fibers were not included in counts for samples and blanks. Black curly fibers were also removed from

microparticle counts for samples collected during the initial sampling expeditions in August 2017.

Field data for which the particle counts (by morphology) were less than the average of the field and laboratory blanks plus two times the standard deviation were qualified, indicating potential influence of background contamination. All field and laboratory blank data are reported with the field samples so individual readers can make their own inferences regarding the data.

METHODS USED TO ESTIMATE MICROPARTICLE ABUNDANCE AND CONCENTRATION

The reading on the digital current flowmeter (General Oceanics, Inc. Model 2030R) was recorded at the beginning and end of each trawl, following standard protocols (General Oceanics, 2018; GESAMP, 2019; Lippiatt et al., 2013). The difference in the value was used to calculate the distance traveled, based on the following equations:

$$\text{Distance in meters} = \frac{\text{Difference in Flowmeter Value (final - initial)} \times \text{Rotor Constant}}{999,999}$$

The Standard Speed Rotor Constant (26,873) is used with this model.

The distance traveled multiplied by the width of the trawl (0.61 m) provided the surface area sampled, allowing microparticle abundance per square kilometer (particles/km²) to be calculated. The distance traveled multiplied by the width of the trawl and the height of the trawl that was submerged below the sea surface (0.095 m) was used to calculate the volume of water sampled and determine the concentration of microparticles (particles/L).

Results

Quality assurance results

BACKGROUND CONTAMINATION: FIELD AND LABORATORY BLANKS

Manta trawl samples

Seven laboratory blanks were collected during manta trawl sample analysis; fibers were enumerated in three of these blanks, and the other particle morphologies (fragment, foam, sphere, film) were enumerated in all seven. Only fibers were observed in laboratory blanks. One to two fibers were detected in the three blanks in which fibers were analyzed (Table A-4.4). Of the five fibers detected in laboratory blanks, Raman/FTIR spectroscopy identified one as plastic (acrylic), one as anthropogenic cellulosic, two as cellulosic, and one as cotton. The anthropogenic cellulosic fiber indicates that it was dyed and therefore man-made.

Eight manta trawl field blanks were collected; fibers (including fiber bundles) were analyzed in four of the blanks. Fragment, foam, sphere, and film particles were analyzed in all of the field blanks. Fibers were a significant portion of the particles detected in field blanks as compared to other morphologies, with an average of 42 fibers per blank ($n = 4$) and a range from 12 to 66 in number. Fragments were the next most frequently detected shape in field blanks, with an average fragment count of 1.9 ($n = 8$) and a range of zero to six.

Based on the field and laboratory blanks, the average and standard deviations of fibers, fiber bundles, and fragments found in the blanks were 24.7 ± 26.7 , 2.1 ± 3.1 , and 1.0 ± 1.7 , respectively. The average was calculated using the sum of the laboratory and field blanks. A single foam microparticle was detected in one field blank. No spheres or films were found in the field blanks. Foam, sphere, and film particles were not identified in any laboratory blanks.

Raman/FTIR spectroscopy of the 56 fibers and fiber bundles in field blanks indicated a total of 55% were plastic, made up of polyester (38%), polyethylene (5%), polypropylene (4%), acrylic (2%), nylon (2%), polytetrafluoroethylene (PTFE; 2%), and polyurethane (2%). An additional 15% were identified as anthropogenic unknown and unknown, which may or may not be made of plastic. The remaining 30% were identified as natural fibers.

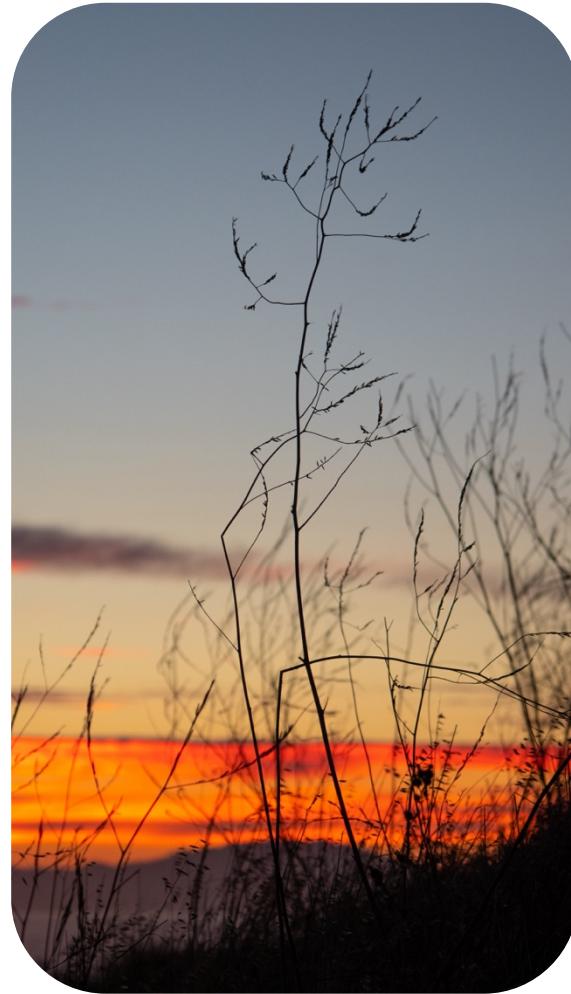
Raman/FTIR spectroscopy conducted on a subset of microparticle fragments (14 fragments) showed that 58% of the fragments were plastic, including polyethylene (29%), polypropylene (15%), polyethylene terephthalate (7%), and nylon (7%). After plastic, 21% of the fragments were identified as anthropogenic unknown, which may or may not be plastic. The rest of the fragments were evenly split (21%) between anthropogenic unknown, which may or may not be plastic, and natural fragments.

The field and laboratory blanks indicated that sample contamination can occur during collection, processing, and analysis. The variability in blank contamination suggests caution must be exercised in interpreting data. We did not blank correct our sample results, but values were qualified when they were below a conservative threshold for data qualification determined by taking the average of the laboratory and field blanks plus two times the standard deviation. A conservative threshold was derived for each morphology, which is appropriate because the blanks had different levels of contamination for each morphology. The thresholds calculated for each particle morphology were 78.2 for fibers, 8.3 for fiber bundles, 4.3 for fragments, and 0.6 for foams, respectively. All wet season fiber counts in the field samples (collected in Bay and marine sanctuaries) were above the threshold, while only 50% of dry season samples were above the threshold. Except for one sample, all fragment counts in Bay samples collected during dry and wet season samples were above the threshold, while 68% of the fragment counts in the sanctuary samples (wet and dry) were above the threshold. All foam counts were above the threshold.

One-liter grab samples

Total microparticle counts (including fibers) in the laboratory blanks ($n = 6$) ranged between one and ten, and in the field blanks ($n = 3$) between one and eight. The combined average microparticle count in laboratory and field blanks ($n = 9$) was 4.4, with a standard deviation of 3.5. Fibers were found to be 85% of the microparticles, fragments 10%, films 5%, and no foams or spheres were observed. Spectroscopy was not conducted.

To assess whether external contamination of field samples was significant, a threshold for data qualification, determined by taking the average of the laboratory and field blanks plus two times the standard deviation, was calculated for each particle morphology (i.e., 10.2 for fibers, 2.5 for fragments, and 1.1 for films; no foam or sphere particles were detected; Table A-4.6). These thresholds were used to evaluate the field sample counts (Tables A-4.5a and A-4.5b). Microparticle counts in field samples ranged between one and 42, with an



average of 6.2 microparticles/sample. Three 1 L samples (5% of all samples) contained fiber counts greater than the respective threshold (two collected in the Bay and one in the marine sanctuaries, all during the wet season). Five 1 L samples (9% of all samples) contained fragment counts greater than the respective threshold (one collected in the Bay and four in the marine sanctuaries, all during the wet season). Six 1 L samples (11%) contained a greater number of films than the respective threshold (three each in the Bay and marine sanctuaries, collected during both wet and dry seasons). One sample contained a foam particle, a morphology not observed in blanks.

Because so few 1 L grab samples were above these thresholds, a robust analysis of results is not possible. Improvements to the sample collection methodology are indicated, including sampling greater volumes of surface water to overcome background thresholds. Further controls to prevent background contamination in both the field and the laboratory setting are recommended.

PRECISION AND VARIABILITY: FIELD DUPLICATES

Manta trawl samples

Seven manta trawl duplicate samples were collected: five in the Bay and two in the marine sanctuaries (Table 4.1). Fibers were analyzed in two of the duplicate sample pairs, and other morphologies were analyzed in all seven (Figures 4.7 and 4.8).



Figure 4.7. Primary field and corresponding field duplicate manta trawl samples analyzed for fragments, foams, spheres, and films (fibers not included).

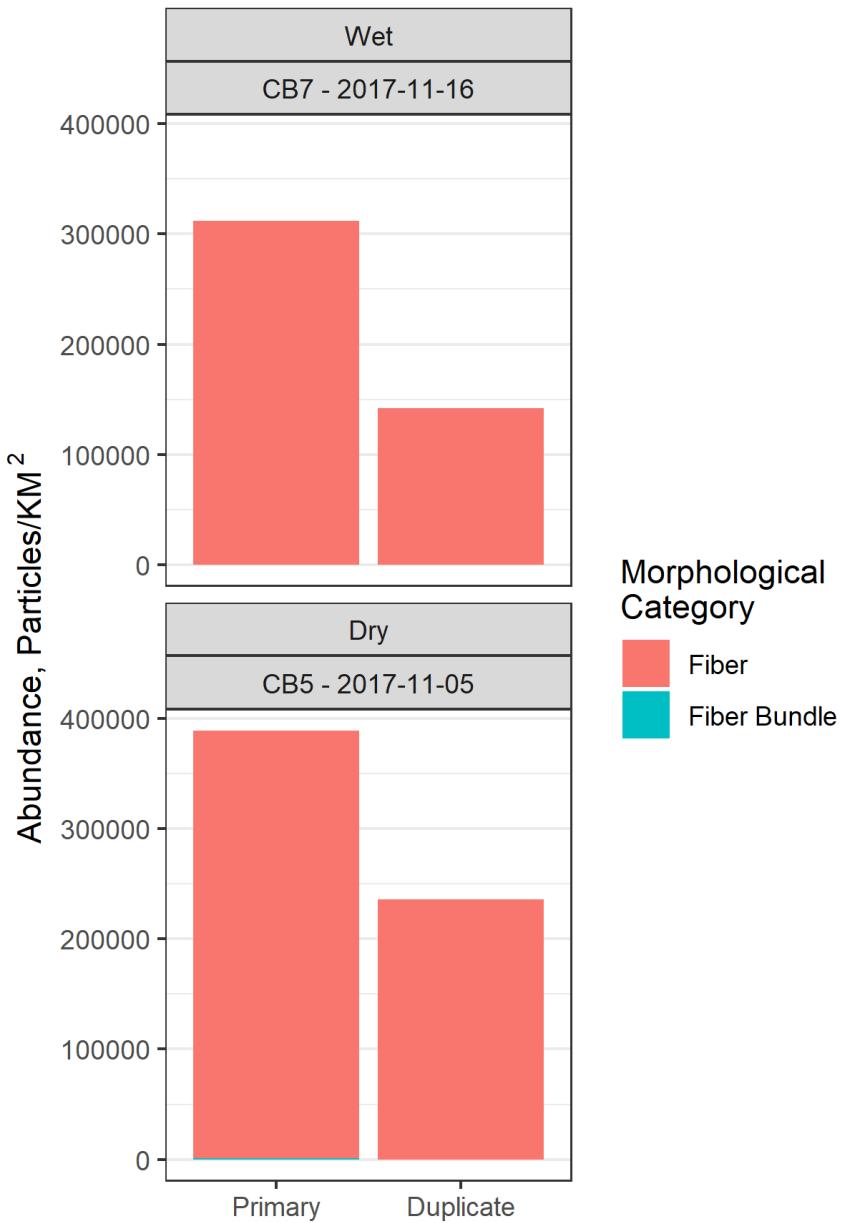


Figure 4.8. Primary field samples and corresponding field duplicates for manta trawl samples at CB5 and CB7 (fibers and fiber bundles only). Fiber bundle concentrations are much lower than fibers and are not always visible at this scale.

The relative percent difference (RPD) for total particle abundances (including fragments, foams, spheres, and films) ranged from 2% to 105%, indicating that collecting reproducible manta trawl samples can be challenging. Samples from the marine sanctuaries had RPDs of less than 35%, indicating good reproducibility and likely reflecting the well-mixed oceanic water in these areas.

In contrast, RPDs from the Bay were generally much higher and likely reflect the more heterogeneous nature of Bay water. The duplicate sample was typically collected between 45 minutes to one hour after collection of the primary sample, the time it takes for the vessel to reposition at the latitude and longitude of the initial sample collection starting point. These values provide some indication of the variation associated with trawl samples.

Comparing the abundance of individual morphologies in the paired samples indicated considerable variability, with very few of the morphologies showing similar abundances. The marine sanctuary samples showed less variation than the Bay samples.



Microparticle occurrence and morphology

FRAGMENTS, FOAMS, SPHERES, AND FILMS: ABUNDANCE, CONCENTRATION, AND COMPOSITION IN MANTA TRAWL SAMPLES

Abundance of fragments, foams, spheres, and films

Microparticle abundance was calculated for all manta trawl samples. For the entire dataset, total abundance (excluding fibers) ranged greatly, from zero microparticles/km² (at CBNMS24, collected September 13, 2017, during dry weather) to 6,200,000 microparticles/km² (at CB9, collected January 11, 2018, during wet weather; Tables A-4.3a, A-4.3b, A-4.3c, and A-4.3d). These abundances correspond to a concentration range of zero to 0.066 particles/L (see Tables A-4.3e and A-4.3f for additional concentration data).

Tables 4.2, 4.3, 4.4, and 4.5 show microparticle abundance (excluding fibers) by location (Bay vs. marine sanctuaries) and season (dry vs. wet). The average abundance varied by several

orders of magnitude in the Bay, while abundance only varied by one order of magnitude in the marine sanctuaries.

Table 4.2. Microparticle abundance for San Francisco Bay samples collected during the dry season (n = 18).

<i>Morphology</i>	<i>Abundance (particles/km²)</i>			
	<i>Min</i>	<i>Max</i>	<i>Median</i>	<i>Mean</i>
Fragment*	1,300	240,000	15,000	37,000
Foam	0	67,000	3,100	10,000
Sphere	0	9,100	480	1,600
Film	0	20,000	1,800	4,400
Total	2,400	280,000	27,000	53,000

*One sample had a fragment count below the data qualification threshold, indicating potential for significant influence of background contamination.

Table 4.3. Microparticle abundance for San Francisco Bay samples collected during the wet season (n = 16).

<i>Morphology</i>	<i>Abundance (particles/km²)</i>			
	<i>Min</i>	<i>Max</i>	<i>Median</i>	<i>Mean</i>
Fragment	7,600	5,000,000	40,000	400,000
Foam	0	790,000	13,000	75,000
Sphere	0	300,000	3,300	26,000
Film	750	200,000	5,500	20,000
Total	9,900	6,200,000	98,000	520,000

Table 4.4. Microparticle abundance for National Marine Sanctuaries samples collected during the dry season (n = 11).

<i>Morphology</i>	<i>Abundance (particles/km²)</i>			
	<i>Min</i>	<i>Max</i>	<i>Median</i>	<i>Mean</i>
Fragment*	0	12,000	2,400	4,100
Foam	0	3,100	0	580
Sphere	0	1,600	0	360
Film	0	6,200	0	1,300
Total	0	16,000	4,100	6,300

*Seven samples had fragment counts below the data qualification threshold, indicating potential for significant influence of background contamination.

Table 4.5. Microparticle abundance for National Marine Sanctuaries samples collected during the wet season ($n = 13$; two additional samples were collected at the Golden Gate site compared to dry weather sites to provide additional information for modeling purposes).

<i>Morphology</i>	<i>Abundance (particles/km²)</i>			
	<i>Min</i>	<i>Max</i>	<i>Median</i>	<i>Mean</i>
Fragment*	1,600	27,000	7,300	10,000
Foam	0	19,000	620	2,400
Sphere	0	3,800	620	760
Film	0	5,700	690	1,100
Total	2,600	38,000	9,400	15,000

*Two samples had fragment counts below the data qualification threshold, indicating potential for significant influence of background contamination.

Microparticle abundance (excluding fibers) within the Bay was higher than in the marine sanctuaries (i.e., medians of 44,000 and 7,900 particle/km², respectively; one-sided Mann Whitney U Test, $U = 706$, $p = 2.6 \times 10^{-7}$). Bay and sanctuary samples were also statistically different than field blanks (Kruskal-Wallis two-sided test, $p = 2.36 \times 10^{-10}$; Dunn's pairwise tests, Bay: $z = 4.87$, $p = 1.12 \times 10^{-6}$; marine sanctuary: $z = 2.08$, $p = 0.0375$).

When the individual morphologies (i.e., fragment, foam, sphere, and film) were compared, abundances within the Bay were also statistically different from abundances within the marine sanctuaries (all p -values < 0.013 ; Figure 4.9). The percentage of samples with counts of fragments above the potential threshold (average of the fragments in blanks plus two times the standard deviation) was 97% for the Bay and 65% for the sanctuaries. Counts for other morphologies were all above the respective thresholds for all samples.

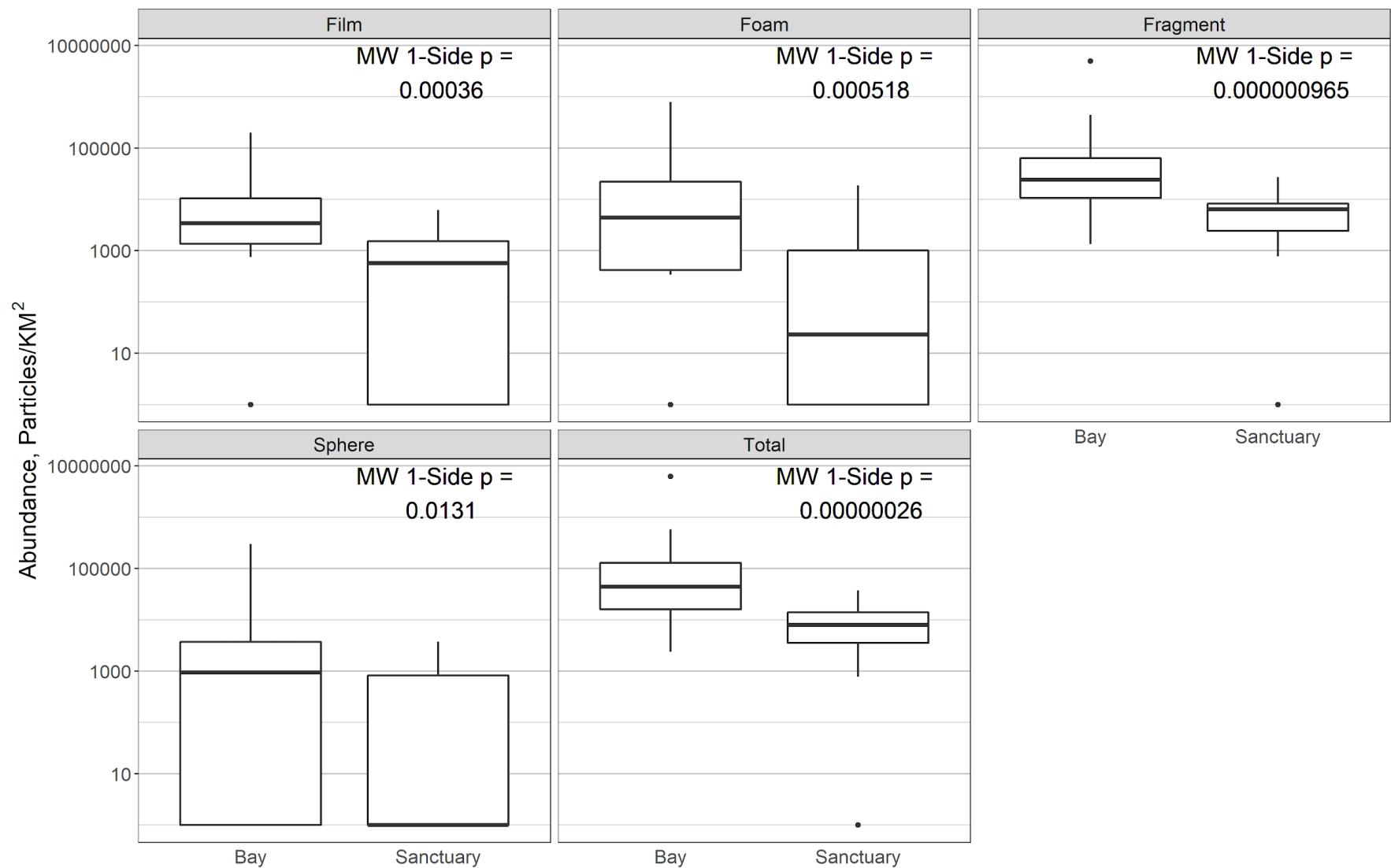


Figure 4.9. Abundances for film, foam, fragment and sphere particles in the Bay and marine sanctuaries. Statistical differences were assessed by one-sided Mann-Whitney U tests.

Within the Bay, there were no significant differences between microparticle abundances (excluding fibers) between the subembayments (Kruskal-Wallis two-sided test, chi-squared = 3.80, degrees of freedom = 3, $p = 0.284$). Additionally, abundances for each morphology (fragment, foam, sphere, and film) were not significantly different.

Across all sites (Bay and sanctuary) microparticle abundances were significantly higher during the wet season than the dry season (one-sided Mann Whitney U test, $U = 558$, $p = 0.0162$). The median microparticle abundance (excluding fibers) was 27,000 microparticles/km² for the wet weather samples, ranging from 2,600 to 6,200,000 microparticles/km². The median microparticle abundance for the dry weather samples was 14,000 microparticles/km², ranging from zero to 280,000 microparticles/km².

In addition to this broader comparison of wet and dry season abundances (excluding fibers), seasonal differences within the Bay were found to be statistically significant (Wilcoxon signed-rank one-sided test, $V = 115$, $p = 0.0066$), while there was no significant difference between seasons in the marine sanctuaries (Wilcoxon signed-rank one-sided test, $V = 46$, $p = 0.14$ for the marine sanctuaries).

Composition of fragments, foams, spheres, and films

Among the four particle types analyzed in all manta trawl samples, fragments were the major morphology observed, accounting for an average of 73% of all microparticles, followed by foams (17%), spheres (5%), and films (5%). Of the 7,952 fragments enumerated, 979 particles were further characterized using Raman/FTIR spectroscopy; 87% of the fragments were identified as plastic, with a majority identified as polyethylene (47%) and polypropylene (25%; Figure 4.10). Of the fragments measured, they ranged in size from 0.15 mm to larger than 5 mm, with the majority between 0.5 and 2 mm in size (Figure 4.11). Most fragments were white (31%) or clear (33%).

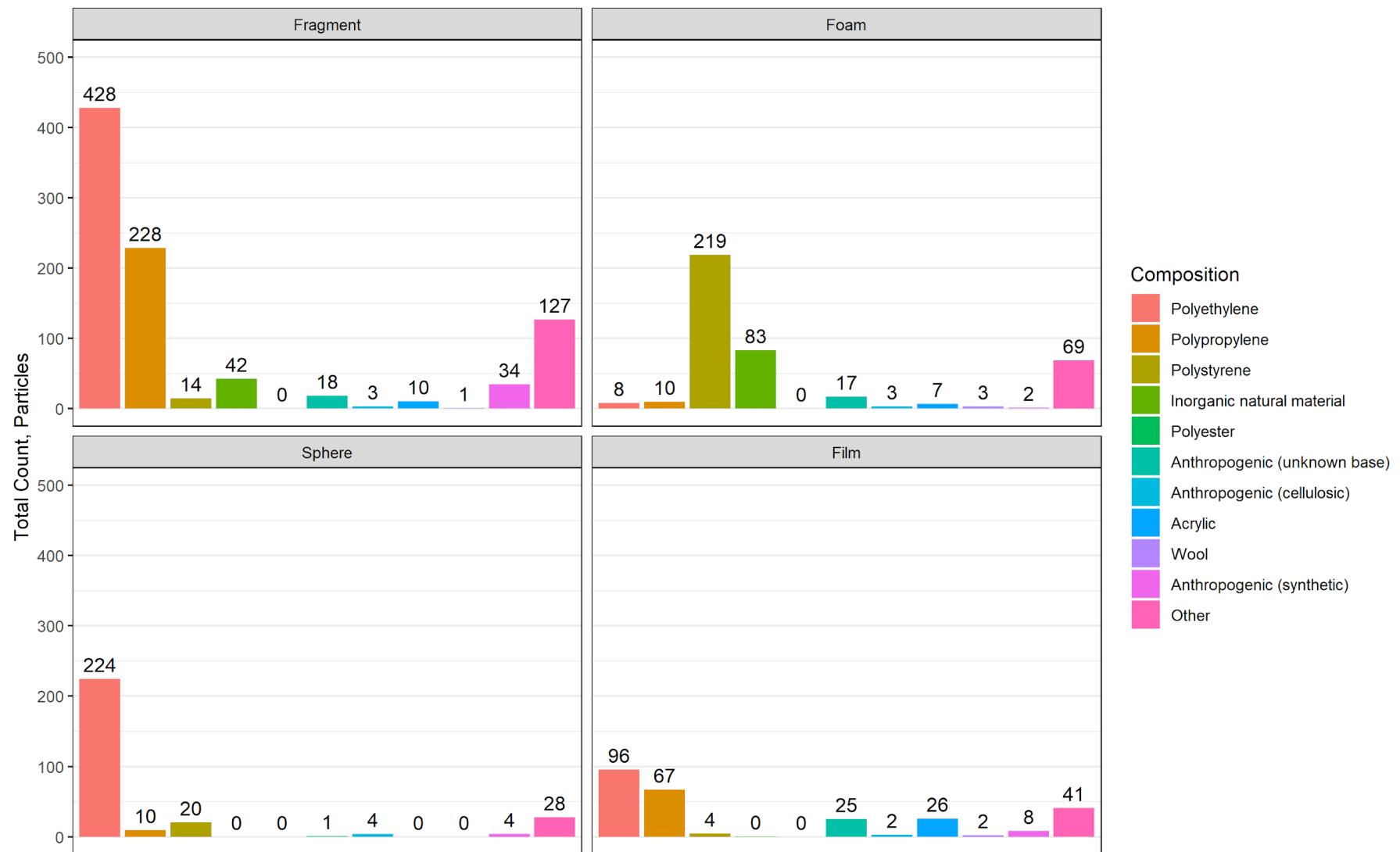


Figure 4.10. Composition of the subset of microparticles (excluding fibers) subjected to spectroscopy for the manta trawl dataset. Fragments were predominantly made of polyethylene (428); foam pieces were primarily made of polystyrene (219); spheres were almost exclusively made of polyethylene (224); and film exhibited a wider distribution of material with polyethylene (96) identified most frequently. The “Other” category includes plastic polymers and non-plastic particles.

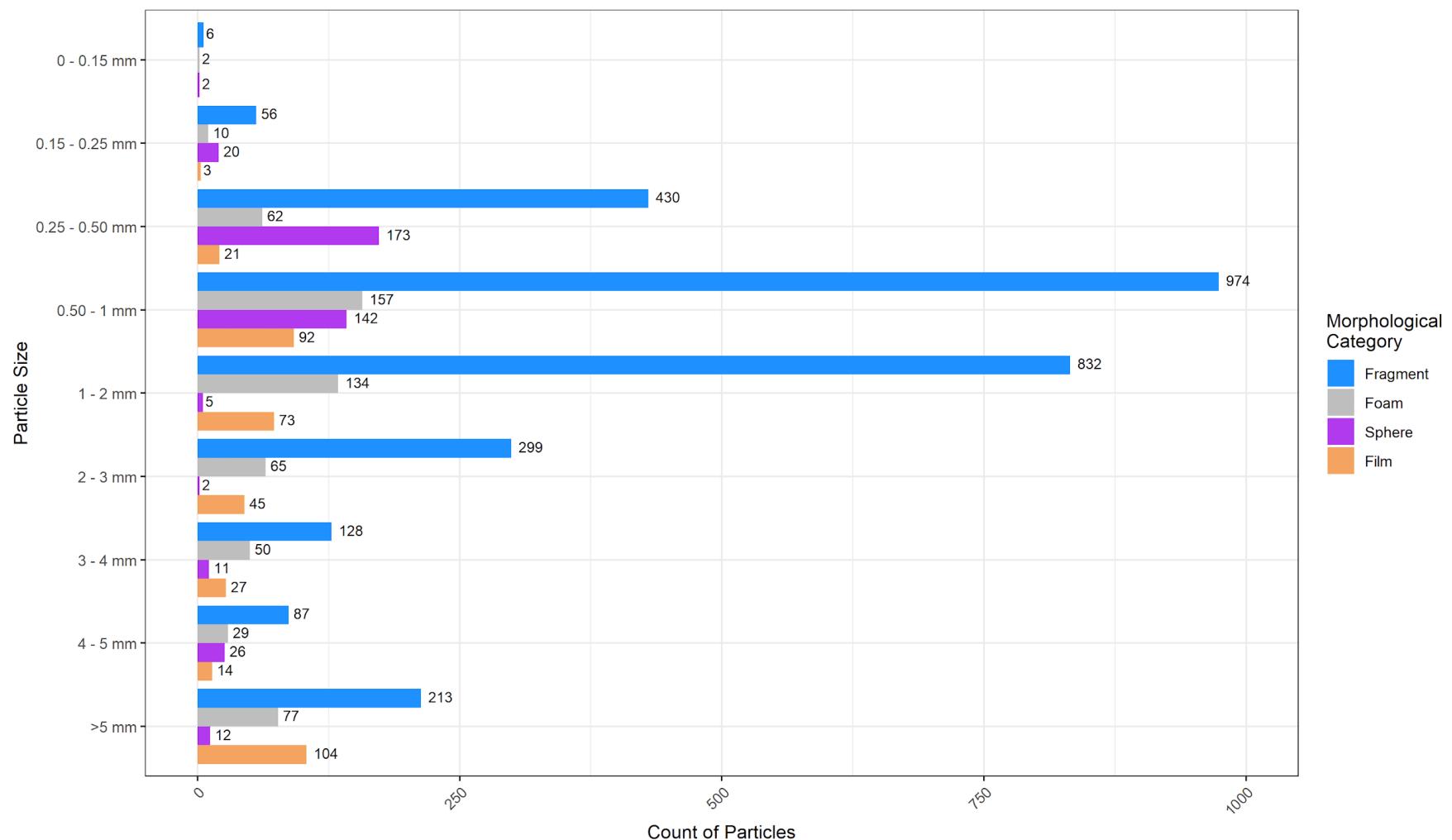


Figure 4.11. Distribution of particle sizes by morphology for particles measured in manta trawl samples (excluding fibers). Most fragments identified were 0.50–1 mm in size (974); most foam particles were 0.50–1 mm in size (157); most spheres were 0.25–0.50 mm in size (173); and most film particles were greater than 5 mm in size (104).



Of the 1,882 foam particles enumerated, 473 were further characterized using Raman/FTIR spectroscopy; 68% were identified as plastic with the majority being polystyrene (53%). Inorganic natural materials represented 19% of the particles. From the particles measured, the majority were 0.5–2 mm in size (Figure 4.11) and were white in color (92%).

Of the 498 spheres enumerated, 317 were further characterized using Raman/FTIR spectroscopy; 97% were identified as plastic with the majority being polyethylene (77%) and polystyrene (7%). From the spheres measured, two distinct size classes were observed, with most falling into either a smaller size range of 0.15–1 mm, or a larger size range of between 3 mm to more than 5 mm (Figure 4.11). Most of the spheres identified were either white (22%) or blue (20%).

Of the 515 films enumerated, 298 were further characterized using Raman/FTIR spectroscopy; 83% were identified as plastic with the majority being polyethylene (36%), polypropylene (23%), acrylic (10%) and anthropogenic synthetic (3%). Of the films measured, the majority were between 0.5 mm and 2 mm, and 27% were longer than 5 mm in length (Figure 4.11).

FIBERS: ABUNDANCE, CONCENTRATION, AND COMPOSITION IN MANTA TRAWL SAMPLES

Abundance and concentration of fibers

In the manta trawl samples where fibers were counted ($n = 32$), fibers were the dominant morphology, making up 74% of all microparticles; in contrast, fragments made up only 18%, followed by foam (5%), film (2%) and spheres (1%). Overall, the average total abundance of all microparticles, including fibers, in this subset of samples was 270,000 microparticles/km², ranging from 18,000 to 1,800,000 microparticles/km² (Tables A-4.3b and A-4.3d). These abundances correspond to an average concentration of 0.0028 particles/L with a range of 0.0002 to 0.020 particles/L (see Tables A-4.3e and A-4.3f for additional concentration data). The average fiber abundance was 210,000 fibers/km², and ranged from 16,000 to 1,700,000 fibers/km² (Table 4.7; Figure 4.12). The average fiber concentration was 0.002 fibers/L, ranging from 0.0002 to 0.02 fibers/L (Table 4.7).

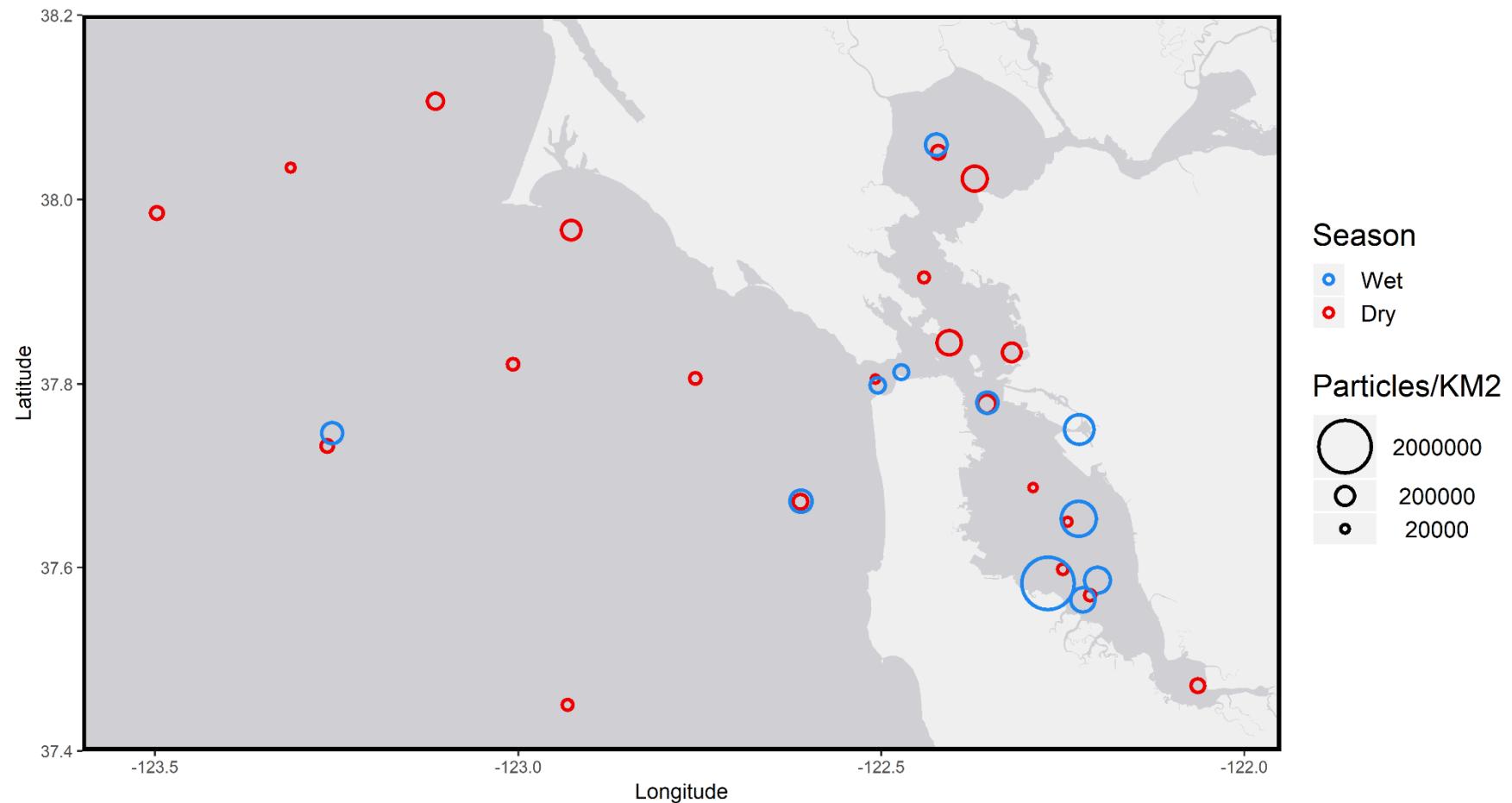


Figure 4.12. Fiber abundance for the subset of manta trawl samples analyzed for fibers.

Fiber abundances were significantly different between the Bay and marine sanctuaries (one-sided Mann Whitney U test, $U = 171$, $p = 0.0456$). The median fiber abundance in the Bay was 198,000 fibers/km², ranging from 17,000 to 1,700,000 fibers/km². The median fiber abundance in the marine sanctuaries was 77,000 fibers/km², ranging from 16,000 to 260,000 fibers/km² (Tables 4.6, A-4.3b and A-4.3d).

Table 4.6. Fiber abundance* in the Bay and marine sanctuaries during wet and dry season sampling.

	Season	Fiber Abundance (fibers/km ²)			
		Min	Max	Median	Mean
Bay	Dry (n = 11)	17,000	330,000	75,000	120,000
	Wet (n = 7)	230,000	1,700,000	360,000	580,000
Sanctuary	Dry (n = 10)	16,000	180,000	53,000	68,000
	Wet (n = 4)	92,000	260,000	160,000	170,000

Table 4.7. Fiber concentrations* in the Bay and marine sanctuaries during wet and dry season sampling.

	Season	Fiber Concentration (fibers/L)			
		Min	Max	Median	Mean
Bay	Dry (n = 11)	0.0002	0.0035	0.0008	0.0012
	Wet (n = 7)	0.0024	0.018	0.0038	0.0061
Sanctuary	Dry (n = 10)	0.0002	0.0019	0.0006	0.0007
	Wet (n = 4)	0.0010	0.0027	0.0017	0.0018

*Eighty percent of Bay samples above the data qualification threshold; 50% of sanctuary samples above threshold.

To assess the potential influence of procedural contamination on fiber abundance and concentration, field and laboratory blanks were compared to Bay and sanctuary samples using a Kruskal Wallis two-sided test (chi-squared = 15.7, degrees of freedom = 3, $p = 0.001$). Relative to field blanks, fiber counts in Bay samples were significantly different (unadjusted Dunn's test $Z = 2.38$, $p = 0.017$), while marine sanctuary samples were not (unadjusted Dunn's test $Z = 1.04$, $p = 0.295$). Both Bay and marine sanctuary samples were significantly different from the blanks (unadjusted Dunn's test Bay: $Z = 3.43$, $p < 0.001$; marine sanctuaries: $Z = 2.23$, $p = 0.026$).

Fiber abundances in wet and dry seasons for the Bay and sanctuaries combined dataset were significantly different (one-sided Mann Whitney U test, $U = 208$, $p < 0.001$). The median fiber abundance during the dry season was 58,000 fibers/km², while the median fiber abundance during the wet season was 260,000 fibers/km². Significant differences between wet and dry seasons within the Bay and within the marine sanctuaries were also found (one-sided Mann-Whitney U test Bay: $U = 71$, $p < 0.001$; marine sanctuaries: $U = 35$, $p = 0.018$). The average fiber abundance for wet weather samples collected in the Bay was more than three times the average wet weather fiber abundance in the sanctuary samples (Table 4.6).

This project also identified fiber bundles (a number of fibers that cannot be disentangled), making up less than 1% of the total particle counts. Fiber bundle abundances and concentrations were not calculated because they represented such a small portion of the total particle count; however, the composition of fiber bundles is discussed with fiber composition.

Composition of fibers

The fibers (including fibers and fiber bundles) analyzed by Raman/FTIR spectroscopy were made of a range of polymer types, including polyester (24%), anthropogenic cellulosic (14%), wool (10%), acrylic (10%), polypropylene (9%), polyethylene (3%), anthropogenic synthetic (1%), polystyrene (1%), and organic natural material (less than 2%; Figure 4-13). Overall, at least 53% of the fibers that underwent spectroscopy were identified as plastic. Additional fibers and fiber bundles were classified as anthropogenic unknown (14%) when identification of the underlying material could not be identified because the dye on the fibers masked the spectrum. Some were also classified as unknown, meaning the spectra did not match any of the spectra in the library. It is possible that a portion of the anthropogenic unknown and unknown fibers and fiber bundles are plastic. Most of the fibers and fiber bundles were 1–2 mm in length (Figure 4.14). More than half the fibers and fiber bundles (64%) were black, blue, or dark blue in color.

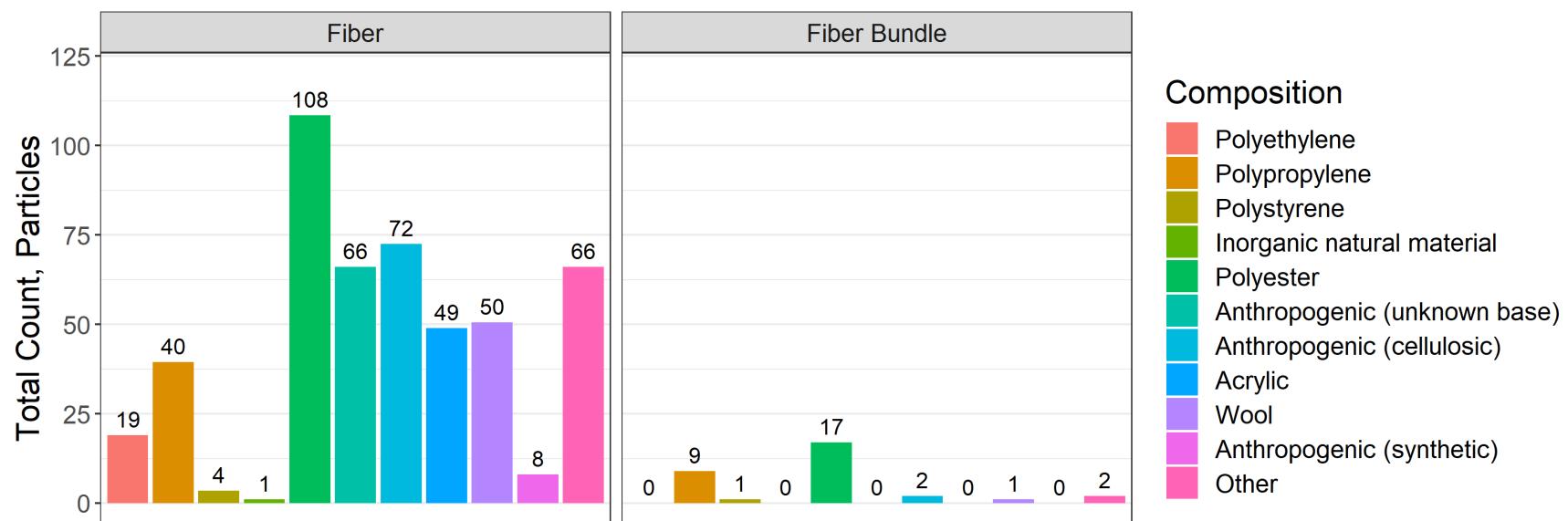


Figure 4.13. Composition of fibers and fiber bundles analyzed in manta trawl subsample. The “Other” category includes plastic polymers and non-plastic particles.

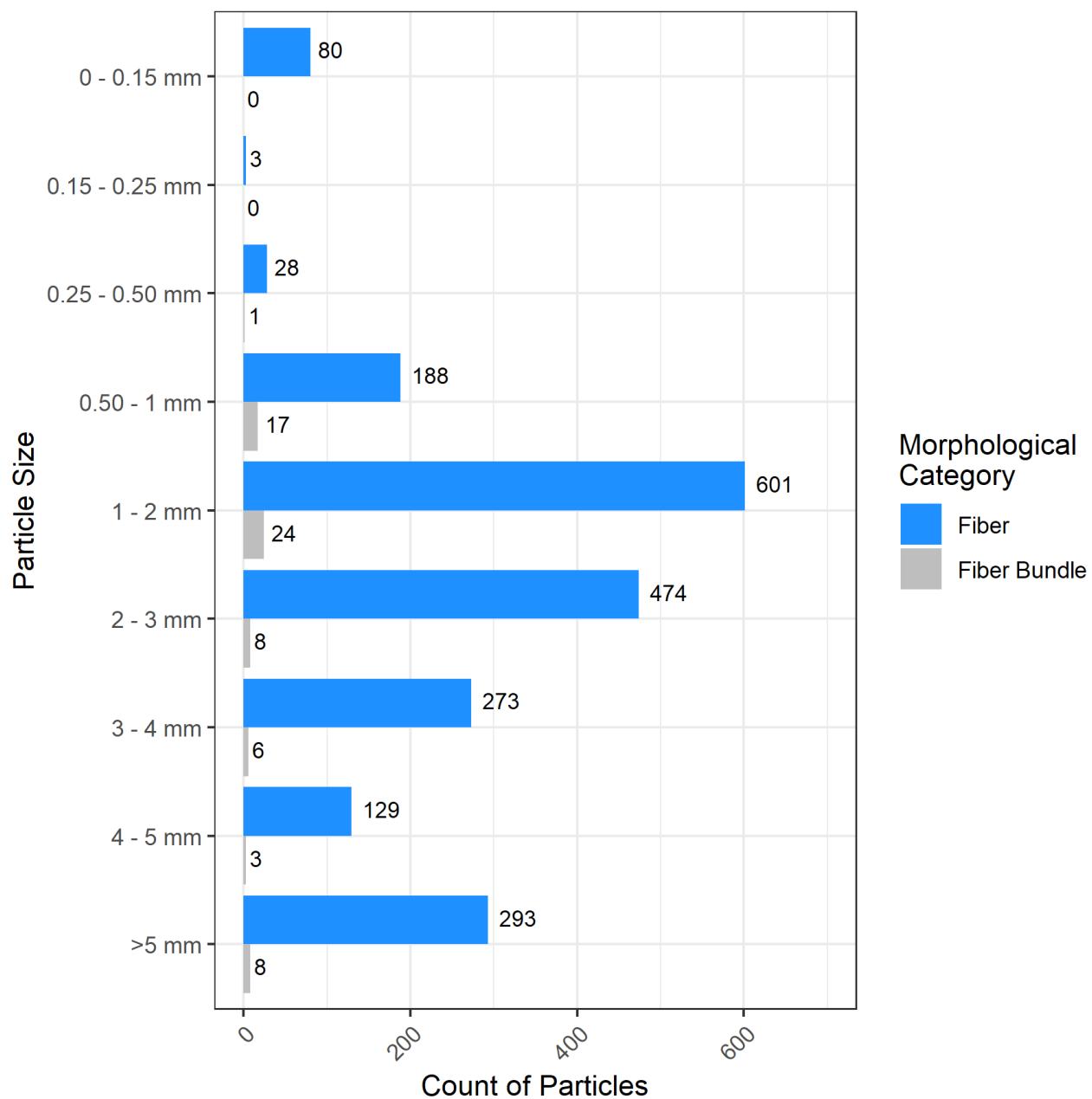


Figure 4.14. Distribution of particle sizes for fibers and fiber bundles measured in manta trawl samples.

Discussion

Surface water sampling techniques must be improved to quantify fibers

A lack of standardized methods for the collection, extraction, and analysis of microplastics has been widely acknowledged as a significant challenge (GESAMP, 2019; Hidalgo-Ruz et al., 2012; Woodall et al., 2015). In terms of sample collection, the manta trawl has been considered the signature piece of equipment to collect microplastics in surface waters, based on historical precedent (Eriksen et al., 2014; GESAMP, 2019; Law et al., 2014).

More recently, as microplastics research expands to smaller particles, other field methods such as grab samples have been deployed (Barrows et al., 2018; Miller et al., 2017; Rochman, 2018). Some studies evaluating multiple methods have suggested that the manta trawl sampling method could be underestimating the number of particles, specifically fibers (Barrows et al., 2017; Covernton et al., 2019; Kang et al., 2015). This is likely a function of the manta net mesh size, which may allow particles smaller in two dimensions, like fibers, to escape.

A goal of this study was to evaluate some of these different sampling techniques against the manta trawl to determine the efficacy of the methods and the types of particles captured. As described above, relatively few 1 L grab samples were above the data qualification thresholds determined for these samples. In future studies, to assure that grab samples are not strongly influenced by background contamination, it is recommended that larger sample volumes be collected.

Fibers were the major particle type observed in field and laboratory blanks for both the manta trawl and 1 L grab samples, which attests to their ubiquitous presence in the environment. Airborne deposition has been shown to be a significant pathway, particularly for fibers (Cai et al., 2017; Dris et al., 2016, 2015). Sample contamination by airborne fibers can be significant (SAPEA, 2019; Silva et al., 2018). Development of standard methods for the collection and analysis of microplastics that mitigate the contamination of samples with airborne fibers during collection and analyses is needed.

The manta trawl remains a useful means of collecting and evaluating microparticle types with more uniform dimensions greater than 355 µm, with the particular advantage of being able to sample and filter relatively large volumes via towing. However, given its suspected limitation regarding collection of a representative number of fibers from surface water, it may be desirable to pair the manta trawl collection method with another method better designed to capture fibers. Surface water grab samples of at least three to four liters are suggested as a complementary method; evaluation and control of background contamination with fibers is

essential for both methods. Further exploration of *in situ* filtration methods may be warranted as well; while tests conducted as part of this study of surface water were compromised by background contamination, conceptually similar water grab devices were used successfully at depth in a recent study of the Monterey Bay National Marine Sanctuary (Choy et al., 2019).

Surface water microparticle abundance varied greatly in the region

The present results indicated significant variation in microparticle abundance in surface water over time and space. Measurements of abundance exhibited large variability with respect to overall numbers of particles and individual morphologies, even for duplicate samples collected in the same location and within an hour of one another.

One particular consideration is that tides, currents, and wind have been shown to concentrate buoyant and semi-buoyant material in oceanic and estuarine fronts such as tidal fronts, windrows, and eddies (Welden and Lusher, 2017). The plankton and organic debris that accumulate in these fronts create areas of high productivity and are biologically important both at the ocean surface and at depth (Franks, 1992; Owen, 1981; Payton, 2017). Diverse marine life such as zooplankton, fish, sea birds, and mammals have been observed foraging and spawning at fronts (Bennett and Burau, 2015; Owen, 1981; Payton, 2017), demonstrating the importance of these accumulation zones to animals at a variety of trophic levels.

In the present study, at least one case of sampling occurred in which the trawl passed through a prominent tidal front halfway through the sample collection. From the side of the vessel,



vegetation, woody debris, trash, and plastic fragments were observed floating at the surface and were captured in the sample. Collected in the Central Bay, the CB9-Manta-11Jan18 sample filled 5 sample jars (three 500 mL and two 1.9 L jars) and was found to have a microparticle abundance of 6,200,000 particles/km² (excluding fibers). This was the highest microparticle abundance recorded within the study (with the next closest particle abundance, excluding fibers, being 580,000 particles/km²). This sample could be indicative of the microparticle levels found in these biologically important fronts within the Bay and the nearshore marine sanctuaries. It supports previous findings of microparticle accumulation within fronts in the oceanic gyres (GESAMP Working Group 40, 2016; Moore et al., 2001) and draws attention to the negative impacts microplastics could have on the abundant marine life feeding in these areas.

Despite this variability, when comparing the microparticle abundance in the Bay to that in the adjacent marine sanctuaries, the difference in values is consistent with the hypothesis that the movement of water out of the Bay may be transporting microparticles into the marine sanctuaries, particularly during wet weather events. Additionally, microparticle abundance is also presumably lower outside of the Bay because of flushing and dilution in the open ocean waters. Nevertheless, the variability noted in duplicate samples suggests that more monitoring may be needed to refine estimates of baseline condition and evaluate trends.

Microparticles in the Bay and marine sanctuaries relative to other regions

Surface water microparticle abundances observed in the Bay and sanctuaries can be readily compared to those reported in studies using a comparable sample collection method. The manta trawl, which is able to capture particles greater than 355 µm from the top few centimeters of the water surface, has been deployed previously to investigate microparticles and microplastics in estuarine and marine settings. These studies provide a suitable frame of reference from which to evaluate the relative abundance of microparticles observed in the present study (Tables 4.8 and 4.9). Values with and without the contribution of fibers are reported, given the uncertainties around the manta trawl as an appropriate sample collection method for this morphology.

San Francisco Bay microparticle abundances (excluding sanctuaries) were compared to a previous study of the Bay (Sutton et al., 2016), as well as other studies of large, urbanized water bodies for which the manta trawl was used (Table 4.8). The previous Bay study did not use spectroscopy to verify whether the particles captured were in fact plastic; as a result, the abundances reported must be considered microparticles, rather than microplastics.

Sutton et al. (2016) collected samples during the wet season, but not immediately following a precipitation event. In contrast, samples in the current study were collected from San Francisco Bay within one to three days of a storm event, which met specific criteria to define the size of the storm.

Table 4.8. San Francisco Bay microparticle abundance relative to abundances collected via manta trawl in large, urban water bodies (select studies).

Reference	Location	Abundance (particles/km²) without Fibers*			Abundance (particles/km²) with Fibers			Plastic ID method
		Min	Max	Median/Mean	Min	Max	Median/Mean	
Present study	San Francisco Bay	2,400	6,200,000	Median 44,000 Mean 270,000	34,000	1,800,000	Median 280,000 Mean 390,000	FTIR, Raman
Sutton et al., 2016	San Francisco Bay	10,000	1,300,000	Median 460,000 Mean 510,000	15,000	2,000,000	Median 730,000 Mean 700,000	visual only
Eriksen et al., 2013	Laurentian Great Lakes	0	465,000	Mean 43,000	0	466,000	Mean 43,000	visual only
Davis & Murphy, 2015	Salish Sea and Inside Passage	0	130,000	Mean 19,000	0	131,000	Mean 19,200	visual only
Gewert et al., 2017	Baltic Sea	1,400	255,000	Mean 17,850	15,600	618,000	Mean 109,800	FTIR
Yonkos et al., 2014	Chesapeake Bay Tributaries	fibers present, % of total particles not reported			5,000	298,000	Median 71,000	Raman

*Estimated from reported % fibers out of total particles if not reported directly.

In general, the Bay abundances reported in the present study are consistent with those of the previous study (Table 4.8); the ranges of values in both studies are comparable. Without accounting for season, the present Bay data means and medians are lower than those previously reported; however, the wet season mean and median abundances in the present study (520,000 and 98,000 particles/km² excluding fibers; 730,000 and 610,000 particles/km² including fibers) are closer in value to the previous study. Approximately half of the particles identified in the previous study were fibers, while in the present study, around 70% of the particles identified in a subset of samples were fibers.

In the present study, we did not observe a trend of higher particle counts in South and Lower South Bay compared to the Central Bay, as observed by Sutton et al. (2016). One reason for this may be that the prior study captured considerable vegetation in many of the samples, which can entrain more microparticles in the surface water prior to or during sampling. Additionally, the current study did not enumerate fibers in all samples, including many in the South Bay, and there is a chance that the samples that were collected as part of the wet weather sampling may not evenly represent the subembayments.

As reported previously (Sutton et al., 2016), microparticle abundance in the Bay appeared to be higher than abundances reported for large urbanized water bodies elsewhere, including the Great Lakes (Eriksen et al., 2013a), Chesapeake Bay tributaries (Yonkos et al., 2014), Salish Sea (Davis and Murphy, 2015), and most recently in the Baltic Sea (Gewert et al., 2017; Table 4.8). Additionally, the maximum microparticle measurement in the Central Bay (CB9, collected on January 11, 2018) of 6,200,000 particles/km² (excluding fibers) was among the highest ever measured and reported for any setting (a sample in the North Pacific was found to have 12,000,000 particles/km²; Law et al., 2014; Table 4.9).

In contrast, microparticle abundance in the marine sanctuaries did not appear exceptional, and fell within the range of abundances reported in other open ocean marine settings (Table 4.9). A broad summary of open ocean marine data from multiple oceans indicated 70% of samples had abundances between 1,000 and 100,000 particles/km² (Eriksen et al., 2014). This study's marine sanctuaries median of 82,000 particles/km² and mean of 110,000 particles/km² (including fibers) suggested comparable levels of contamination.

A handful of the studies discussed above have employed Raman or FTIR spectroscopy or other techniques to provide secondary confirmation as to whether individual microparticles observed were, in fact, microplastics. For example, FTIR spectroscopy on Arctic microparticles determined 50% of the particles examined were plastic (15% polyester, 15% nylon, 5% polyethylene, 10% acrylic, 5% polyvinyl chloride; Lusher et al., 2015). A study carried out in the Baltic Sea used FTIR spectroscopy to determine that 81% of the particles characterized were plastic (53% polypropylene, 24% polyethylene, 4% polystyrene; Gewert et al., 2017). Another study in the northwest Pacific Ocean, near Japan, determined that 97% of the particles were plastic (58% polyethylene, 36% polypropylene, 3% nylon; Pan et al., 2019). In this study, 68% of the particles analyzed by spectroscopy (FTIR/Raman) were found to be plastic, within the range reported by these studies.

Table 4.9. Microparticle abundance found in the marine sanctuaries relative to abundances collected via manta trawl in other parts of the ocean (select studies).

Reference	Location	Abundance (particles/km ²) without Fibers*			Abundance (particles/km ²) with Fibers			Plastic ID method
		Min	Max	Median/Mean	Min	Max	Median/Mean	
Present study	Bay Area National Marine Sanctuaries	0	39,300	Median 8,400; Mean 11,800	0	261,000	Median 82,000; Mean 107,000	FTIR, Raman
Pedrotti et al., 2016	Mediterranean Sea	fibers present, % of total particles not reported			21,000	578,000	Mean 196,000	FTIR
Pan et al., 2019	Northwestern Pacific	580	38,000	Median 3,100; Mean 9,100	640	42,000	Median 3,400; Mean 10,000	Raman, SEM
Reisser et al., 2013	Australian coast	fibers present, % of total particles not reported			0	49,000	Median 1,900; Mean 4,300	FTIR
Law et al., 2010	North Atlantic subtropical gyre	0	580,000	Mean 20,300	fibers not counted**			visual only
Law et al., 2014	North Pacific subtropical gyre	0	12,320,000	Medan 17,300	fibers not counted**			visual only
Eriksen et al., 2013	South Pacific subtropical gyre	0	344,000	Mean 23,400	0	396,000	Mean 26,900	visual only
Lusher et al., 2015	Arctic waters	0	5,400	Mean 1,400	0	108,000	Mean 28,000	FTIR
ter Halle et al., 2017	North Atlantic subtropical gyre	whether fibers present not reported			pieces > 1mm 10,000; pieces < 1mm 500,000	pieces > 1mm 250,000; pieces < 1mm 7,000,000	FTIR, GC-MS	

*Estimated from reported % fibers out of total particles if not reported directly. **Dr. Kara Lavendar Law, personal communication, August 2019.

Estimation of the Abundance of Microplastics in Bay Surface Water

We estimated an upper and lower bound average of the abundance of microplastics in the Bay by season, using different assumptions that were more and less conservative. Estimates for the marine sanctuaries were not developed because a relatively limited number of samples were analyzed for fibers, and among those for which fibers were enumerated, few had fiber levels greater than the data qualification threshold.

First, we estimated an upper and lower bound of the average number of microfibers in Bay surface water. The upper bound was estimated by blank-correcting microfiber counts by using the average of the laboratory and field blanks (26.9 microfibers/sample). The lower bound was estimated by blank-correcting microfiber counts using the conservative data qualification threshold (85.2 microfibers/sample). Since not all microfibers are plastic, we then estimated the percentage of plastic microfibers in surface water for both wet and dry seasons based on the composition of fibers that were analyzed by spectroscopy. An upper bound estimate of plastic microfibers for the Bay was calculated using the percentage of fibers confirmed to be plastic (53%) and assuming that 60% of the anthropogenic unknown fibers (14%) were also plastic. The estimate of 60% of anthropogenic unknown fibers being plastic is based on the industry estimate that approximately 60% of textiles today are made from nylon and polyester (Almroth et al., 2018). The lower bound estimate did not include any potential plastic contribution from the percentage of fibers identified as anthropogenic unknown. Wet season abundance estimates resulted in a lower bound of 270,000 plastic microfibers/km² and an upper bound of 340,000 plastic microfibers/km². Dry season abundance estimates resulted in a lower bound of 40,000 plastic microfibers/km² and an upper bound of 59,000 microfibers/km². These calculations indicated that more conservative assumptions regarding prevalence of microplastics, when including fibers, did not significantly decrease estimated average abundances.

The upper and lower bounds for non-fiber particles (fragments, foams, spheres, and films) were calculated in a similar manner. The upper bound was calculated by blank-correcting using the average of the laboratory and field blanks (1 for fragments and 0.1 for foams; all other particles were either not detected or only detected once in blanks). The lower bound estimate was calculated after blank-correcting by morphology using the conservative data qualification threshold (4.3 for fragments and 0.6 for foams). An estimate of the upper bound of Bay microplastic abundance for non-fiber particles was calculated by multiplying the individual abundances for each morphology by the percentage of plastic identified by spectroscopy (i.e., fragments 87%; foams 68%; spheres 97%; and films 83%). In addition, 50% of the anthropogenic unknown and unknown potentially rubber particles were assumed to be plastic. For the lower bound, all unknown particles were assumed not to be plastic.

Based on this, average microplastic abundance (excluding fibers) for the wet and dry seasons were calculated for the Bay. Wet season microplastic average abundance estimates were 440,000–450,000 microplastics/km². Dry season average abundance estimates were 42,000–45,000 microplastics/km². These calculations also indicated that more conservative assumptions regarding prevalence of non-fiber microplastics did not significantly decrease estimated average abundances.

Potential sources of microplastics in surface waters

Identifying the potential sources of microplastics allows better prioritization of management actions. The morphology and chemical composition of individual microplastics can often hint at their source. The manta trawl samples contained a variety of microplastics for which tentative source identification was possible.

FRAGMENTS

Most fragments in the manta trawl samples were 0.5–2 mm and white or clear hard plastics (47% polyethylene, 25% polypropylene). These results were consistent with other observations of marine microplastics (GESAMP Working Group 40, 2016; Hidalgo-Ruz et al., 2012).

Fragments may be derived from the breakdown of larger plastic macrodebris, including single-use plastic items. Both polyethylene and polypropylene are common polymers used in single-use items such as packaging and foodware, as well as a plethora of other items that may end up as pollution in the Bay, such as toys, durable goods, furniture, and construction materials (Eriksen et al., 2016). Breakdown of materials used in fishing and marine industries (e.g., buoys, spacers) also contribute microplastic fragments (Andrady, 2011; GESAMP Working Group 40, 2016). Studies have shown that polyethylene and polypropylene items in marine environments produce microplastic fragments after several weeks of weathering (Barnes et al., 2009; ter Halle et al., 2016; Weinstein et al., 2016).

Another potential source of fragments are particles of plastic in the size range of 10–500 µm that are intentionally added to personal care and cleaning products as abrasives. Together with spherical particles, these are known as microbeads (Browne, 2015; Chang, 2015; Fendall and Sewell, 2009; Rochman, 2018; Rochman et al., 2015; Scudo et al., 2017; Verschoor et al., 2016). Polyethylene and polypropylene fragments were observed in treated wastewater effluent from Bay Area treatment plants (see Chapter 3 Wastewater), and a portion of the fragments observed in these wastewater samples and in the manta trawl samples may be from this source. The use of microbeads in personal care products has been reduced through legislative, educational, and corporate reduction efforts; however, there are other products that continue to use plastic microbeads that could enter wastewater.

FOAM

Most of the foam microparticles were polystyrene (53%), between 0.5 and 2 mm in size, and white in color. Expanded polystyrene foam microparticles have been observed previously in surface water samples (e.g., Bimali Koongolla et al., 2018; Davis and Murphy, 2015; Song et al., 2015). Data also suggest that expanded polystyrene foam is one of the main contributors to beach pollution (Allen et al., 2017; Bimali Koongolla et al., 2018; Davis and Murphy, 2015; Sagawa et al., 2018).

These microplastics were likely derived from the breakdown of larger expanded polystyrene foam debris. Expanded polystyrene is often used for single-use packaging, particularly for food items. The 2018 Better-Alternatives-Now (B.A.N.) List of the top 20 plastic litter items observed by volunteers during international beach clean-ups includes expanded polystyrene foam take-out containers (ranked ninth; 3.2% of total items) and cups and plates (ranked 13th; 2.6% of total items). While the size of identifiable beach litter items is larger than the microparticles analyzed in the present study, it is still noteworthy that expanded polystyrene foam microparticles made up a similar percentage of the total particles (approximately 5%) observed in surface waters. An additional source of expanded polystyrene foam to surface waters is marine equipment such as buoys and floating docks.

SPHERES

The majority of spheres found in surface water were polyethylene (77%) or polystyrene (7%) and ranged in size from 0.25 to 1 mm. Most of the spheres collected in the surface water samples fell into two distinct size categories: those less than 1 mm in diameter (379 recorded in manta trawl samples), and those between 1 and 5 mm (101 recorded in manta trawl samples).

Spheres are identifiable as primary microplastics that were manufactured to be this size, as larger plastic debris does not break down into this shape. Plastic spheres are used either in their original form (e.g., microbeads in personal care products or industrial abrasives; usually less than 1 mm in diameter) or are molded into larger plastic products (e.g., pre-production pellets, also known as nurdles; usually 2–5 mm in diameter). The size distribution of spheres observed in manta trawl samples suggested both sources, microbeads and pre-production pellets, were likely contributing to microplastic pollution in the region.

Polyethylene microbeads are often associated with personal care products or industrial abrasives and make their way to the Bay through the wastewater and stormwater pathways. Polystyrene microbeads are sometimes used for ion-exchange in water softening and other water purification treatments, as well as for other industrial applications (Ballent et al., 2016; Mani et al., 2019).

FILM

The majority of detected films were polyethylene (36%), polypropylene (23%), or acrylic (10%); most were 0.5–2 mm or more than 5 mm in size; and most were clear (46%) or white (16%), with the remaining 21% made up of black, blue, green, and yellow films. Films were detected in the Bay more frequently than is commonly observed in open-ocean marine studies, likely due to the highly urbanized nature of the Bay Area.

The polyethylene and polypropylene films were likely the result of breakdown of larger film-like plastic debris such as plastic bags and other packaging (Harrison et al., 2018; McKeen, 2014; Piehl et al., 2018; Plastic Europe, 2017).

FIBERS

This study identified high quantities of microfiber contamination throughout the Bay. Many fibers were identified as plastic (24% polyester, 10% acrylic, 9% polypropylene, 3% polyethylene, and 1% polystyrene), and some were identified as anthropogenic synthetic in origin due to the presence of dyes (1%).

The production and use of synthetic textiles is a known source of plastic microfibers to the environment (Almroth et al., 2018; Bomgardner, 2017; Browne et al., 2011; Bruce et al., 2016; Gustafsson et al., 2019; Henry et al., 2019; Hernandez et al., 2017; McIlwraith et al., 2019; Pirc et al., 2016). Among the types of fibers observed, polyester and acrylic are commonly used in textiles (Almroth et al., 2018; De Falco et al., 2019; Henry et al., 2019; The Fiber Year Consultants, 2018). Microfibers are shed and can become airborne as clothing is worn and textiles are used in a variety of residential and commercial applications. Plastic microfibers are a common component of indoor and outdoor dust (Dris et al., 2018, 2015; Liu et al., 2019), and were also a common source of blank contamination in this study, likely due to their ubiquitous airborne presence in urban areas.

Washing, drying, and wearing or using synthetic textiles such as clothing, bedding, and other synthetic fabrics releases microfibers directly into the wastewater stream as well as to the air (Almroth et al., 2018; Browne et al., 2011; Bruce et al., 2016; Gustafsson et al., 2019; Hernandez et al., 2017; Mason et al., 2016; McIlwraith et al., 2019; Pirc et al., 2016). Wastewater treatment does not effectively remove all fibers from wastewater. Wastewater treatment



facilities are able to remove microparticles with a relatively high efficiency, in the range of 83% to 99.9% (Carr et al., 2016; Dris et al., 2015; Michielssen et al., 2016; Mintenig et al., 2017; Murphy et al., 2016; Talvitie et al., 2017). Removal efficiencies for fibers specifically have not been widely reported; a study of three treatment facilities in North Carolina observed fiber removal efficiencies ranging from 84% to 97% (Conley et al., 2019). Fibers are ubiquitously detected in final effluent and are frequently the most common morphology identified (Dris et al., 2015; Magnusson and Norén, 2014; Murphy et al., 2016; Sutton et al., 2016; Talvitie et al., 2017; Ziajahromi et al., 2017).

Fishing and other aquatic industries represent an additional input of fibers to surface waters. Polypropylene fibers can shed from synthetic ropes used in the marine industry (Gewert et al., 2017; Mohamed Nor and Obbard, 2014). Fishing nets and line are mainly made of polyethylene (and nylon) and may also be a source of fiber pollution.

Conclusions

The purpose of this study was to characterize microparticles and microplastics in surface waters within San Francisco Bay and the three nearshore National Marine Sanctuaries. In total, 73 manta trawl samples (field samples, duplicates, and field blanks) were collected during the project. In all samples, fragments, foams, spheres, and films were enumerated; fibers were only analyzed in approximately half (52%) of the manta trawl samples.

Average microparticle abundance varied throughout the project area, ranging from 6,300 microparticles/km² during dry weather sampling in the sanctuaries to 520,000 microparticles/km² during wet weather sampling within the Bay (excluding fibers). For the subset of samples for which fibers were quantified, average fiber abundance ranged from 68,000 fibers/km² during dry weather sampling in the sanctuaries to 580,000 microparticles/km² during wet weather sampling within the Bay. Microparticle abundance was statistically higher in the Bay than in the adjacent marine sanctuaries, and microparticle abundances in the Bay included some of the highest observed to date in nearshore urban environments.

Microparticle abundance was also higher in San Francisco Bay samples collected during the wet weather season vs. dry weather. This result suggested that wet weather may mobilize microplastics from the surrounding Bay Area watersheds. A statistically significant seasonal effect was not observed in the marine sanctuaries, possibly due to the low abundance observed in the ocean.

Excluding fiber particles, the dominant morphology in the manta trawl samples was fragments, which made up 73% of the particles, followed by foams, which made up 17% of the particles. In samples where fibers were analyzed, more than 74% of the microparticles were fibers. For all of the samples, a subsample of particles were analyzed with Raman or FTIR spectroscopy, showing that 53% of fibers (including fiber bundles), 87% of fragments, 68% of foams, 97% of spheres, and 83% of films were plastic.

Polyethylene and polypropylene fragments, polystyrene foams, and polyethylene and polypropylene films made up a majority of the non-fiber microparticles that underwent spectroscopy. These polymer and particle types may be linked to the breakdown of single-use plastic items, packaging, and bags. Polyethylene beads were also identified in the surface waters, possibly linked to microbeads found in personal care and cleaning products. For the fibers that underwent spectroscopy, about half of the fibers were identified as plastic. However, it was difficult to determine the composition of many fibers because of the presence

of dyes and coloring agents, though they did indicate that many of the fibers were anthropogenic in origin.

Because the long and narrow shape of fibers and their orientation can dramatically affect whether they are caught by the manta trawl net, it is not clear that manta trawl sampling can capture a representative count of these particles (Barrows et al., 2018; Covernton et al., 2019). The 1 L grab samples were collected to determine whether this method might provide a more representative characterization of fibers in surface water. Most of the field samples could not be differentiated from the field and laboratory blanks. To improve this sampling method, larger volumes, at least three to four liters, should be explored in future field work.

Compared to other reports, San Francisco Bay appears to have higher levels of microparticles, with one sample collected across a tidal front having an abundance of over 6 million particles/km². The types of microplastics observed in the Bay were consistent with derivation from multiple sources common in urban settings, including fragmentation of single-use plastic items, pre-production pellets and microbeads, and synthetic textiles.

References

- Allen, K., Cohen, D., Culver, A., Cummins, A., Curtis, S., Eriksen, M., Gordon, M., Howe, A., Lapis, N., Prindiville, M., Thorpe, B., Wilson, S., 2017. Better Alternatives Now B.A.N. List 2.0. 5 Gyres, Algalita, Californians Against Waste, Clean Production Action, Plastic Pollution Coalition, Responsible Purchasing Network Story of Stuff, Surfrider Foundation, UPSTREAM.
- Almroth, B.M.C., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles; a source of microplastics released into the environment. *Environmental Science Pollution Research* 25, 1191–1199. <https://doi.org/10.1007/s11356-017-0528-7>
- Andrade, A.L., 2011. Microplastics in the marine environment. *Marine Pollution Bulletin* 62, 1596–1605. <https://doi.org/10.1016/j.marpolbul.2011.05.030>
- Ballant, A., Corcoran, P.L., Madden, O., Helm, P.A., Longstaffe, F.J., 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. *Marine Pollution Bulletin* 110, 383–395. <https://doi.org/10.1016/j.marpolbul.2016.06.037>
- Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. *Philosophical Transactions of the Royal Society B: Biological Sciences* 364, 1985–1998. <https://doi.org/10.1098/rstb.2008.0205>
- Barrows, A.P.W., Cathey, S.E., Petersen, C.W., 2018. Marine environment microfiber contamination: Global patterns and the diversity of microparticle origins. *Environmental Pollution* 237, 275–284. <https://doi.org/10.1016/j.envpol.2018.02.062>
- Barrows, A.P.W., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab vs. neuston tow net: A microplastic sampling performance comparison and possible advances in the field. *Analytical Methods* 9, 1446–1453. <https://doi.org/10.1039/C6AY02387H>
- Bennett, W.A., Burau, J.R., 2015. Riders on the storm: Selective tidal movements facilitate the spawning migration of threatened delta smelt in the San Francisco Estuary. *Estuaries and Coasts* 38, 826–835. <https://doi.org/10.1007/s12237-014-9877-3>
- Bimali Koongolla, J., Andrade, A.L., Terney Pradeep Kumara, P.B., Gangabadage, C.S., 2018. Evidence of microplastics pollution in coastal beaches and waters in southern Sri Lanka. *Marine Pollution Bulletin* 137, 277–284. <https://doi.org/10.1016/j.marpolbul.2018.10.031>
- Bomgardner, M.M., 2017. The great lint migration. *Chemical & Engineering News*.

Chapter 4—Surface Water

Browne, M.A., 2015. Sources and Pathways of Microplastics to Habitats, in: Bergmann, M., Gutow, L., Klages, M. (Eds.), *Marine Anthropogenic Litter*. Springer International Publishing, Cham, pp. 229–244. https://doi.org/10.1007/978-3-319-16510-3_9

Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environmental Science & Technology* 45, 9175–9179. <https://doi.org/10.1021/es201811s>

Bruce, N.J., Hartline, N.L., Karba, S.N., Ruff, B., Sonar, S.U., 2016. Microfiber pollution and the apparel industry (Master's Thesis). Bren School of Environmental Science & Management, University of California, Santa Barbara.

Cai, L., Wang, J., Peng, J., Tan, Z., Zhan, Z., Tan, X., Chen, Q., 2017. Characteristic of microplastics in the atmospheric fallout from Dongguan city, China: Preliminary research and first evidence. *Environmental Science Pollution Research* 24, 24928–24935. <https://doi.org/10.1007/s11356-017-0116-x>

Carr, S.A., Liu, J., Tesoro, A.G., 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* 91, 174–182. <https://doi.org/10.1016/j.watres.2016.01.002>

Chang, M., 2015. Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. *Marine Pollution Bulletin* 101, 330–333. <https://doi.org/10.1016/j.marpolbul.2015.10.074>

Choy, C.A., Robison, B.H., Gagne, T.O., Erwin, B., Firl, E., Halden, R.U., Hamilton, J.A., Katija, K., Lisin, S.E., Rolsky, C., S. Van Houtan, K., 2019. The vertical distribution and biological transport of marine microplastics across the epipelagic and mesopelagic water column. *Science Reports* 9, 7843. <https://doi.org/10.1038/s41598-019-44117-2>

Conley, K., Clum, A., Deepe, J., Lane, H., Beckingham, B., 2019. Wastewater treatment plants as a source of microplastics to an urban estuary: Removal efficiencies and loading per capita over one year. *Water Research* X 3, 100030. <https://doi.org/10.1016/j.wroa.2019.100030>

Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. *Science of The Total Environment* 667, 124–132. <https://doi.org/10.1016/j.scitotenv.2019.02.346>

Davis, W., Murphy, A.G., 2015. Plastic in surface waters of the Inside Passage and beaches of the Salish Sea in Washington State. *Marine Pollution Bulletin* 97, 169–177. <https://doi.org/10.1016/j.marpolbul.2015.06.019>

Chapter 4—Surface Water

De Falco, F., Di Pace, E., Cocca, M., Avella, M., 2019. The contribution of washing processes of synthetic clothes to microplastic pollution. *Science Reports* 9, 6633.
<https://doi.org/10.1038/s41598-019-43023-x>

Dehaut, A., Cassone, A.-L., Frère, L., Hermabessiere, L., Himber, C., Rinnert, E., Rivière, G., Lambert, C., Soudant, P., Huvet, A., Duflos, G., Paul-Pont, I., 2016. Microplastics in seafood: Benchmark protocol for their extraction and characterization. *Environmental Pollution* 215, 223–233. <https://doi.org/10.1016/j.envpol.2016.05.018>

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: A case study in Greater Paris. *Environmental Chemistry* 12, 592.
<https://doi.org/10.1071/EN14167>

Dris, R., Gasperi, J., Saad, M., Mirande, C., Tassin, B., 2016. Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? *Marine Pollution Bulletin* 104, 290–293.
<https://doi.org/10.1016/j.marpolbul.2016.01.006>

Dris, R., Gasperi, J., Tassin, B., 2018. Sources and Fate of Microplastics in Urban Areas: A Focus on Paris Megacity, in: Wagner, M., Lambert, S. (Eds.), *Freshwater Microplastics : Emerging Environmental Contaminants?* Springer International Publishing, Cham, pp. 69–83.
https://doi.org/10.1007/978-3-319-61615-5_4

Eriksen, M., Lebreton, L.C.M., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: More than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. *PLoS ONE* 9, e111913.
<https://doi.org/10.1371/journal.pone.0111913>

Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S., 2013a. Microplastic pollution in the surface waters of the Laurentian Great Lakes. *Marine Pollution Bulletin* 77, 177–182. <https://doi.org/10.1016/j.marpolbul.2013.10.007>

Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., Rifman, S., 2013b. Plastic pollution in the South Pacific subtropical gyre. *Marine Pollution Bulletin* 68, 71–76. <https://doi.org/10.1016/j.marpolbul.2012.12.021>

Eriksen, M., Prindiville, M., Thorpe, B., 2016. The Plastics Better Alternatives Now Report. 5 Gyres.

Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face: Microplastics in facial cleansers. *Marine Pollution Bulletin* 58, 1225–1228.
<https://doi.org/10.1016/j.marpolbul.2009.04.025>

Chapter 4—Surface Water

- Franks, P., 1992. Phytoplankton blooms at fronts: Patterns, scales, and physical forcing mechanisms. *Reviews in Aquatic Sciences* 6, 121–137.
- Free, C.M., Jensen, O.P., Mason, S.A., Eriksen, M., Williamson, N.J., Boldgiv, B., 2014. High-levels of microplastic pollution in a large, remote, mountain lake. *Marine Pollution Bulletin* 85, 156–163. <https://doi.org/10.1016/j.marpolbul.2014.06.001>
- General Oceanics, 2018. 2030 and 2031 Series Mechanical and Electronic Digital Flowmeter Operators Manual.
- GESAMP, 2019. Guidelines on the monitoring and assessment of plastic litter and microplastics in the ocean (GESAMP No. 99). GESAMP.
- GESAMP Working Group 40, 2016. Sources, fate and effects of microplastics in the marine environment – Part two of a global assessment (DRAFT for GESAMP Review).
- Gewert, B., Ogonowski, M., Barth, A., MacLeod, M., 2017. Abundance and composition of near surface microplastics and plastic debris in the Stockholm Archipelago, Baltic Sea. *Marine Pollution Bulletin* 120, 292–302. <https://doi.org/10.1016/j.marpolbul.2017.04.062>
- Gustafsson, R.E., Hagman, H., Lindberg, K., Rehnberg, F., 2019. Microplastic emissions from domestic laundry (Bachelor's Thesis). Institution of Architecture and Civil Engineering, Division of Water Environment Technology, Chalmers University of Technology, Gothenburg, Sweden.
- Harrison, J.P., Boardman, C., O'Callaghan, K., Delort, A.-M., Song, J., 2018. Biodegradability standards for carrier bags and plastic films in aquatic environments: A critical review. *Royal Society Open Science* 5, 171792. <https://doi.org/10.1098/rsos.171792>
- Henry, B., Laitala, K., Klepp, I.G., 2019. Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. *Science of The Total Environment* 652, 483–494. <https://doi.org/10.1016/j.scitotenv.2018.10.166>
- Hernandez, E., Nowack, B., Mitrano, D.M., 2017. Polyester textiles as a source of microplastics from households: A mechanistic study to understand microfiber release during washing. *Environmental Science & Technology* 51, 7036–7046. <https://doi.org/10.1021/acs.est.7b01750>
- Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: A review of the methods used for identification and quantification. *Environmental Science & Technology* 46, 3060–3075. <https://doi.org/10.1021/es2031505>

Chapter 4—Surface Water

Kang, J.-H., Kwon, O.Y., Lee, K.-W., Song, Y.K., Shim, W.J., 2015. Marine neustonic microplastics around the southeastern coast of Korea. *Marine Pollution Bulletin* 96, 304–312.
<https://doi.org/10.1016/j.marpolbul.2015.04.054>

Law, K.L., Moret-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J., Reddy, C.M., 2010. Plastic accumulation in the North Atlantic Subtropical Gyre. *Science* 329, 1185–1188. <https://doi.org/10.1126/science.1192321>

Law, K.L., Morét-Ferguson, S.E., Goodwin, D.S., Zettler, E.R., DeForce, E., Kukulka, T., Proskurowski, G., 2014. Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environmental Science & Technology* 48, 4732–4738.
<https://doi.org/10.1021/es4053076>

Lippiatt, S., Opfer, S., Arthur, C., 2013. Marine Debris Monitoring and Assessment: Recommendations for Monitoring Debris Trends in the Marine Environment 88.

Liu, C., Li, J., Zhang, Y., Wang, L., Deng, J., Gao, Y., Yu, L., Zhang, J., Sun, H., 2019. Widespread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. *Environment International* 128, 116–124.
<https://doi.org/10.1016/j.envint.2019.04.024>

Lusher, A.L., Tirelli, V., O'Connor, I., Officer, R., 2015. Microplastics in Arctic polar waters: The first reported values of particles in surface and sub-surface samples. *Scientific Reports* 5.
<https://doi.org/10.1038/srep14947>

Lusher, A.L., Welden, N.A., Sobral, P., Cole, M., 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Analytical Methods* 9, 1346–1360.
<https://doi.org/10.1039/C6AY02415G>

Magnusson, K., Norén, F., 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant (No. C 55). IVL Swedish Environmental Research Institute.

Mani, T., Blarer, P., Storck, F.R., Pittroff, M., Wernicke, T., Burkhardt-Holm, P., 2019. Repeated detection of polystyrene microbeads in the Lower Rhine River. *Environmental Pollution* 245, 634–641. <https://doi.org/10.1016/j.envpol.2018.11.036>

Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., Fink, P., Papazissimos, D., Rogers, D.L., 2016. Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution* 218, 1045–1054.
<https://doi.org/10.1016/j.envpol.2016.08.056>

Chapter 4—Surface Water

Masura, B., Foster, A., 2015. Laboratory Methods for the Analysis of Microplastics in the Marine Environment (No. Technical Memorandum NOS-OR & R-48), NOAA Marine Debris Program. National Oceanic and Atmospheric Administration.

McIlwraith, H.K., Lin, J., Erdle, L.M., Mallos, N., Diamond, M.L., Rochman, C.M., 2019. Capturing microfibers – Marketed technologies reduce microfiber emissions from washing machines. *Marine Pollution Bulletin* 139, 40–45. <https://doi.org/10.1016/j.marpolbul.2018.12.012>

McKeen, L., 2014. The Effect of Long Term Thermal Exposure on Plastics and Elastomers. Elsevier. <https://doi.org/10.1016/C2013-0-00091-6>

Michielssen, M.R., Michielssen, E.R., Ni, J., Duhaime, M.B., 2016. Fate of microplastics and other small anthropogenic litter (SAL) in wastewater treatment plants depends on unit processes employed. *Environmental Science: Water Research & Technology* 2, 1064–1073. <https://doi.org/10.1039/C6EW00207B>

Miller, R.Z., Watts, A.J.R., Winslow, B.O., Galloway, T.S., Barrows, A.P.W., 2017. Mountains to the sea: River study of plastic and non-plastic microfiber pollution in the northeast USA. *Marine Pollution Bulletin* 124, 245–251. <https://doi.org/10.1016/j.marpolbul.2017.07.028>

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research* 108, 365–372. <https://doi.org/10.1016/j.watres.2016.11.015>

Mohamed Nor, N.H., Obbard, J.P., 2014. Microplastics in Singapore's coastal mangrove ecosystems. *Marine Pollution Bulletin* 79, 278–283. <https://doi.org/10.1016/j.marpolbul.2013.11.025>

Moore, C.J., Moore, S.L., Leecaster, M.K., Weisberg, S.B., 2001. A Comparison of Plastic and Plankton in the North Pacific Central Gyre. *Marine Pollution Bulletin* 42, 1297–1300. [https://doi.org/10.1016/S0025-326X\(01\)00114-X](https://doi.org/10.1016/S0025-326X(01)00114-X)

Munno, K., Helm, P.A., Jackson, D.A., Rochman, C., Sims, A., 2018. Impacts of temperature and selected chemical digestion methods on microplastic particles: Impacts of temperature and digestion method on microplastics. *Environmental Toxicology and Chemistry* 37, 91–98. <https://doi.org/10.1002/etc.3935>

Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science & Technology* 50, 5800–5808. <https://doi.org/10.1021/acs.est.5b05416>

Chapter 4—Surface Water

Owen, R.W., 1981. 7 Fronts and Eddies in the Sea: Mechanisms, Interactions and Biological Effects 37.

Pan, Z., Guo, H., Chen, H., Wang, S., Sun, X., Zou, Q., Zhang, Y., Lin, H., Cai, S., Huang, J., 2019. Microplastics in the Northwestern Pacific: Abundance, distribution, and characteristics. *Science of The Total Environment* 650, 1913–1922.
<https://doi.org/10.1016/j.scitotenv.2018.09.244>

Payton, T.G., 2017. Microplastic in the Estuarine Food Web of Charleston Harbor, SC [WWW Document]. URL <http://repository.library.cofc.edu/handle/123456789/3485> (accessed 8.29.19).

Pedrotti, M.L., Petit, S., Elineau, A., Bruzaud, S., Crebassa, J.-C., Dumontet, B., Martí, E., Gorsky, G., Cózar, A., 2016. Changes in the floating plastic pollution of the Mediterranean Sea in relation to the distance to land. *PLOS ONE* 11, e0161581.
<https://doi.org/10.1371/journal.pone.0161581>

Piehl, S., Leibner, A., Löder, M.G.J., Dris, R., Bogner, C., Laforsch, C., 2018. Identification and quantification of macro- and microplastics on an agricultural farmland. *Scientific Reports* 8, 17950. <https://doi.org/10.1038/s41598-018-36172-y>

Pirc, U., Vidmar, M., Mozer, A., Kržan, A., 2016. Emissions of microplastic fibers from microfiber fleece during domestic washing. *Environmental Science Pollution Research* 23, 22206–22211.
<https://doi.org/10.1007/s11356-016-7703-0>

Plastic Europe, 2017. Plastics – the Facts 2017 An analysis of European plastics production, demand and waste data.

R Core Team (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.

Reisser, J., Shaw, J., Wilcox, C., Hardesty, B.D., Proietti, M., Thums, M., Pattiaratchi, C., 2013. Marine plastic pollution in waters around Australia: Characteristics, concentrations, and pathways. *PLOS ONE* 8, e80466. <https://doi.org/10.1371/journal.pone.0080466>

Rochman, C.M., 2018. Microplastics research—From sink to source. *Science* 360, 28–29.
<https://doi.org/10.1126/science.aar7734>

Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Scientific evidence supports a ban on microbeads. *Environmental Science & Technology* 49, 10759–10761. <https://doi.org/10.1021/acs.est.5b03909>

Chapter 4—Surface Water

Sagawa, N., Kawai, K., Hinata, H., 2018. Abundance and size of microplastics in a coastal sea: Comparison among bottom sediment, beach sediment, and surface water. *Marine Pollution Bulletin* 133, 532–542. <https://doi.org/10.1016/j.marpolbul.2018.05.036>

SAPEA, S.A. for P. by E.A., 2019. A Scientific Perspective on Microplastics in Nature and Society. <https://doi.org/10.26356/microplastics>

Scudo, A., Liebmann, B., Corden, C., Tyrer, D., Kreissig, J., Warwick, O., 2017. Intentionally added microplastics in products (No. 39168 Final Report 17271i3). Amec Foster Wheeler Environment & Infrastructure UK Limited.

Sedlak, M., Sutton, R., Box, C., Sun, J., Lin, D., 2017. FINAL Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries. SFEI Contribution No. 819. San Francisco Estuary Institute and 5 Gyres, Richmond, CA.

Silva, A.B., Bastos, A.S., Justino, C.I.L., da Costa, J.P., Duarte, A.C., Rocha-Santos, T.A.P., 2018. Microplastics in the environment: Challenges in analytical chemistry - A review. *Analytica Chimica Acta* 1017, 1–19. <https://doi.org/10.1016/j.aca.2018.02.043>

Song, Y.K., Hong, S.H., Jang, M., Han, G.M., Shim, W.J., 2015. Occurrence and distribution of microplastics in the sea surface microlayer in Jinhae Bay, South Korea. *Arch. Environ. Contam. Toxicol. Archives of Environmental Contamination and Toxicology* 69, 279–287. <https://doi.org/10.1007/s00244-015-0209-9>

Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. *Marine Pollution Bulletin* 109, 230–235. <https://doi.org/10.1016/j.marpolbul.2016.05.077>

Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA.

Talvitie, J., Mikola, A., Setälä, O., Heinonen, M., Koistinen, A., 2017. How well is microlitter purified from wastewater? – A detailed study on the stepwise removal of microlitter in a tertiary level wastewater treatment plant. *Water Research* 109, 164–172. <https://doi.org/10.1016/j.watres.2016.11.046>

ter Halle, A., Ladirat, L., Gendre, X., Goudouneche, D., Pusineri, C., Routaboul, C., Tenailleau, C., Dupoyer, B., Perez, E., 2016. Understanding the fragmentation pattern of marine plastic debris. *Environmental Science & Technology* 50, 5668–5675. <https://doi.org/10.1021/acs.est.6b00594>

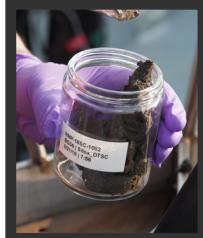
Chapter 4—Surface Water

- ter Halle, A., Jeanneau, L., Martignac, M., Jardé, E., Pedrono, B., Brach, L., Gigault, J., 2017. Nanoplastic in the North Atlantic Subtropical Gyre. *Environmental Science & Technology* 51, 13689–13697. <https://doi.org/10.1021/acs.est.7b03667>
- The Fiber Year Consultants, 2018. The fiber year 2018: World survey on textiles and nonwovens.
- Verschoor, A., de Poorter, L., Dröge, R., Kuenen, J., de Valk, E., 2016. Emission of microplastics and potential mitigation measures: Abrasive cleaning agents, paints and tyre wear (RIVM Report 2016-0026). Dutch National Institute for Public Health and the Environment (RIVM).
- Weinstein, J.E., Crocker, B.K., Gray, A.D., 2016. From macroplastic to microplastic: Degradation of high-density polyethylene, polypropylene, and polystyrene in a salt marsh habitat: Degradation of plastic in a salt marsh habitat. *Environmental Toxicology and Chemistry* 35, 1632–1640. <https://doi.org/10.1002/etc.3432>
- Welden, N.A., Lusher, A.L., 2017. Impacts of changing ocean circulation on the distribution of marine microplastic litter: Changing Ocean Circulation and Marine Microplastic Litter. *Integrated Environmental Assessment and Management* 13, 483–487. <https://doi.org/10.1002/ieam.1911>
- Woodall, L.C., Gwinnett, C., Packer, M., Thompson, R.C., Robinson, L.F., Paterson, G.L.J., 2015. Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments. *Marine Pollution Bulletin* 95, 40–46. <https://doi.org/10.1016/j.marpolbul.2015.04.044>
- Yonkos, L.T., Friedel, E.A., Perez-Reyes, A.C., Ghosal, S., Arthur, C.D., 2014. Microplastics in four estuarine rivers in the Chesapeake Bay, U.S.A. *Environmental Science & Technology* 48, 14195–14202. <https://doi.org/10.1021/es5036317>
- Ziajahromi, S., Neale, P.A., Rintoul, L., Leusch, F.D.L., 2017. Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research* 112, 93–99. <https://doi.org/10.1016/j.watres.2017.01.042>

CHAPTER
5

Microparticles and Microplastics

IN SAN FRANCISCO BAY SEDIMENT : by Diana Lin



Highlights

- ◆ This study measured microparticles and microplastics in sediment throughout San Francisco Bay and at a reference area with minimal urban influence (Tomales Bay). Sixteen of the 18 Bay sites were in the nearshore “margins” (< 250 m from shore), and most were near wastewater and/or urban stormwater discharge locations.
- ◆ Fibers, followed by fragments, were the most abundant type of microparticles in Bay sediment. Bay microparticle concentrations ranged between 1.5 and 49 microfibers per gram dry weight (median 2.1 microfibers/g dw), and 0.05 and 11 non-fiber microparticles/g dw (median 1.4 microparticles/g dw).
- ◆ A subset of microparticles was analyzed using Raman spectroscopy to establish whether they were plastic. The average concentration of plastic fibers analyzed in Bay sediment was 0.8–2.2 microplastics/g dw. The average concentration of plastic non-fibers was 1.0–1.1 microplastics/g dw. These estimated ranges are based on upper and lower bound methods that take into account the percentage of fiber and non-fiber microparticles that were confirmed to be plastic by spectroscopy, as well as accounting for background contamination through blank subtraction.
- ◆ The highest concentrations of microparticles were measured in Lower South Bay, which is strongly influenced by wastewater and urban stormwater discharges. Concentrations at the reference site were similar to the lowest concentrations measured in Bay margins.
- ◆ Black fragments that had a rubbery texture were frequently detected in sediment samples. The composition could not be confirmed using spectroscopy; however, laboratory analysts reported that based on secondary characteristics, these particles were similar to particles that had been previously identified as rubber by FTIR.
- ◆ Microparticle and microplastic concentrations in Bay sediment were higher than those reported in the majority of other regions.

Objectives

Sediment is likely a sink for microplastics that are denser than water. Microplastics that were originally buoyant and present in surface water, wastewater, or stormwater may also become less buoyant and sink to the Bay sediment floor due to the growth of biofilm, aggregation, or ingestion by organisms that eventually die and sink to the Bay floor (Anderson et al., 2016).

Characterizing microplastics in Bay sediment is important for several reasons. First, microplastics in sediment may be a source of microplastics to the Bay food web. Benthic organisms such as worms, crustaceans and bivalves may ingest microplastics from sediment (Hurley et al., 2018; Wright et al., 2013), and larger predators such as fish and birds may consume the benthos. Second, establishing a baseline level of contamination can be useful for evaluating trends and the effectiveness of management actions. Third, sediment concentrations can be used to identify potential hotspots in the Bay, which may warrant further work to evaluate sources, pathways, and potential risks to Bay wildlife. Lastly, sediment data can be used to inform transport modeling of microplastics in the Bay and the coastal ocean environment.

This is the first study to evaluate microplastics in San Francisco Bay sediment, a data gap noted in the Microplastic Monitoring and Science Strategy (Sutton and Sedlak, 2017). In this report, we have distinguished between microparticles, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and microplastics, which have been confirmed to be plastic through spectroscopy.

By assessing sediment microparticle and microplastic abundances and characteristics in San Francisco Bay and a background reference site, this study sought to address the following objectives.

- 1. Quantify the abundance of microparticles and microplastics in sediment.**

Understanding the abundance of microparticles and microplastics in sediment is important for determining the magnitude of microplastic pollution in the Bay [Management Question (MQ) 1]. This will shed light on the sources, pathways, loadings, and processes (MQ3) by which microparticles and microplastics end up in sediment through an evaluation of spatial patterns in concentration levels (e.g., proximity to stormwater and wastewater pathways). In addition, this evaluation provides a baseline to which future monitoring efforts can be compared to evaluate changes in pollution levels (MQ4).

- 2. Characterize types of microparticles found in sediment and their chemical composition.**

Understanding the types of microparticles and microplastics found in sediment will also shed light on MQ3. Future monitoring efforts can inform whether management actions on sources of specific types of microplastics have been effective.

This study was originally designed to test the following hypotheses (Sedlak et al., 2017).

- ◆ Microplastics will be present in sediment.
- ◆ Concentrations of microparticles and microplastics in sediment will be lower in remote areas such as Tomales Bay in comparison to sediment from urban San Francisco Bay.
- ◆ Concentrations of microparticles and microplastics in sediment from the main channel of the Bay will be lower than sediment concentrations in the Bay margins.
- ◆ Within the Bay, concentrations of microparticles and microplastics will be higher in areas with limited flushing such as Lower South Bay.
- ◆ Margin sediment will contain different types of microplastics than the main channel (e.g., morphology and chemical composition).

Due to challenges with analyzing sediment samples and the time required to complete analysis, the number of sediment samples analyzed for this study had to be scaled back. We focused on characterizing margin sites near pathways, sites co-located with fish collection sites, and sites from the reference area. Therefore, this study is a preliminary exploration of many of these hypotheses, rather than a robust statistical analysis.

Methods

Site selection

Sediment samples from 18 sites distributed around San Francisco Bay and two sites from a less urban reference area were collected and analyzed for microplastics (Table 5.1 and Figure 5.1). Additional sediment samples were collected as part of the original sampling and analysis plan (Sedlak et al., 2017), but given the unforeseen time required to analyze samples and the limited time available, we focused on samples from 20 high priority sites for extraction and analysis.

Sediment sites were selected to characterize microplastic concentrations near possible pathways in the nearshore “margins” of the Bay, in open or “ambient” portions of the Bay, and in a less urban reference area (Table 5.1). “Margins” are mudflats and other very shallow areas around the Bay that range from mean lower low water to the unvegetated shoreline (roughly mean high water). Most of the selected sites were in Bay margins because these areas are closely linked to potential conduits of microplastics, such as urban stormwater runoff and shallow wastewater discharges. Sites co-located with fish sampling sites were also prioritized for analysis.

Because samples were not randomly chosen, but instead selectively chosen to represent open Bay and margin areas, they cannot be used to make strict spatial inferences for the Bay as a whole.

Table 5.1. Attributes of sediment site locations.

Embayment	Sediment Type	Site ID	Rationale	Co-located site		
				Small fish site	Stormwater site	Effluent site
Tomales Bay	Margins	TB102	Reference	TB102		
Tomales Bay	Margins	TB101	Reference	TB101		
Suisun Bay	Margins	SUB53	Wastewater / Urban creek – Pacheco Creek			CCCSD
Suisun Bay	Margins	SUB52	Bay background characterization			
San Pablo Bay	Margins	SPB15	Urban river			
San Pablo Bay	Margins	SPB128	Bay background characterization			Refugio Creek
Central Bay	Open Bay	CB001S	Bay background characterization - S&T Site			
Central Bay	Margins	CB10	Bay background characterization	Near CB10		Meeker Slough
Central Bay	Margins	CB15	Urban creek - Temescal			Line12A
Central Bay	Margins	CB32	Urban creek - East Creek Slough - fish/stormwater	CB101		Line12F & H
Central Bay	Margins	CB37	Urban creek - Colma Creek - near trash hotspot tributary	CB037		Colma Creek
South Bay	Open Bay	SB002S	Bay background characterization			
South Bay	Margins	SB051	Bay background characterization			
South Bay	Margins	SB074	Urban creek	SB074		
South Bay	Margins	SB056	Stormwater			
Lower South Bay	Margins	LSB02	Wastewater discharge, proximity to fish	Near LSB06		Palo Alto WWTP
Lower South Bay	Margins	LSB04	Eastside near Mowry background characterization			
Lower South Bay	Margins	LSB06	Westside - Hooks Point	LSB06		
Lower South Bay	Margins	SOSL16	Wastewater and urban creek		Guadalupe Slough	Sunnyvale WWTP
Lower South Bay	Margins	SOSL40	Wastewater and urban creek	SOSL40	Coyote Creek	San Jose WWTP



Figure 5.1. Sites sampled for microparticles and microplastics in sediment.

Sample collection

Analysis of microparticles was conducted on archived samples collected during the 2014 RMP Annual Status and Trends Sediment Cruise (Applied Marine Sciences, 2014), the RMP 2015 Central Bay Margins Sampling (Yee et al., 2017), as well as samples collected specifically for this study via the RMP 2017 South Bay Margins Sampling (Shimabuku et al., 2017). Briefly, samples were collected using a 0.1 m² modified Van Veen sediment grab constructed of stainless steel.

The jaws and doors are coated with Kynar™ to improve chemical inertness. Samples were collected from the center of the grab (away from the sides) and at a depth of less than 5 cm using a clean stainless steel spoon. Collecting samples from the top 5 cm of sediment is consistent with recommendations from the European Union in the Marine Strategy Framework Directive for sampling microparticles and microplastics (European Commission Joint Research Center, 2013), and is also consistent with the RMP protocol for monitoring Bay surface sediment (Yee et al., 2018). The grab was cleaned at each site using a brush and soapy water, after which the grab was rinsed with site water using a high-powered hose.

Two field blanks were collected by rinsing the sampling tools in the field with filtered deionized water, with the resulting rinseate collected in a pre-cleaned sample container (Shimabuku et al., 2017). Only one of the two field blank results was used due to differences in size fractions analyzed as described below. Field blanks were collected prior to the collection of field samples during the 2017 sampling event. Two field duplicates from separate sediment grabs were collected to assess variation in sediment.

Sample extraction and analysis

The method used for microparticle and microplastic extraction from sediment included a density separation method modified from Stolte et al. (2015). Sediment samples were first sieved using a 45 µm sieve and then dried at 60°C in a drying oven. Once dry, 4.4–150 grams of dry sediment were wet sieved through a column of stacked sieves consisting of 45 µm, 125 µm, 355 µm, 500 µm, and 1 mm sieves. Each size fraction was rinsed with a CaCl₂ solution into a separatory funnel for density separation. After two rounds of density separation in 1.4 g/cm³ of CaCl₂ solution, the floating fractions were sieved through their respective sieves and rinsed with reverse-osmosis (RO) water into clean glass jars.

After density separation, microparticles resembling microplastics were visually enumerated using a dissecting microscope. Ten particles from each morphology-color category were picked, photographed, and measured for length and width. A subset of the picked particles underwent spectral analysis using Raman spectroscopy to confirm composition (Table A-5.4).

Microparticles in the 45 µm sieve (representing an operational size grouping of 45–125 µm) were not quantified in field samples due to the challenges associated with counting these particles, particularly in the absence of automated Raman software. One field blank (17MMP-S-SB-51-MP-FB) was not included in the data analysis because only the 45 µm sieve was used to separate microparticles, and therefore all of the quantified microparticles were in the 45 µm sieve fraction. For one of the laboratory blanks (Lab Blank-1), only two sieves (45 and 500 µm) were used to extract microparticles, and therefore only the 500 µm sieve fraction was used for

analyses. Results from one field blank and three laboratory blanks were used in the analyses to evaluate background contamination.

All laboratory blanks were composed of RO water processed using the same methods as the field samples. Laboratory practices to avoid procedural contamination included wearing white cotton lab coats, using a HEPA filter in the laboratory, wiping laboratory benches daily to reduce dust, cleaning all glassware with detergent and RO water, and working in a clean cabinet when possible.

Analytical method recovery evaluation

Prior to analyzing the field samples, a laboratory study was conducted to assess the efficacy of the extraction method. Bay sediment was spiked with particles of a range of sizes, colors, morphologies, and polymers. The spikes consisted of ten each the following: white/clear polyethylene terephthalate fragments, brown polystyrene fragments, red cellulose acetate beads, green polyethylene beads, and red polyester fibers, ranging in particle size from 250 μm to 1 mm. Spiked microplastics had unique colors and morphologies that made them easily distinguishable from microplastics originally present in the sediment.

In the method recovery evaluation, a 45 and 500 μm sieve were used, and particle recovery in each sieve fraction (45–500 μm and $> 500 \mu\text{m}$) was quantified to evaluate total recovery. The method recovery evaluation was conducted before the analytical method was updated to include the additional sieve sizes (125 and 355 μm).

Treatment of blanks

Laboratory and field blank results are reported alongside field sample results. Reported microparticle counts and concentrations in field samples were not blank corrected (i.e., blank counts were not subtracted from the field sample counts) due to the non-uniform nature of the background field and laboratory contamination observed.

Data for field and laboratory blanks were used to develop conservative thresholds for data qualification equal to the average laboratory blank plus two times the standard deviation for each morphology. Any field samples with values below the thresholds were qualified. Qualified values should be treated with caution because they may be influenced by background contamination from collection, processing, and analysis of samples. The field and laboratory blank data as well as the threshold values are reported so individual readers can make their own assessment.

Results

Quality assurance results

ANALYTICAL METHOD RECOVERY EVALUATION

Recovery was 80% or greater for all spiked microplastics (Table 5.2), indicating acceptable laboratory performance.

Table 5.2. Recovery of spiked microplastics in Bay sediment samples. Ten particles of each type were spiked into the samples, except for cellulose acetate beads where three particles were spiked.

Particle and Plastic Type	Particle Size	Replicate 1 Recovery	Replicate 2 Recovery
Polyethylene terephthalate fragment (clear/white)	1 mm	8 (80%)	8 (80%)
Polystyrene fragment (brown)	2 mm	9 (90%)	10 (100%)
Cellulose acetate bead (red)	1 mm	3 (100%)	3 (100%)
Polyethylene bead (green)	250–300 µm	10 (100%)	9 (90%)
Polyester fiber (red)	3 mm (length)	9 (90%)	10 (100%)

BACKGROUND CONTAMINATION: FIELD AND LABORATORY BLANKS

Microparticles were detected sporadically in laboratory blanks. The three laboratory blanks contained 18, 1, and 0 particles. Any particles in the 45 µm sieve fraction of the laboratory blanks were not included in the particle count because the particles in this sieve fraction were not quantified in field samples. All but one of the microparticles in the laboratory blanks were fibers; one fragment was detected. The fibers were various colors, and the top colors were blue (7 fibers) and clear (5 fibers). All but one of the microparticles in the laboratory blanks were analyzed by spectroscopy; many were anthropogenic unknown (i.e., dyed material that may or may not be plastic). Thirty-three percent of the particles were confirmed to be plastic. Laboratory blanks suggested that limited sample contamination occurred during processing and analysis.

The one usable field blank contained 151 microparticles from the 125 µm sieve fraction and above. Almost all of the microparticles in field blank were fibers (143 fibers, 95% of microparticles), with one film and seven fragments detected. Foam and spheres were not detected in the field blank. The dominant colors in the field blank were dark blue and blue (62 microfibers, 43% of fibers), followed by white (20 microfibers) and clear fibers (11 microfibers). Thirteen percent of the field blank particles were analyzed by spectroscopy. Forty-three

percent of the fibers were confirmed to be plastic; the remaining 57% were natural or cellulosic material. Two of the four fragments were plastic, and the remaining two were natural material. The one film particle detected in the field blank was classified as anthropogenic unknown. The field blank sample suggested the potential for elevated microfiber contamination to occur during sample collection, with limited contamination of fragment and film microparticles.

Field data were qualified when particle counts (by morphology) were less than the average of the field and laboratory blanks plus two times the standard deviation. The average number of each detected morphology in blank samples was 40.3 fibers/sample, 2.0 fragments/sample, and 0.3 films/sample. The calculated thresholds for qualifying field samples for fibers, fragments, and films were 178.1 fibers/sample, 8.7 fragments/sample, and 1.3 films/sample, respectively. Qualifying threshold concentrations were not calculated for spheres and foams because these morphologies were not detected in the blanks.

PRECISION AND VARIABILITY: FIELD DUPLICATES

Two field duplicates were collected. At site SOSL16, the primary and duplicate field samples had 17 and 12 microparticles/g dry weight (dw). At site SPB15, the primary and duplicate field samples both had 4 microparticles/g dw. The relative percent difference (RPD) for both sets of samples is 36% and 1%, respectively, indicating acceptable reproducibility. While there are no established data quality standards for microplastic analysis, a RPD of 35% is generally considered acceptable for organic contaminant analysis (Yee et al., 2018).

Particle occurrence and morphology

Microparticles were identified in sediment from all 20 sites. In total, 5,843 microparticles were enumerated from all sediment samples. Total microparticle concentrations in field samples ranged between 0.5 and 60 microparticles/g dw. The average field sample concentration was 8.5 microparticles/g dw (median 3.0 microparticles/g dw, n = 20). The field samples had statistically significantly higher concentrations than the blanks (Mann Whitney U test, $p = 0.005$).

The highest concentration of microparticles was found in a sample from the Lower South Bay margin (LSB02), containing 963 microparticles and a concentration of 60 microparticles/g dw. The lowest concentration sample was from Suisun Bay (SUB53), with a concentration of 0.5 microparticles/g dw, which is qualified due to microfiber counts below the conservative data qualification threshold. Microparticle counts at our reference sites were slightly higher than two in-Bay sites (Figure 5.2). The microparticle abundance in the top two samples was more than four times higher than the average field sample concentration of 8.5 microparticles/g dw (Figure 5.2). The dominant morphology observed in sediment samples was fibers, and the

composition of the combined field sediment samples was 69% fibers, 26% fragments, 2% films, 2% foams, and 2% spheres (Table A-5.4). However, 12 out of 20 samples were qualified due to microfiber concentrations below the conservative threshold derived from the number of fibers found in the field and laboratory blanks.

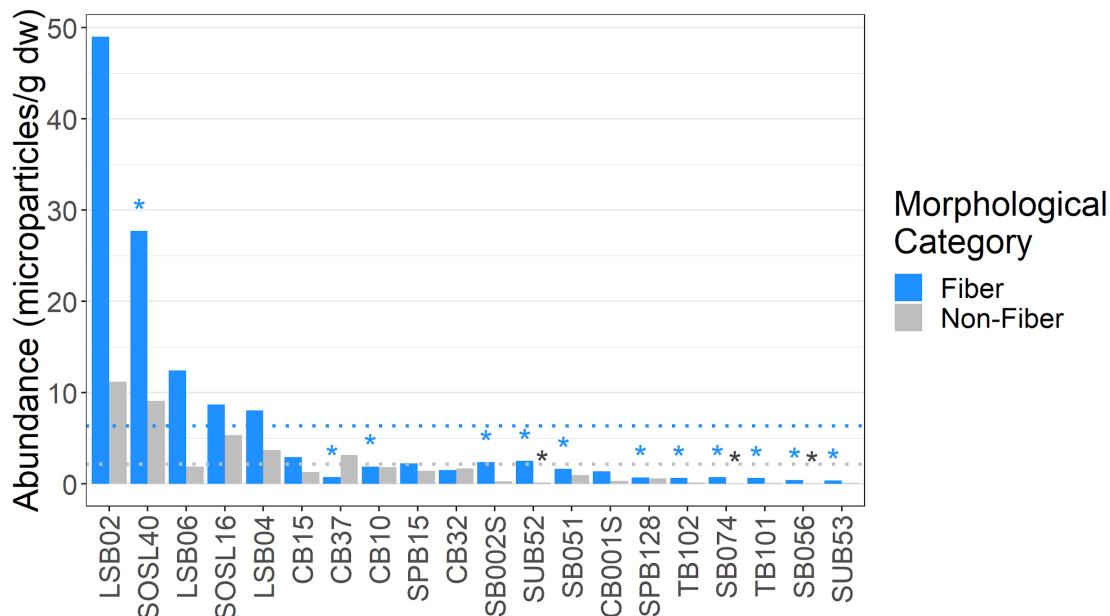


Figure 5.2. Abundance of microparticles and microplastics quantified and analyzed at 20 locations in San Francisco Bay and the reference site (Tomales Bay). Blue dotted line = average fiber concentration in Bay sediment samples (6.3 microfibers/g dw), gray dotted line = average non-fiber concentration in Bay sediment samples (2.2 microparticles/g dw). A * above bars represent values that were qualified because they were below the conservative data qualification threshold based on microparticles found in blanks. Sample SOSL40 fiber count was qualified because it had the smallest sample mass, even though the calculated concentration was higher than other samples. Non-fiber values represent the sum of film, foam, fragment, and sphere concentrations.

The most frequently detected fiber size was 1–5 mm in length (Figure 5.3), based on measuring 53% of the fibers in field samples. Fibers in the 1–5 mm fraction represented 66% of the fibers measured. The most frequently detected non-fiber microparticle size measured in sediment was in the 125–355 µm size range, which corresponds to the smallest sieve size (125 µm) quantified.

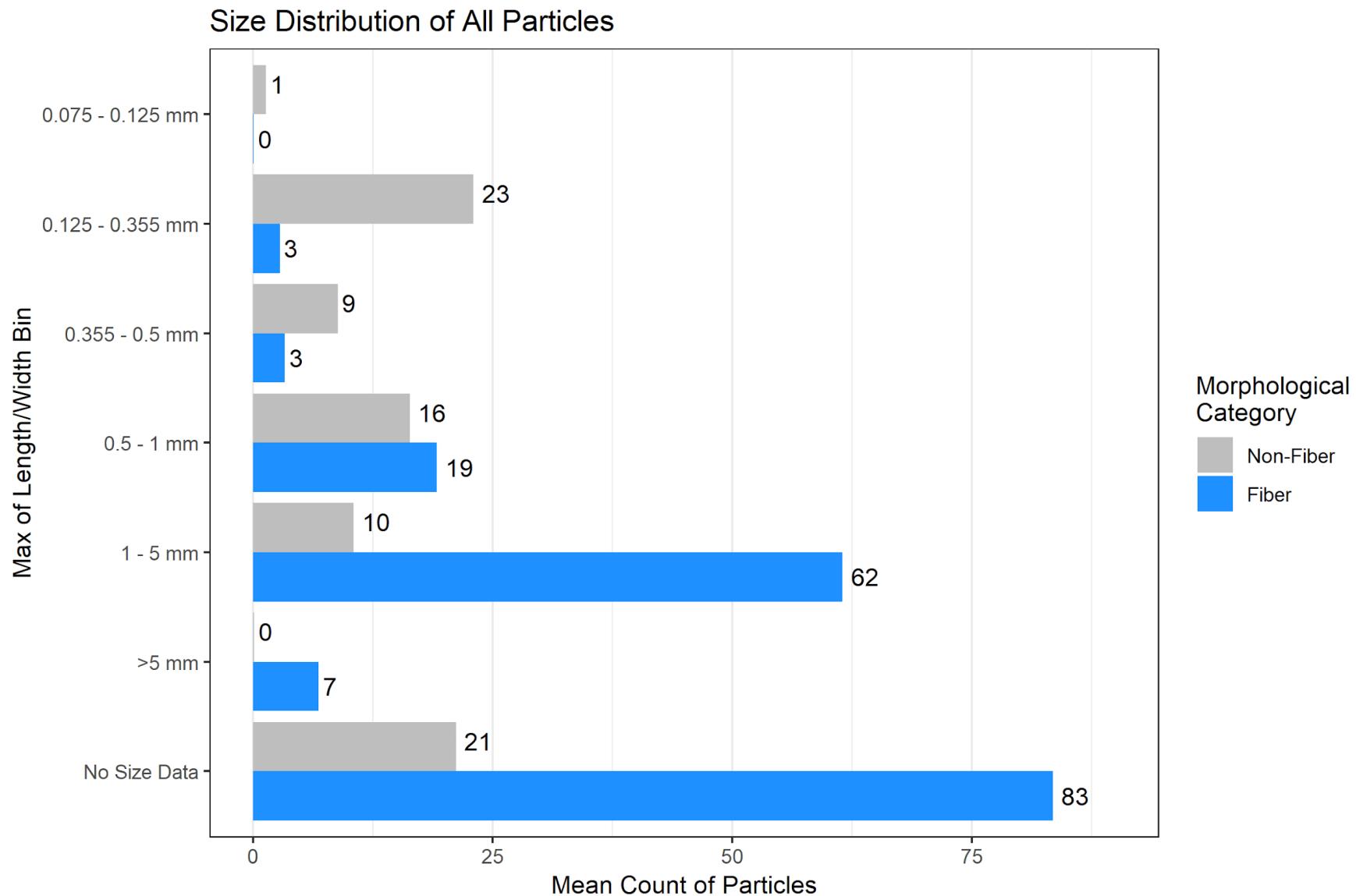


Figure 5.3. Size distribution for fibers and non-fiber particles in sediment samples. Particle size refers to the maximum dimension of the microparticle. The numbers by the bars indicate the total number of microparticles in each size bin.

Composition

FIBER CONCENTRATIONS AND COMPOSITION

Fibers were the most frequently identified morphology in sediment, with a total of 3,960 microfibers identified in field samples (Figure 5.4a). Fiber concentrations ranged between 0.4 fibers/g dw and 49 fibers/g dw (mean 6.3 fibers/g dw; median 1.8 fibers/g dw). Twelve out of 20 of the microfiber concentrations were qualified because they were below the conservative threshold derived for microfibers (178.1 fibers/sample).

Seven percent of the microfibers were analyzed using Raman spectroscopy (Table A-5.4). Thirty-one percent of the fibers analyzed were confirmed to be plastic, and the top polymers identified were polyester (28 particles, 10% of fibers analyzed), cellulose acetate (20 particles, 7% of fibers analyzed), and acrylic (14 particles, 5% of fibers analyzed). Fourteen percent of the fibers analyzed were categorized as anthropogenic unknown because they contained dyes that interfered with the Raman spectrum. Forty-eight percent of the fibers analyzed were natural-based material (139 microparticles), and seven percent (20 microparticles) could not be identified and were categorized as unknown (Table A-5.4). Twenty-six percent of the natural-based fibers could be specifically identified as cotton, while 37% were classified as anthropogenic cellulosic (dyed fibers made of cotton, rayon, modal, or Lyocell) and 35% were classified as cellulosic (similar fibers that are not dyed).

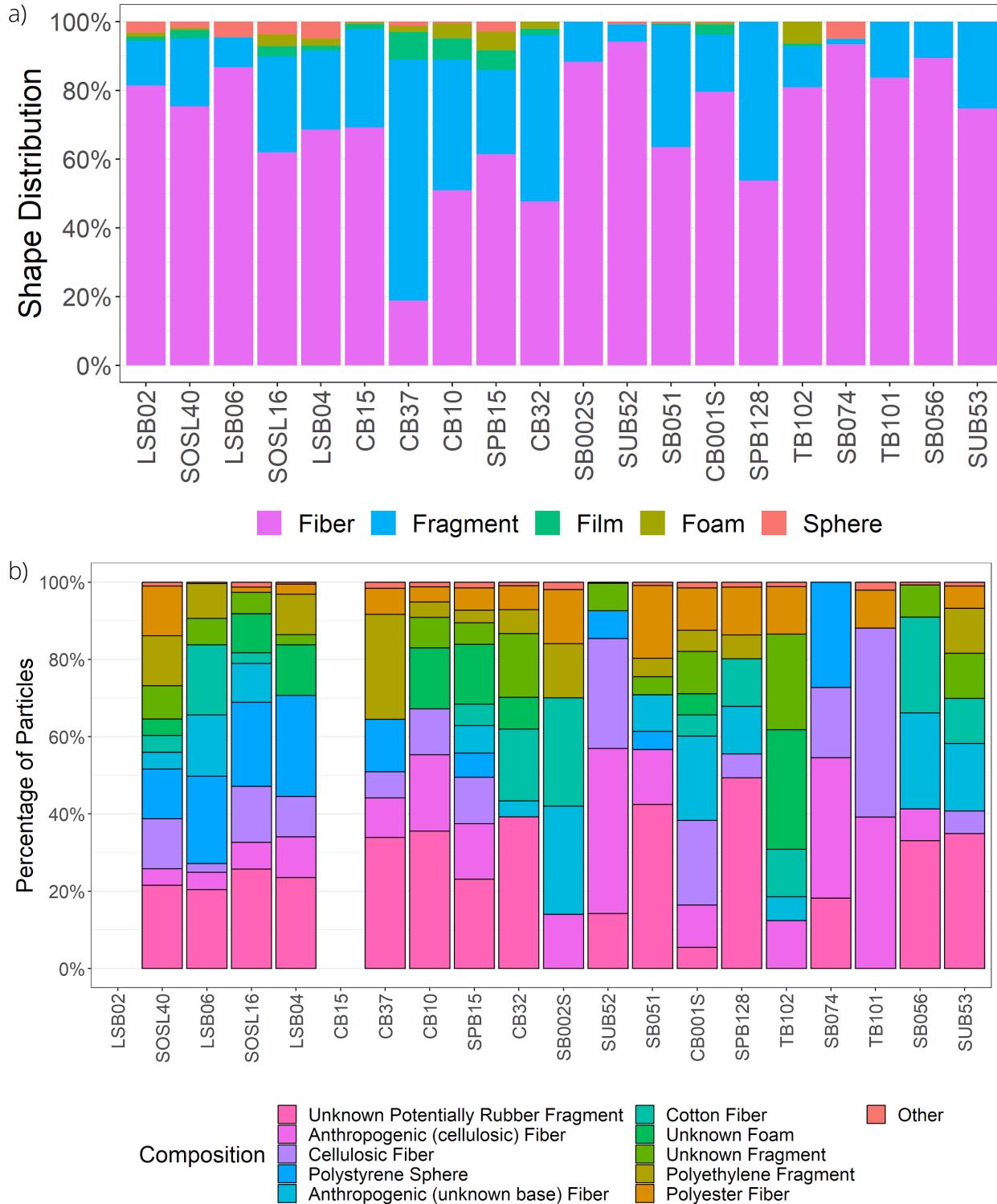


Figure 5.4. (a) Shape distribution and (b) polymer distribution of microparticles and microplastics quantified and analyzed at 20 locations in San Francisco Bay and reference sites (Tomales Bay). Composition “Other” includes all other combined polymers and morphologies. Microparticles from samples LSB02 and CB15 were not analyzed by spectroscopy.

FRAGMENT CONCENTRATIONS AND COMPOSITION

After fibers, fragments were the second most frequently detected morphology, and a total of 1,516 fragments were identified in field samples (Figure 5.4a). The highest measured concentration was 7.8 fragments/g dw, and the lowest concentration was 0.01 fragments/g dw (mean 1.7 fragments/g dw; median 0.9 fragments/g dw). The three lowest fragment concentrations were qualified because the particle counts were below the conservative data qualification threshold (Table A-5.3).

Twenty-one percent of the fragments were analyzed using Raman spectroscopy (Table A-5.4). The top plastic polymers identified were polyethylene (10% of fragments analyzed), polypropylene (7%), and polystyrene (6%; Table A-5.4). An additional 17% of fragments analyzed were identified as various other types of plastics, including 1% rubber. Rubber particles had Raman spectra that matched styrene-butadiene copolymer or styrene isoprene. Four percent were anthropogenic unknown as a result of dyes that interfered with the Raman spectrum.

Twenty-nine percent (429 particles) of the total number of fragments were black fragments that had a distinctive rubbery texture when handled with tweezers. One-hundred twenty-four of these particles were analyzed by Raman spectroscopy, but these black, rubbery particles were challenging to analyze because it was difficult to obtain a good spectrum and, therefore, polymer identification. The Raman spectra for these particles often matched to Carbon Black



or similar materials (Ivory Black, Diamond-like Carbon, Carbon, Van Dyke Brown, Vine Black). Some particle spectra had no discernible peaks or matches, or were similar to spectra from burnt carbon. Based on these identifications, it was challenging for the laboratory to conclusively identify these particles.

However, secondary characteristics such as compression, color, and texture suggested that these particles were similar to other black, rubbery particles that have been positively identified as rubber using FTIR spectroscopy. Other studies have also identified abundant presence of black fragments in the environment that may potentially come from car tires (Bråte et al., 2018; Gray et al., 2018; Unice et al., 2013). Therefore, based on this information, these black rubbery particles are classified as unknown potentially rubber (39% of fragments analyzed). Of note, carbon black is a major ingredient in tire rubber, so spectral matches to this material are not inconsistent with this classification (Edil, 2008).

Six percent of fragments analyzed were natural materials, including organic and inorganic natural material. The remaining 11% of fragments analyzed by spectroscopy could not be identified and were classified as unknown.

FILM CONCENTRATIONS AND COMPOSITION

A total of 140 film microparticles were identified in field samples, which is ten times less than the number of fragments. Concentrations of films ranged between zero and 0.9 microparticles/g dw (mean 0.2 microparticles/g dw; median 0.03 microparticles/g dw). Two of the lowest film concentrations were qualified because the one film particle detected in each of the samples was below the conservative threshold for film microparticles (1.3 films/sample).

Fifty-five percent of the 140 film particles were analyzed by spectroscopy, and 65% of those were confirmed to be plastic (Table A-5.4). The top plastic polymers identified were polystyrene (10 particles, 13% of film particles analyzed) and polyethylene (11 particles, 14% of film particles analyzed) (Table A-5.5). Six percent of the film particles analyzed were cellulosic or natural material.

FOAM CONCENTRATIONS AND COMPOSITION

Foam microparticles in sediment samples ranged between zero and 0.6 microparticles/g dw (mean 0.1 microparticles/g dw; median 0.01 microparticles/g dw). Because foam microparticles were not detected in blank samples, none of the foam data were qualified.

Sixty-seven percent of the 108 foam particles were further analyzed by spectroscopy, and 31% were confirmed to be plastic (Table A-5.4). The top plastic polymers were polystyrene (9 particles, 13% of foam particles analyzed) and polyethylene (5 particles, 7% of foam particles

analyzed; Table A-5.5). Thirteen percent of the foam particles analyzed were cellulosic or natural material.

SPHERE CONCENTRATIONS AND COMPOSITION

Sphere microparticles in sediment samples ranged between zero and 2 microparticles/g dw (mean 0.2 microparticles/g dw; median 0.02 microparticles/g dw). None of the sphere data were qualified because sphere particles were not detected in blank samples.

Fifty-five percent of the 128 spheres were analyzed by spectroscopy, and 74% were confirmed to be plastic. All of the plastic spheres were confirmed by spectroscopy to be made of polystyrene. Only one of the spheres analyzed was inorganic natural material (1% of sphere microparticles analyzed).

Regional variation

The sediment concentrations from the reference area were among the lowest concentrations measured, and the average concentration in the reference area was more than an order of magnitude lower than the average measured in San Francisco Bay (Figure 5.5). This comparison was true for both fiber and combined non-fiber microparticle concentrations. Two sites (out of 18) in the Bay had lower concentrations of fiber microparticles, and three sites had lower non-fiber concentrations compared to the reference area (Table A-5.3).

Concentrations of fiber and non-fiber microparticles at the two open Bay sites were lower than most Bay margins sites. The average fiber concentration at the open Bay sites was one-fourth of the average concentration at margin sites. Fiber results should be treated with caution because 60% of the data (12 out of 20 samples) were qualified because microfiber counts were below the conservative data qualification threshold for fibers. Average non-fiber concentrations at open Bay sites were nearly eight times lower than the average concentration in the margins. Only four margin sites (out of 16 margin sites) had lower concentrations than open Bay sites, and two of these data points were qualified (Table A-5.3). The reference site concentrations were also lower than open Bay sites. However, only two samples were analyzed from open Bay areas, while 16 samples were analyzed from the Bay margins. Because samples were not randomly chosen and there was a limited number of samples representing the open Bay, these results cannot be used to make strict spatial inferences for the Bay as a whole.



Sediment concentrations measured in Lower South Bay were higher than margin sites in the rest of the Bay (Mann Whitney U test, $p < 0.001$). The five highest fiber concentrations measured in the Bay were all in Lower South Bay. The second highest concentration (at SOSL40) is qualified because a very small sample mass was used (4.4 g), and the number of fibers enumerated in the sample was below the conservative threshold (178.1 fibers/sample). The four highest non-fiber sediment concentrations were also in Lower South Bay. These results suggest that Lower South Bay may have higher concentrations of microparticles compared to the rest of the Bay.

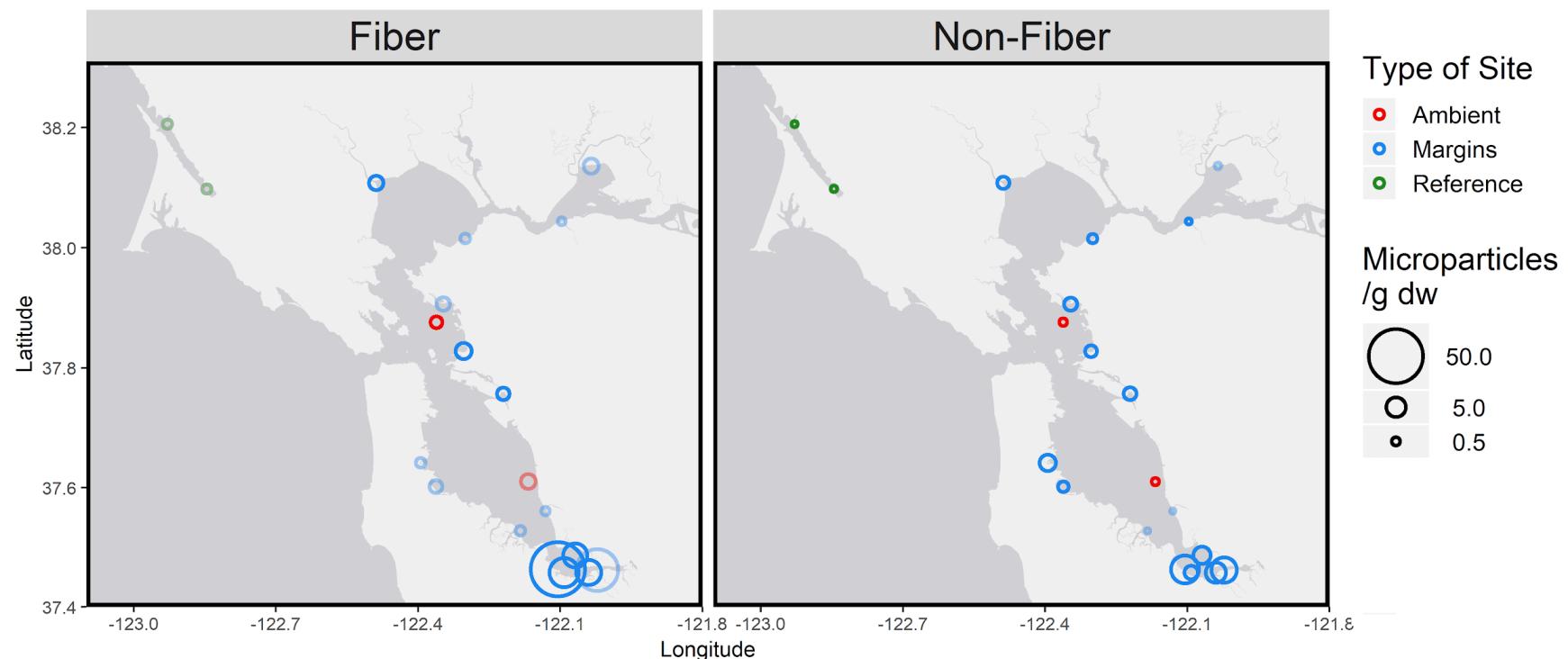


Figure 5.5. Microparticle concentrations measured in San Francisco Bay and the reference site, Tomales Bay, for fibers and non-fibers. The lighter outline represents qualified concentrations that were below conservative data qualification thresholds.

Discussion

Estimation of average microplastic concentrations in Bay sediment

We used this dataset to estimate the average microplastic concentrations in Bay sediment. The calculated average from this dataset is likely biased high relative to a true Bay-wide average because most of the selected sites were in margin areas where microparticle and microplastic concentrations are suspected to be higher than the rest of the Bay. Additionally, fiber results from 12 out of 20 of the sites were qualified because fiber counts were below the conservative data qualification threshold for fibers. The variability in the number of fibers detected in the laboratory and field blanks (0–143, n = 4) makes the microfiber counts less definitive. There were significantly fewer non-fiber microparticles in the laboratory and field blanks (0–8, n = 4), which indicated the level of background contamination of non-fiber microparticles is significantly less than that of fibers.

We estimated plastic fibers and non-fiber microplastics independently due to the disparity in blank contamination for these types of particles.

First, we estimated an upper and lower bound of the average number of microfibers in sediment samples. The upper bound was calculated after blank-correcting microfiber counts by the average of the laboratory and field blanks (40.3 microfibers/sample). The lower bound estimate was calculated after blank-correcting microfiber counts using the conservative data qualification threshold (178.1 microfibers/sample). Because not all microfibers were plastic, we then estimated the percentage of the microfibers in sediment likely to be plastic based on the composition of fibers that were analyzed by spectroscopy (Tables A-5.4 and A-5.5). An upper bound estimate of fiber concentrations in sediment was calculated using the percentage of fibers confirmed to be plastic (31%, Table A-5.4) and assuming that 60% of the anthropogenic unknown fibers (14%) were also plastic. The estimate of 60% of anthropogenic unknown fibers being plastic is based on the industry estimate that 60% of textiles today are made from nylon (polyamide) and polyester (Almroth et al., 2018). The lower bound uses only the percentage of particles confirmed to be plastic (Table A-5.4).

We also estimated an upper and lower bound of the average concentration of non-fiber microparticles in Bay sediment. The upper bound was calculated by blank-subtracting particle counts of each morphology (except fibers) using the average of the blanks for each morphology, and taking the average of the resulting total non-fiber microparticle concentrations. The lower bound was calculated similarly, but values were blank-corrected using the conservative data qualification threshold for each morphology. The amount of microplastics was estimated from the microparticle averages based on the estimated percentage of plastics using a similar methodology as described above for fibers. The upper

bound used the percentage of non-fiber microparticles confirmed to be plastic and assumed that 50% of anthropogenic unknown was plastic. The lower bound uses only the percentage of particles confirmed to be plastic (Table A-5.4).

The resulting estimate of the average concentration of fiber microplastics in Bay sediment was 0.8–2.2 microplastics/g dw. The resulting estimate of the average concentration of non-fiber microplastics in Bay sediment was 1.0–1.1 microplastics/g dw. The estimated average number of non-fiber microplastics was a narrower range than the fiber estimates because there was less background contamination and therefore less variation based on the upper and lower bound methods of blank correction.

Bay sediment microplastic concentrations were high relative to other regions

Comparison of the abundance of microplastics in Bay sediment with observations from other regions is difficult due to variation in methodology and reporting. Counts of microparticles in sediment have been reported as both wet and dry weights, as well as by mass, volume, and surface area. Without data on density and water content of the sediment, which are almost never reported, it is not possible to standardize units. This highlights the need for standard reporting practices. We only compared our results to other studies that reported microparticle abundances on a per dry weight mass basis (Table 5.3).

Almost all other studies used a smaller sieve size than the Bay study, which means the Bay results represent a conservative estimate of particle abundance compared to those made for other regions. Nevertheless, the averages and range of Bay sediment microparticle abundances were higher than those reported in the majority of other regions, often by several orders of magnitude.

Other studies also varied in terms of whether additional plastic confirmation methods were employed, and if so, whether reported microplastic concentrations were adjusted to account for particles that were identified as non-plastic. However, Bay microplastic abundances were also higher than microparticle counts reported in other regions. The exceptions to this were abundances reported in sediment from Lake Ontario (Ballent et al., 2016) and the Norwegian Sea (Haave et al., 2019), both of which included significantly smaller microparticles by using a smaller sieve size. If smaller microparticles were analyzed in Bay samples, it is likely that Bay abundances would be among the highest reported globally.

Fibers and fragments were consistently the dominant morphologies reported in sediment around the world, including in the Bay, although the dominant polymer types vary by region.

Table 5.3. Summary of microparticle concentrations measured in this study and in comparable studies around the world.

Reference	Location	Type of Aquatic System	Average Concentration (microparticles/kg d.w.)	Concentration Range (microparticles/kg d.w.)	Smallest Filter Size (µm)	Particles ID'd as Plastic	Dominant Morphology and Polymers	Polymer ID Method / Lab Blanks (Y/N)
Present Study	San Francisco Bay	estuarine	all particles: fibers 2,600 - 5,700; non-fibers 2,200 - 2,500 plastic estimate: fibers 800 - 2,200; non-fibers 1,000 - 1,100	all particles: fibers up to 49,000; non-fibers up to 11,000 plastic estimate: fibers up to 18,000; non-fibers up to 5,400	125	31% fibers, 39% fragments, 65% films, 74% spheres, 31% foams	fibers: polyester, cellulose acetate, acrylic; non-fibers: PS, PE, PP, PET	Raman / (Y)
Haave et al. 2019	Norwegian Sea	marine		48 to 211 12,000 to 200,000	500 0.7	188/429 not reported	fibers: Polyurethane/acrylate varnish resins, rubber types 3 and 1, PP, ethylene-vinyl-acetate, PA, polyester, PE, polychloroprene	FTIR / (Y)
Ballent et al. 2016	Lake Ontario	freshwater	760	20 to 28,000	0.053	67%	fibers and fragments PE, PS	Raman and X-ray fluorescence / (Y)
Qiu et al. 2015	South Sea and Beibu Gulf, China	marine beach		4,300 to 12,000	not reported	not reported	fibers, fragments, and films HDPE, PET, PE, PS	fluorescence microscopy and µ-FTIR / (N)
Vianello et al. 2013	Lagoon of Venice, Italy	marine		670 to 2,200	0.7	not reported	fragments and fibers PE, PP, poly(ethylene-propylene), polyester, polyacrylonitrile, PS, alkyd resin, PVC, PVA, nylon	µ-FTIR and ESEM-EDS / (Y)

Reference	Location	Type of Aquatic System	Average Concentration (microparticles/kg d.w.)	Concentration Range (microparticles/kg d.w.)	Smallest Filter Size (µm)	Particles ID'd as Plastic	Dominant Morphology and Polymers	Polymer ID Method / Lab Blanks (Y/N)
Wang et al. 2019	South Yellow Sea, China	marine		300 to 2,100	50	~95%	fibers, pellets, and fragments PP, PE, nylon, PS, PET	FTIR / (Y)
Alomar et al. 2016	Mediterranean sea	marine		100 ± 60 to 900 ± 100	63	n/a	fibers and fragments polymers not measured	just microscopy / (Y)
Claessens et al. 2011	Belgian coast	marine	170 ± 92	92 to 210	38	not reported	fibers: PP, nylon, PVA fragments: PP, PE, PS	FTIR / (N)
		marine beach	93 ± 37	53 to 130			fibers and fragments rayon, polyester, acrylic, PET, poly (ethylene:propylene:diene), PS	
Peng et al. 2017	Changjiang Estuary, China	estuary	120 ± 9	20 to 340	1	65 out of unknown total		μ-FTIR / (Y)
Frias et al. 2016	Portuguese coast	marine	10 ± 1	0 to 27	~1	not reported	fibers and fragments rayon	μ-FTIR / (Y)
Stolte et al. 2015	German Baltic coast	marine beach		14 to 530 fragments 0 to 7 fibers	55	not applicable	polymers not measured	just microscopy / (Y)
Tsang et al. 2017	Hong Kong	marine		44 to 460	0.7	110/240	fragments and pellets PP, HDPE, LDPE, PP-PE blend, styrene acrylonitrile	ATR-FTIR / (N)
		marine	15 ± 6	7 to 25	0.45	not reported	fibers and fragments PET, PP, PE, PA, cellophane, PVC, PS, LDPE	
Zheng et al. 2019	Jiaozhou Bay, China	estuary	25 ± 13	0 to 43			fibers and fragments PET, PE, rayon	ATR-μ-FTIR / (Y)

Microparticle concentrations and compositions in Bay margins

While care should be taken when extrapolating the results from this dataset to the whole Bay, the clearest trend was that the highest concentration of microparticles were measured in Lower South Bay. The limited flushing in Lower South Bay and the influence of wastewater and urban stormwater discharges may explain the higher concentrations measured in this subembayment (Smith and Hollibaugh, 2006). Previous studies have reported elevated concentrations of other contaminants in sediment from South and Lower South Bay compared to the rest of the Bay, including pharmaceuticals (Klosterhaus et al., 2013), triclosan (Kerrigan et al., 2015), perfluorooctane sulfonate (Sedlak et al., 2017), and specific alternative flame retardants (Sutton et al., 2019).

The four Lower South Bay sites had the highest levels of polystyrene in all morphologies, with concentrations at these sites ranging between 0.4 and 1.1 microplastics/g dw. Polystyrene microparticles were detected at 14 of 18 Bay sites, and were not detected in samples from Tomales Bay reference sites. Concentrations of polystyrene microparticles at all Bay margin sites ranged between zero and 1.1 microplastics/g dw, with a median of 0.04 microplastics/g dw. Concentrations measured at the two open Bay sites were 0.04 and 0.01 microplastics/g dw, respectively, which were within the range measured at the margin sites.

Unknown potentially rubber fragments were only identified in Bay margins samples, and were not detected at open Bay nor Tomales Bay reference sites. The concentration of these unknown potentially rubber fragments ranged between zero and 0.2 microparticles/g dw. Concentrations of 0.2 microparticles/g dw of these rubbery microparticles were not unique to sites near stormwater discharges.

Apart from these observations, there were no clear differences in the microparticle morphologies or microplastic polymers identified at open Bay sites relative to Bay margins. Likewise, morphologies and polymers identified at the reference sites were not significantly different from those identified at Bay sites. These comparisons were limited by the number of sites from the open Bay ($n = 2$) and the reference area ($n = 2$). Overall, the concentrations of microparticles measured at the open Bay sites and Tomales Bay reference sites were in the low range of values measured at the margin Bay sites, despite the overall similarity in distribution of morphologies and polymers. Polyester, acrylic, and cellulose acetate fibers were detected in a majority of the field samples.

Potential sources of microplastics in sediment

Identifying potential sources of microplastics allows better prioritization of management actions. The morphology and chemical composition of individual microparticles can provide indications of their source. The sediment samples contained a variety of microparticles that

were also commonly detected in wastewater, stormwater, and surface water samples, and for which tentative source identification was possible.

FIBERS

Fibers were abundant and ubiquitous across all field samples. Thirty percent of the fibers analyzed were confirmed to be plastic, and the most common polymers identified were polyester (28% of fibers analyzed), cellulose acetate (18%), and acrylic (14%, Table A-5). Polyester and acrylic fibers were among the most abundant types of fibers identified in wastewater and surface water samples. Cellulose acetate fibers were frequently detected in stormwater samples, in addition to polyester. The highest concentration of microfibers in sediment from this study was measured near a wastewater discharge location in Lower South Bay.

The potential sources of fibers are diverse. Fibers released from synthetic textiles are a known source of plastic microfibers to the environment (Almroth et al., 2018; Bomgardner, 2017; Browne et al., 2011; Bruce et al., 2016; Gustafsson et al., 2019; Henry et al., 2019; Hernandez et al., 2017; McIlwraith et al., 2019; Pirc et al., 2016). Among the types of fibers observed, polyester and acrylic are commonly used in textiles (Almroth et al., 2018). Microfibers shed from textiles can become airborne, and were also a common source of blank contamination in this study. Airborne fibers can be transported to the Bay through stormwater or direct settling to Bay surface water. Microfibers are also shed during washing and drying of synthetic textiles and have been frequently identified in wastewater from studies in other locations (Almroth et al., 2018; Browne et al., 2011; Hernandez et al., 2017; Mintenig et al., 2017; Murphy et al., 2016; Wolff et al., 2019). Microfibers in wastewater can be released into the environment through final effluent and land-application of biosolids (Gustafsson et al., 2019; Habib et al., 1998; He et al., 2019; Nizzetto et al., 2016; Zubris and Richards, 2005). Microfibers deposited after release from clothing dryer vents (Pirc et al., 2016) onto nearby impervious surfaces can be transported via urban stormwater to the environment.

Fishing and other aquatic industries represent an additional input of fibers to surface waters. Fishing nets and lines are primarily made of polyethylene (and nylon) and may also be a source of fiber pollution. Cellulose acetate is used in textiles, but also in cigarette filters and diapers. Cigarette filters are among the top ten plastic litter items identified in California and nationwide, and diapers are also a common litter item (Allen et al., 2017; Eriksen et al., 2016)

FRAGMENTS

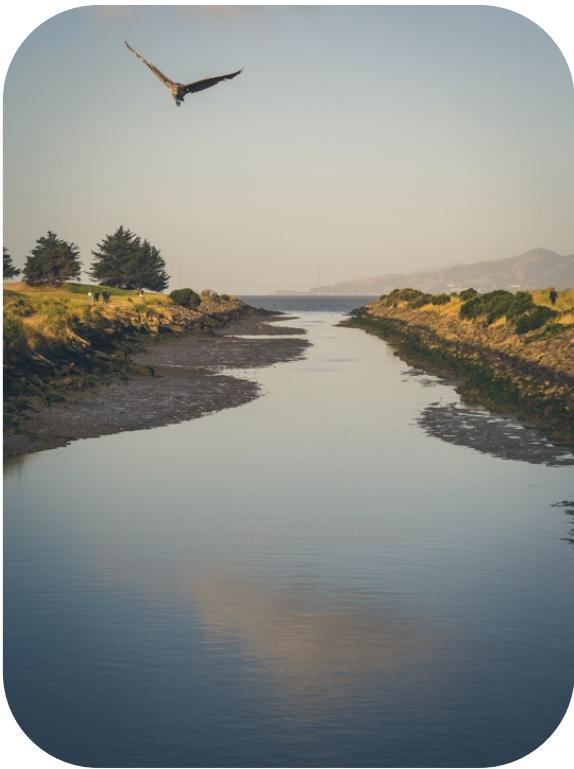
Black fragments that had a rubbery texture comprised 23% of non-fiber microparticles. Particles that were visually similar and had similar secondary characteristics (rubbery texture)

were also a dominant type of microparticle identified in stormwater samples, but were not common in wastewater and surface water samples.

These particles were challenging to identify with Raman spectroscopy because of fluorescence from other compounds, or because the particle burned. In addition, a majority of these particles had a minimum dimension of less than 200 µm, which made them challenging to analyze using FTIR spectroscopy. As a result, in several instances, the laboratory analysts were not able to positively confirm the polymer type of these particles. However, secondary characteristics such as compression, color, and texture suggested that these particles were similar to the same type of black, rubbery particles that have been previously identified as rubber using FTIR spectroscopy.

Tire wear is a suspected source of these rubbery microparticles. Tire wear has been connected to abundant rubber microparticles identified in sediment samples in other studies (Gray et al., 2018; Unice et al., 2013). Modeling studies indicate tire wear may be one of the top sources of microplastics to the environment globally (Boucher and Friot, 2017; Kole et al., 2017). Other studies have successfully identified rubber particles in environmental samples using thermoextraction desorption gas chromatography mass spectrometry (TED-GC-MS) and pyrolysis-GC-MS methods (Eisentraut et al., 2018; Unice et al., 2013). Tire and road wear particles were detected in 97% of sediment samples in another study comparing urban watersheds in the Seine (France), Chesapeake (U.S.), and Yodo-Lake Biwa (Japan); average concentrations were 4,500 µg/g dw (Unice et al., 2013). Measured tire wear particles are mostly smaller than 100 µm (Kole et al., 2017), which is smaller than the smallest sieve size fraction analyzed in this study (125 µm), so the present study likely undercounts the number of tire wear microparticles present in sediment. Rubber particles are denser than seawater, which may explain why these particles were not commonly identified in surface water samples. Additionally, surface water samples used a larger minimum sieve size (355 µm), which was larger than the maximum dimension of most of these particles.

Polyethylene, polypropylene, and polystyrene fragments were also frequently identified in sediment samples (Table A-5.5). These polymers are commonly used in single-use items such as packaging and foodware, as well as a plethora of other items that may end up as trash in the Bay such as toys, appliances, furniture, and construction materials. Polyethylene and polypropylene fragments were also identified in surface water, stormwater, and wastewater samples. Polyethylene and polypropylene fragments intentionally added to household and personal care products (i.e., microbeads) are also a potential source of these microplastics that could be transported to the Bay through wastewater. Breakdown of materials used in the aquaculture and fishing industries (e.g., buoys, ropes, spacers) can also contribute microplastic



fragments to the environment (Andrady, 2011; Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, 2015).

SPHERES

Sixty-five percent of the sphere microparticles in sediment were clear, and 96% of the clear spheres examined with spectroscopy were made of polystyrene. All of the measured spheres were 125–700 µm in size, half of which were 200 µm or smaller. These spheres identified in sediment samples were much smaller than the majority of spheres identified in surface water samples, which were greater than 1 mm in size. Spheres were identified in 15 out of 18 Bay sediment samples. The highest concentration of spheres was measured at LSB02, and the concentration of spheres at Lower South Bay sites ranged between

0.5–2.0 microparticles/g dw. This range was over an order of magnitude higher than the median sphere concentration in the whole dataset, 0.02 microparticles/g dw.

Polystyrene microbeads are sometimes used for ion-exchange in water softening and other water purification treatments, as well as for other industrial applications; other studies have identified these applications as potential sources of microplastic pollution (Ballent et al., 2016; Mani et al., 2019). Clear polystyrene microspheres (less than 1 mm) are also frequently used in biomedical and biotechnology applications and laboratory research (Ma et al., 2013). These spheres are used in everything from microbead-based assays to research studies and clinical applications due to their use in cell sorting, cell separation, immunoprecipitation, protein binding, magnetic separation, chromatography, and next generation sequencing procedures. Therefore, another possible source of the high abundance of small clear polystyrene spheres is the high concentration of bioscience companies in the Bay Area (Diehl, 2018).

Microbeads that are added to personal care and cleaning products are another potential source of the polystyrene spheres (Rochman et al., 2015), although previous studies indicate most facial cleansers use microbeads made of polyethylene (Chang, 2015; Fendall and Sewell, 2009). Recent federal legislation has banned the sale of wash-off personal care products containing microbeads, and spheres detected in sediment may come from products used or purchased before the ban was implemented.

Conclusions

This study sought to quantify the abundance of microparticles and microplastics in Bay sediment, characterize the type and chemical composition of microparticles found in sediment, identify sites or regions of particular concern, and compare results with other studies. This was the first study to measure microparticles and microplastics in Bay sediment.

Microparticles were present in all sediment samples from the Bay and from the reference site, Tomales Bay. Fibers were the most abundant type of microparticle in sediment samples—1.5–49 microfibers/g dw—and eight out of 20 samples were above the data qualification threshold for fibers of 178 fibers/sample. Non-fiber microparticles were also present in all samples. Only three of the non-fiber microparticle concentrations were qualified based on the data qualification threshold. Non-fiber microparticle concentrations ranged from 0.05 to 11 microparticles/g dw.

A subset of microparticles were analyzed using Raman spectroscopy to estimate the fraction of microparticles that were plastic. An upper and lower bound method was used to estimate the average concentration of microplastics in sediment based on the composition of particles that underwent spectroscopy and accounting for background contamination through blank subtraction. The average concentration of microplastic fibers at the sampled sites in the Bay was between 0.8 and 2.2 microplastics/g dw, while the average concentration of non-fibers was between 1.0 and 1.1 microplastics/g dw.

Lower South Bay sites had higher concentrations of microparticles compared to the rest of the Bay. The averages and range of Bay sediment microparticle abundances were higher than those reported in the majority of other regions, often by orders of magnitude. While sampled sites in the present study may be biased high because most of the sites were near stormwater and wastewater discharge sites, the present study also likely undercounted microparticles in Bay sediment relative to studies in other regions because a larger sieve mesh size was used to quantify microparticles.

Black, rubbery fragments that were prevalent in stormwater samples were also detected in sediment samples. The polymer identity of these rubbery fragments could not be definitively confirmed because these particles were challenging to analyze by Raman spectroscopy. These particles are suspected to be rubber, which has also been reported in sediment from other studies.

References

- Allen, K., Cohen, D., Culver, A., Cummins, A., Curtis, S., Eriksen, M., Gordon, M., Howe, A., Lapis, N., Prindiville, M., Thorpe, B., Wilson, S., 2017. Better Alternatives Now B.A.N. List 2.0. 5 Gyres, Algalita, Californians Against Waste, Clean Production Action, Plastic Pollution Coalition, Responsible Purchasing Network Story of Stuff, Surfrider Foundation, UPSTREAM.
- Almroth, B.M.C., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles: A source of microplastics released into the environment. *Environmental Science Pollution Research* 25, 1191–1199.
<https://doi.org/10.1007/s11356-017-0528-7>
- Alomar, C., Estarellas, F., Deudero, S., 2016. Microplastics in the Mediterranean Sea: Deposition in coastal shallow sediments, spatial variation and preferential grain size. *Marine Environmental Research* 115, 1–10. <https://doi.org/10.1016/j.marenvres.2016.01.005>
- Anderson, J.C., Park, B.J., Palace, V.P., 2016. Microplastics in aquatic environments: Implications for Canadian ecosystems. *Environmental Pollution* 218, 269–280.
<https://doi.org/10.1016/j.envpol.2016.06.074>
- Andrady, A.L., 2011. Microplastics in the marine environment. *Marine Pollution Bulletin* 62, 1596–1605. <https://doi.org/10.1016/j.marpolbul.2011.05.030>
- Applied Marine Sciences, 2014. 2014 RMP Sediment Cruise Report (Contract No. 1084).
- Ballent, A., Corcoran, P.L., Madden, O., Helm, P.A., Longstaffe, F.J., 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. *Marine Pollution Bulletin* 110, 383–395. <https://doi.org/10.1016/j.marpolbul.2016.06.037>
- Bomgardner, M.M., 2017. The great lint migration. *Chemical & Engineering News* 95, 16–17.
- Boucher, J., Friot, D., 2017. Primary Microplastics in the Oceans: A Global Evaluation of Sources. Gland, Switzerland: International Union for Conservation of Nature (IUCN), 43pp.
- Bråte, I.L.N., Hurley, R., Iversen, K., Beyer, J., Thomas, K.V., Steindal, C.C., Green, N.W., Olsen, M., Lusher, A., 2018. *Mytilus spp.* as sentinels for monitoring microplastic pollution in Norwegian coastal waters: A qualitative and quantitative study. *Environmental Pollution* 243, 383–393.
<https://doi.org/10.1016/j.envpol.2018.08.077>
- Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: Sources and sinks. *Environmental*

Chapter 5—Sediment

Science & Technology 45, 9175–9179. <https://doi.org/10.1021/es201811s>

Bruce, N.J., Hartline, N.L., Karba, S.N., Ruff, B., Sonar, S.U., 2016. Microfiber pollution and the apparel industry (Master's Thesis). Bren School of Environmental Science & Management, University of California, Santa Barbara.

Chang, M., 2015. Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. Marine Pollution Bulletin 101, 330–333. <https://doi.org/10.1016/j.marpolbul.2015.10.074>

Claessens, M., Meester, S.D., Landuyt, L.V., Clerck, K.D., Janssen, C.R., 2011. Occurrence and distribution of microplastics in marine sediments along the Belgian coast. Marine Pollution Bulletin 62, 2199–2204. <https://doi.org/10.1016/j.marpolbul.2011.06.030>

Diehl, P., 2018. Largest U.S. Biotech Hubs: Boston and the San Francisco Bay Area [WWW Document]. The Balance. URL <https://www.thebalance.com/boston-and-san-francisco-biotech-hubs-375641> (accessed 8.13.19).

Edil, T.B., 2008. A review of environmental impacts and environmental applications of shredded scrap tires, in: Hazarika, H., Yasuhara, K. (Eds.), Scrap Tire Derived Geomaterials - Opportunities and Challenges. Taylor & Francis Group, London, pp. 3–18.

Eisentraut, P., Dümichen, E., Ruhl, A.S., Jekel, M., Albrecht, M., Gehde, M., Braun, U., 2018. Two birds with one stone—Fast and simultaneous analysis of microplastics: Microparticles derived from thermoplastics and tire wear. Environmental Science & Technology Letters 5, 608–613. <https://doi.org/10.1021/acs.estlett.8b00446>

Eriksen, M., Prindiville, M., Thorpe, B., 2016. The Plastics BAN List. 5 Gyres, Surfrider Foundation, Clean Production Action, UPSTREAM.

European Commission Joint Research Center, 2013. Guidance on monitoring of marine litter in European seas. Publications Office, Luxembourg.

Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face: Microplastics in facial cleansers. Marine Pollution Bulletin 58, 1225–1228.
<https://doi.org/10.1016/j.marpolbul.2009.04.025>

Frias, J.P.G.L., Gago, J., Otero, V., Sobral, P., 2016. Microplastics in coastal sediments from Southern Portuguese shelf waters. Marine Environmental Research 114, 24–30.
<https://doi.org/10.1016/j.marenvres.2015.12.006>

Chapter 5—Sediment

Gray, A.D., Wertz, H., Leads, R.R., Weinstein, J.E., 2018. Microplastic in two South Carolina estuaries: Occurrence, distribution, and composition. *Marine Pollution Bulletin* 128, 223–233. <https://doi.org/10.1016/j.marpolbul.2018.01.030>

Gustafsson, R.E., Hagman, H., Lindberg, K., Rehnberg, F., 2019. Microplastic emissions from domestic laundry (Bachelor's Thesis). Institution of Architecture and Civil Engineering Division of Water Environment Technology Chalmers University of Technology, Gothenburg, Sweden.

Haave, M., Lorenz, C., Primpke, S., Gerdts, G., 2019. Different stories told by small and large microplastics in sediment - First report of microplastic concentrations in an urban recipient in Norway. *Marine Pollution Bulletin* 141, 501–513. <https://doi.org/10.1016/j.marpolbul.2019.02.015>

Habib, D., Locke, D.C., Cannone, L.J., 1998. Synthetic fibers as indicators of municipal sewage sludge, sludge products, and sewage treatment plant effluents. *Water, Air, & Soil Pollution* 103, 1–8. <https://doi.org/10.1023/A:1004908110793>

He, P., Chen, L., Shao, L., Zhang, H., Lü, F., 2019. Municipal solid waste (MSW) landfill: A source of microplastics? -Evidence of microplastics in landfill leachate. *Water Research* 159, 38–45. <https://doi.org/10.1016/j.watres.2019.04.060>

Henry, B., Laitala, K., Klepp, I.G., 2019. Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. *Science of The Total Environment* 652, 483–494. <https://doi.org/10.1016/j.scitotenv.2018.10.166>

Hernandez, E., Nowack, B., Mitrano, D.M., 2017. Polyester textiles as a source of microplastics from households: A mechanistic study to understand microfiber release during washing. *Environmental Science & Technology* 51, 7036–7046. <https://doi.org/10.1021/acs.est.7b01750>

Hurley, R., Woodward, J., Rothwell, J.J., 2018. Microplastic contamination of river beds significantly reduced by catchment-wide flooding. *Nature Geoscience* 11, 251–257. <https://doi.org/10.1038/s41561-018-0080-1>

Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection, 2015. Sources, Fate and Effects of Microplastics in the Marine Environment: A Global Assessment.

Kerrigan, J.F., Engstrom, D.R., Yee, D., Sueper, C., Erickson, P.R., Grandbois, M., McNeill, K., Arnold, W.A., 2015. Quantification of hydroxylated polybrominated diphenyl ethers (OH-BDEs), triclosan, and related compounds in freshwater and coastal systems. *PLOS ONE* 10, e0138805. <https://doi.org/10.1371/journal.pone.0138805>

Chapter 5—Sediment

Klosterhaus, S.L., Grace, R., Hamilton, M.C., Yee, D., 2013. Method validation and reconnaissance of pharmaceuticals, personal care products, and alkylphenols in surface waters, sediments, and mussels in an urban estuary. *Environment International* 54, 92–99. <https://doi.org/10.1016/j.envint.2013.01.009>

Kole, P.J., Löhr, A.J., Van Belleghem, F., Ragas, A., 2017. Wear and Tear of Tyres: A Stealthy Source of Microplastics in the Environment. *IJERPH* 14, 1265. <https://doi.org/10.3390/ijerph14101265>

Ma, G., Su, Z.-G., Su, Z.-G., 2013. Microspheres and Microcapsules in Biotechnology: Design, Preparation and Applications. Jenny Stanford Publishing. <https://doi.org/10.1201/b14540>

Mani, T., Blarer, P., Storck, F.R., Pitroff, M., Wernicke, T., Burkhardt-Holm, P., 2019. Repeated detection of polystyrene microbeads in the Lower Rhine River. *Environmental Pollution* 245, 634–641. <https://doi.org/10.1016/j.envpol.2018.11.036>

McIlwraith, H.K., Lin, J., Erdle, L.M., Mallos, N., Diamond, M.L., Rochman, C.M., 2019. Capturing microfibers – Marketed technologies reduce microfiber emissions from washing machines. *Marine Pollution Bulletin* 139, 40–45. <https://doi.org/10.1016/j.marpolbul.2018.12.012>

Mintenig, S.M., Int-Veen, I., Löder, M.G.J., Primpke, S., Gerdts, G., 2017. Identification of microplastic in effluents of waste water treatment plants using focal plane array-based micro-Fourier-transform infrared imaging. *Water Research* 108, 365–372. <https://doi.org/10.1016/j.watres.2016.11.015>

Murphy, F., Ewins, C., Carbonnier, F., Quinn, B., 2016. Wastewater treatment works (WwTW) as a source of microplastics in the aquatic environment. *Environmental Science & Technology* 50, 5800–5808. <https://doi.org/10.1021/acs.est.5b05416>

Nizzetto, L., Futter, M., Langaas, S., 2016. Are agricultural soils dumps for microplastics of urban origin? *Environmental Science & Technology* 50, 10777–10779. <https://doi.org/10.1021/acs.est.6b04140>

Peng, G., Zhu, B., Yang, D., Su, L., Shi, H., Li, D., 2017. Microplastics in sediments of the Changjiang Estuary, China. *Environmental Pollution* 225, 283–290. <https://doi.org/10.1016/j.envpol.2016.12.064>

Pirc, U., Vidmar, M., Mozer, A., Kržan, A., 2016. Emissions of microplastic fibers from microfiber fleece during domestic washing. *Environmental Science Pollution Research* 23, 22206–22211. <https://doi.org/10.1007/s11356-016-7703-0>

Chapter 5—Sediment

- Qiu, Q., Peng, J., Yu, X., Chen, F., Wang, J., Dong, F., 2015. Occurrence of microplastics in the coastal marine environment: First observation on sediment of China. *Marine Pollution Bulletin* 98, 274–280. <https://doi.org/10.1016/j.marpolbul.2015.07.028>
- Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Correction to Scientific evidence supports a ban on microbeads. *Environmental Science & Technology* 49, 14740–14740. <https://doi.org/10.1021/acs.est.5b05043>
- Sedlak, M.D., Benskin, J.P., Wong, A., Grace, R., Greig, D.J., 2017. Per- and polyfluoroalkyl substances (PFASs) in San Francisco Bay wildlife: Temporal trends, exposure pathways, and notable presence of precursor compounds. *Chemosphere* 185, 1217–1226. <https://doi.org/10.1016/j.chemosphere.2017.04.096>
- Sedlak, M., Sutton, R., Box, C., Sun, J., Lin, D., 2017. FINAL Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries. SFEI Contribution No. 819. San Francisco Estuary Institute and 5 Gyres, Richmond, CA.
- Shimabuku, I., Sun, J., Trowbridge, P., 2017. 2017 RMP Field Sampling Report. SFEI Contribution No. 849. San Francisco Estuary Institute, Richmond, CA.
- Smith, S.V., Hollibaugh, J.T., 2006. Water, salt, and nutrient exchanges in San Francisco Bay. *Limnology and Oceanography*. 51, 504–517. https://doi.org/10.4319/lo.2006.51.1_part_2.0504
- Stolte, A., Forster, S., Gerdts, G., Schubert, H., 2015. Microplastic concentrations in beach sediments along the German Baltic coast. *Marine Pollution Bulletin* 99, 216–229. <https://doi.org/10.1016/j.marpolbul.2015.07.022>
- Sutton, R., Chen, D., Sun, J., Greig, D.J., Wu, Y., 2019. Characterization of brominated, chlorinated, and phosphate flame retardants in San Francisco Bay, an urban estuary. *Science of The Total Environment* 652, 212–223. <https://doi.org/10.1016/j.scitotenv.2018.10.096>
- Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay. SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA.
- Tsang, Y.Y., Mak, C.W., Liebich, C., Lam, S.W., Sze, E.T.-P., Chan, K.M., 2017. Microplastic pollution in the marine waters and sediments of Hong Kong. *Marine Pollution Bulletin* 115, 20–28. <https://doi.org/10.1016/j.marpolbul.2016.11.003>
- Unice, K.M., Kreider, M.L., Panko, J.M., 2013. Comparison of tire and road wear particle

Chapter 5—Sediment

concentrations in sediment for watersheds in France, Japan, and the United States by quantitative pyrolysis GC/MS analysis. *Environmental Science & Technology* 130710100101002. <https://doi.org/10.1021/es400871j>

Vianello, A., Boldrin, A., Guerriero, P., Moschino, V., Rella, R., Sturaro, A., Da Ros, L., 2013. Microplastic particles in sediments of Lagoon of Venice, Italy: First observations on occurrence, spatial patterns and identification. *Estuarine, Coastal and Shelf Science* 130, 54–61. <https://doi.org/10.1016/j.ecss.2013.03.022>

Wang, J., Wang, M., Ru, S., Liu, X., 2019. High levels of microplastic pollution in the sediments and benthic organisms of the South Yellow Sea, China. *Science of The Total Environment* 651, 1661–1669. <https://doi.org/10.1016/j.scitotenv.2018.10.007>

Wolff, S., Kerpen, J., Prediger, J., Barkmann, L., Müller, L., 2019. Determination of the microplastics emission in the effluent of a municipal waste water treatment plant using Raman microspectroscopy. *Water Research X* 2, 100014.
<https://doi.org/10.1016/j.wroa.2018.100014>

Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution* 178, 483–492.
<https://doi.org/10.1016/j.envpol.2013.02.031>

Yee, D., Franz, A., Trowbridge, P., Wong, A., 2018. Quality Assurance Program Plan for The Regional Monitoring Program for Water Quality in San Francisco Bay. SFEI Contribution No. 890. San Francisco Estuary Institute, Richmond, CA.

Yee, D., Wong, A., Shimabuku, I., Trowbridge, P., 2017. Characterization of Sediment Contamination in Central Bay Margin Areas. SFEI Contribution No. 829. San Francisco Estuary Institute, Richmond, CA.

Zheng, Y., Li, J., Cao, W., Liu, X., Jiang, F., Ding, J., Yin, X., Sun, C., 2019. Distribution characteristics of microplastics in the seawater and sediment: A case study in Jiaozhou Bay, China. *Science of The Total Environment* 674, 27–35. <https://doi.org/10.1016/j.scitotenv.2019.04.008>

Zubris, K.A.V., Richards, B.K., 2005. Synthetic fibers as an indicator of land application of sludge. *Environmental Pollution* 138, 201–211. <https://doi.org/10.1016/j.envpol.2005.04.013>

CHAPTER

6

Microparticles and Microplastics

IN SMALL PREY FISH

: by Diana Lin



Highlights

- ◆ Northern anchovy (*Engraulis mordax*) and topsmelt (*Atherinops affinis*) were collected at multiple sites in San Francisco Bay and at a reference area with minimal urban influence. Fish guts were digested whole in potassium hydroxide (KOH), filtered through a 10 µm polycarbonate filter, and analyzed for microparticles and microplastics down to 20 µm.
- ◆ These two prey fish species were chosen to evaluate whether differences in preferred habitat and foraging behavior affected microparticle concentrations. While there was no statistically significant difference in fiber counts between the two species, there was a statistically significant difference in non-fiber microparticle counts, with topsmelt having higher levels than anchovies.
- ◆ Microparticle levels in fish from San Francisco Bay were statistically higher than microparticle levels in fish from the reference area with minimal urban influence (Tomales Bay).
- ◆ At least 38% of fish from the Bay had microparticle levels above the threshold for data qualification determined from laboratory blanks. The estimated average number of non-fiber microplastics/fish in Bay anchovies and topsmelt was 0.2–0.9 microplastics/fish. The estimated average number of fiber microplastics/fish in Bay anchovies and topsmelt was 0.6–4.5 microplastics/fish. The microplastic counts in the Bay were comparable to counts reported in many other locations.
- ◆ Microfibers were the predominant microparticle morphology in fish samples from the Bay, representing 86% of microparticles present in fish. Of the fibers that were further analyzed via Raman spectroscopy, 23% were confirmed to be plastic, while 60% were classified as anthropogenic unknown because dyes embedded in the microfibers interfered with the laboratory's ability to identify the composition. Twenty-one percent of non-fiber microparticles analyzed by spectroscopy were confirmed to be plastic.
- ◆ Particles smaller than 150 µm represented 16% of fragments and 6% of fibers observed in fish samples. Particles in this size fraction have the potential to translocate out of the gut and bioaccumulate.

Objectives

The goal of this element of the San Francisco Bay Microplastics Project was to characterize microparticles and microplastics in the digestive tracts of prey fish collected in and around San Francisco Bay. Microplastics are a subset of microparticles and microfibers that have been definitively determined as plastic through spectroscopy or other means. Many studies in the literature identify microparticles that appear to be plastic using only visual techniques, such as microscopy. In this report, we refer to particles (< 5 mm) identified visually as microparticles; particles that have been confirmed to be plastic through Raman or Fourier Transform Infrared (FTIR) spectroscopy are referred to as microplastics.

Prey fish may ingest microplastics through passive filtration or active consumption when the microplastics are mistaken for food (Peters et al., 2017). Prey fish are desirable for monitoring microplastic contamination for several reasons.

- ◆ Prey fish serve as indicators of the bioavailability of microplastics in the environment.
- ◆ Prey fish exhibit relatively high site fidelity, allowing for comparisons among subembayments in the Bay. In some cases, correlations can be made between concentrations of contaminants in sediment and concentrations in prey fish, as has been demonstrated for PCBs (Greenfield and Allen, 2013).
- ◆ Prey fish have short life spans (1–2 years), providing a snapshot of conditions in time.
- ◆ Prey fish are important food sources for piscivorous fish, birds, and marine mammals, and provide an index of exposure to higher trophic levels.

Anchovies and topsmelt (Table 6.1) are important pelagic prey fish in the Bay food web, and are a key food source for larger predators, including sport fish, marine mammals, and seabirds. Anchovies are the most abundant fish in San Francisco Bay, especially in the higher salinity waters of South Bay (Kimmerer, 2015). Although not as numerous as anchovies, topsmelt have been used as a biosentinel species for monitoring contaminants in the Bay due to their restricted home range, nearshore habitat, and wide distribution throughout the Bay (Greenfield and Jahn, 2010; Greenfield and Allen, 2013). Tops melt can tolerate a large range of salinities; juveniles are typically in freshwater and move toward saline water as they age. Anchovies and topsmelt also have different feeding habits, providing an opportunity to test the hypothesis that fish at the same trophic level with different foraging strategies ingest different amounts and types of microplastics. Anchovies graze on zooplankton and phytoplankton, crustaceans, fish eggs, and larvae throughout the water column, using a combination of filter-feeding and visual hunting. In contrast, topsmelt tend to graze near the bottom of the water column, and are opportunistic feeders of pelagic invertebrates, diatoms, and plants.

Table 6.1. Species profiles for two species of prey fish in San Francisco Bay that were analyzed for microplastics in their guts.

		
Species	Topsmelt (<i>Atherinops affinis</i>)	Northern Anchovy (<i>Engraulis mordax</i>)
Importance	Used as biosentinel species for monitoring uptake of contaminants in the Bay, such as mercury and PCBs.	Most abundant fish species in the Bay. Critical prey species for higher trophic piscivore species (e.g., fish, birds, marine mammals).
Preferred Location/Habitat	Bay margins: Preference for shallow bays, sloughs, and embayments.	Bay open channels: Preference for more pelagic conditions. Sub-populations move between the Bay and the ocean.
Diet	Primarily benthic and pelagic invertebrates, diatoms, and plants.	Throughout water column — plankton, zooplankton, crustaceans, fish eggs, larvae.

Through assessing microparticle and microplastic abundance and characteristics in topsmelt and anchovy collected from San Francisco Bay, this study sought to accomplish the following objectives.

1. Quantify the abundance of microparticles and microplastics in prey fish. Understanding the abundance of microplastics in prey fish (Management Question [MQ]1: How much microplastic pollution is there in the Bay?) is important for evaluating risk to prey fish, as well as evaluating the potential risk to higher trophic level organisms such as sport fish and humans (MQ2: What are the health risks?).
2. Characterize types of microparticles and microplastics found in prey fish and their chemical composition. Understanding the types of microparticles and microplastics found in prey fish will help determine the likely sources of the particles prey fish consume. Fish may exhibit preferences for specific particle sizes or chemical compositions based on ecological niche, foraging patterns, and dietary preferences. Such patterns could help to inform mitigation measures (MQ5: Which management actions may be effective in reducing microplastic pollution?) that could most directly contribute to a reduction of microplastics in fish.

3. Identify sites of particular concern (high exposure) at a regional and local scale and provide a foundation for tracking interannual trends. This study provides a baseline against which future data may be compared (MQ4: Have the concentrations of microplastics in the Bay increased or decreased?). This will also be important in assessing which management actions may be most effective (MQ5).
4. Compare fish microparticle and microplastic concentrations in the Bay with results from studies in other regions. The results from this study will be placed in context with the literature. Understanding how Bay microparticle and microplastic levels compare with studies from other regions can help motivate the need and urgency for management actions in the Bay (MQ5).

This study was designed to test the following hypotheses (Sedlak et al., 2017):

- ◆ Microplastics will be present in fish.
- ◆ Fish from reference sites will have lower concentrations of microplastics.
- ◆ Microplastic concentrations will vary as a function of species. Species that reside in more open areas of the Bay (e.g., anchovy) will have a different microplastic composition than fish that reside largely in the margins of the Bay (e.g., topsmelt).

We had originally proposed to also compare the concentrations and distribution of microplastics in prey fish and sediment from the same area (Sedlak et al., 2017). However, differences in how microparticles and microplastics were extracted and counted in fish and sediment samples resulted in different particle size distributions and different colors being enumerated in fish and sediment samples. An initial analysis of the data found no relationship between microparticle concentrations in fish and sediment. Additional comparisons between data sets were hampered by the paucity of data for certain size fractions and color types and further analyses were not conducted.

Methods

Site selection

Eight sites were selected to monitor prey fish (Table 6.2 and Figure 6.1). Site selection was driven by four factors: 1) co-location with margin sediment samples to evaluate whether sediment may be a source of microparticles and microplastics to the food web and whether high levels in the sediment correlate with higher levels in prey fish; 2) locations near pathways (e.g., stormwater or wastewater outfalls) to evaluate their influence on uptake of microparticles and microplastics; 3) co-location with surface water sites where possible, to evaluate whether levels in surface water correlate with levels in prey fish; and 4) locations where fish would likely

be available, based upon the Moss Landing Marine Laboratories (MLML) staff's knowledge of the Bay.

Of the eight sites, three were in South and Lower South Bay (collectively referred to as South Bay herein) and three in Central Bay. Both Central Bay and South Bay are surrounded by urban landscapes, but they experience different flow dynamics. Central Bay waters have a shorter residence time, whereas South Bay waters experience more limited flushing and have a longer residence time.

The remaining two sites were located in Tomales Bay, a reference area located adjacent to the Point Reyes National Seashore. Tomales Bay was chosen as a reference area because it has little urban influence, and past examination of other urban contaminants there have shown lower concentrations in biota (e.g., Sedlak and Greig, 2012).



Figure 6.1. Sites where fish samples were collected for microparticle and microplastic analysis.

Table 6.2. Prey fish sample collection sites in San Francisco Bay and reference area.

Location	Site	Site Description	Co-located Sites			
			Sediment	Stormwater	Wastewater	Surface Water
Reference area	TB102	Tomales Bay South	TB102			
Reference area	TB101	Tomales Bay North - near Walker Creek	TB101			
Central Bay	CB10	Marina Bay	CB10	Meeker Slough		
Central Bay	CB37	Oyster Point		Colma Creek		
Central Bay	CB106	San Leandro Bay - NE near East Creek Slough	CB32	Line12F; Line12H		CB8
South Bay	SB074	Redwood Creek channel and launch ramp	SB074			
South Bay	LSB06	Near Hooks Point	LSB06 & LSB02		Palo Alto	Near LSB15
South Bay	SOSL40	Alviso Slough - near confluence with Coyote Creek	SOSL40	Coyote Creek; Guadalupe R. (upstream)	San Jose	Near LSB16

Sample collection

Moss Landing Marine Laboratories collected fish using otter trawls or cast nets at eight locations. Cast nets were primarily used for capturing topsmelt, and otter trawls were primarily used to capture anchovies. Ten fish per species were targeted at each of the eight sites (Figure 6.1). At two sites, the target was not obtained (LSB06 with four anchovies and CB10 with eight topsmelt), resulting in a total sample size of 152 fish. Total length was measured to group fish by size class; then each fish was individually wrapped in foil, placed on ice, and kept at -20°C. Fish were shipped to the University of Toronto on ice. Fish lengths ranged between 4.0–26.5 cm, and mass ranged between 0.4–125.3 g. Fish did not have juvenile markings and were captured in seawater.

Sample extraction and analysis

In the laboratory, the fish digestive tracts were dissected, and microparticles were extracted by digesting each whole gut at room temperature in a 4N KOH solution for 14 days (Dehaut et al., 2016; Rochman et al., 2015). After digestion, the sample was filtered through a 10 µm polycarbonate filter. All microparticles recovered were visually identified, measured, and classified by morphology and color.

In contrast to analyses of other matrices, clear and white fibers were not counted in fish samples because the majority of the analysis took place in a different laboratory where contamination with such fibers was observed. Subsequent investigation suggested the source of the contamination was likely to be paper cleaning wipes (e.g., Kimwipes). Measures were taken to minimize contamination, including wearing white cotton laboratory coats, wiping laboratory benches daily to reduce dust, cleaning all glassware with detergent and reverse osmosis (RO) water, and doing all work possible in a clean cabinet. However, the laboratory did not have a high-efficiency particulate air (HEPA) filter. The lack of a HEPA filter as well as the longer sample processing times (required in order to digest organic material) may explain the higher detections of microparticles in fish sample laboratory blanks compared to other matrices.

For each morphology and color (except clear and white fibers), the first three microparticles identified were analyzed by Raman or FTIR spectroscopy to determine the chemical composition of the particle using a reference spectra library. This resulted in approximately 66% of the microparticles undergoing spectroscopic analysis. If there were more than ten microparticles in a color/category within one fish, Raman spectroscopy was conducted on 30% of that color/category. For fibers, it was frequently difficult to discern the composition of the material due to spectral interferences from dyes. In these instances, the fibers were classified as anthropogenic unknown.

Laboratory blanks comprised of RO water were collected for every ten fish and analyzed in the exact same manner as fish samples; 15 laboratory blanks were analyzed.



Data analysis

Hypotheses for microparticle and microplastic contamination in San Francisco Bay fish were evaluated using multiple statistical methods.

1. Fish from reference sites will have lower concentrations of microparticles and microplastics.

This was evaluated by testing whether there were statistically significant differences among microparticle counts in fish from different regions. Because there were more than two groups being compared (i.e., Central Bay, South Bay, reference area—Tomales Bay, and laboratory blanks), a non-parametric Kruskal-Wallis test was used to evaluate whether groups were different. Once we determined the effect of site was significant, we used Dunn's pairwise post-hoc test to evaluate differences between all pairwise combinations of the sites and laboratory blanks (adjusted two-sided p -value using the Holm method).

2. An open Bay species (anchovy) will have a different microplastic composition than fish that reside largely in the Bay margins (topsmelt). Specifically, microparticle counts in anchovies will be different from topsmelt.

The distribution of microparticle counts for anchovies and topsmelt were found to be non-normal and heteroskedastic; therefore, we used a Mann-Whitney U Test to assess whether the distribution of microparticle counts was significantly different between topsmelt and anchovies.

Statistical evaluations were considered significant at $p < 0.05$.

Treatment of blanks

Laboratory and field blank results are reported alongside field sample results. Reported microparticle counts in field samples were not blank corrected (i.e., blank counts were not subtracted from the field sample counts) due to the non-uniform nature observed in the background field and laboratory contamination. The field and laboratory blanks were used to develop conservative thresholds for data qualification of the average laboratory blank plus two times the standard deviation, below which results are qualified. Qualified values should be treated with caution, because they may be strongly influenced by contamination from processing and analysis. The field and laboratory blank data as well as the threshold values are reported so individual readers can make their own assessment. Average microplastic counts in Bay prey fish are estimated using a lower and upper bound methodology for accounting for procedural contamination through blank correction.

Results

Quality assurance: Laboratory blanks

Fifteen laboratory blanks were analyzed, with microparticle counts (excluding clear and white fibers, as discussed in Methods) ranging between one and 17. The average number of microparticles/blank was 5.7, and the breakdown by morphology was 5.3 fibers, 0.1 fragments, 0.1 film, and 0.1 foam microparticles/blank (Figure 6.2). Out of 86 total microparticles found in the laboratory blanks, 80 were fibers (93%), and there were two microparticles each of fragments, foams, and films.

Based on the spectroscopy conducted on 83% of the microparticles in the blank, 10% were identified as plastic, 8% as natural or cellulosic material, and 76% as anthropogenic unknown (i.e., dyed material that may or may not be plastic). Microparticles that were confirmed to be plastic in laboratory blanks included two acrylic fibers, a single polyester fiber, a polyvinyl alcohol fiber, and a rubber fragment. Most of the fibers in laboratory blanks were of blue or black color (61% and 19%, respectively) and strong Raman signals from the dyes in these fibers resulted in most of these fibers being identified as anthropogenic unknown. Four microparticles (6%) could not be identified (i.e., could not be matched to spectra in the library).

The laboratory blanks indicated that sample contamination can occur during processing and analysis. The variability in blank contamination suggested blank correction via subtraction of the average blank value may not be appropriate. Microparticle counts were not blank corrected, but values were qualified when they were below a conservative threshold for data qualification of the average laboratory blank plus two times the standard deviation. A conservative threshold for data qualification was derived for each morphology, which is appropriate because the blanks have different levels of contamination from each morphology. The conservative thresholds are 13.6 fibers/sample, 0.8 fragments/sample, 0.8 films/sample, and 1.2 foams/sample. The conservative threshold for data qualification for total microparticles was 14.9 microparticles/sample based on the total particle counts in laboratory blanks. Qualified values should be treated with caution, because they may be strongly influenced by contamination from processing and analysis.

Due to the laboratory contamination with clear and white fibers, it should be noted that all of the results presented exclude this category of microparticles.

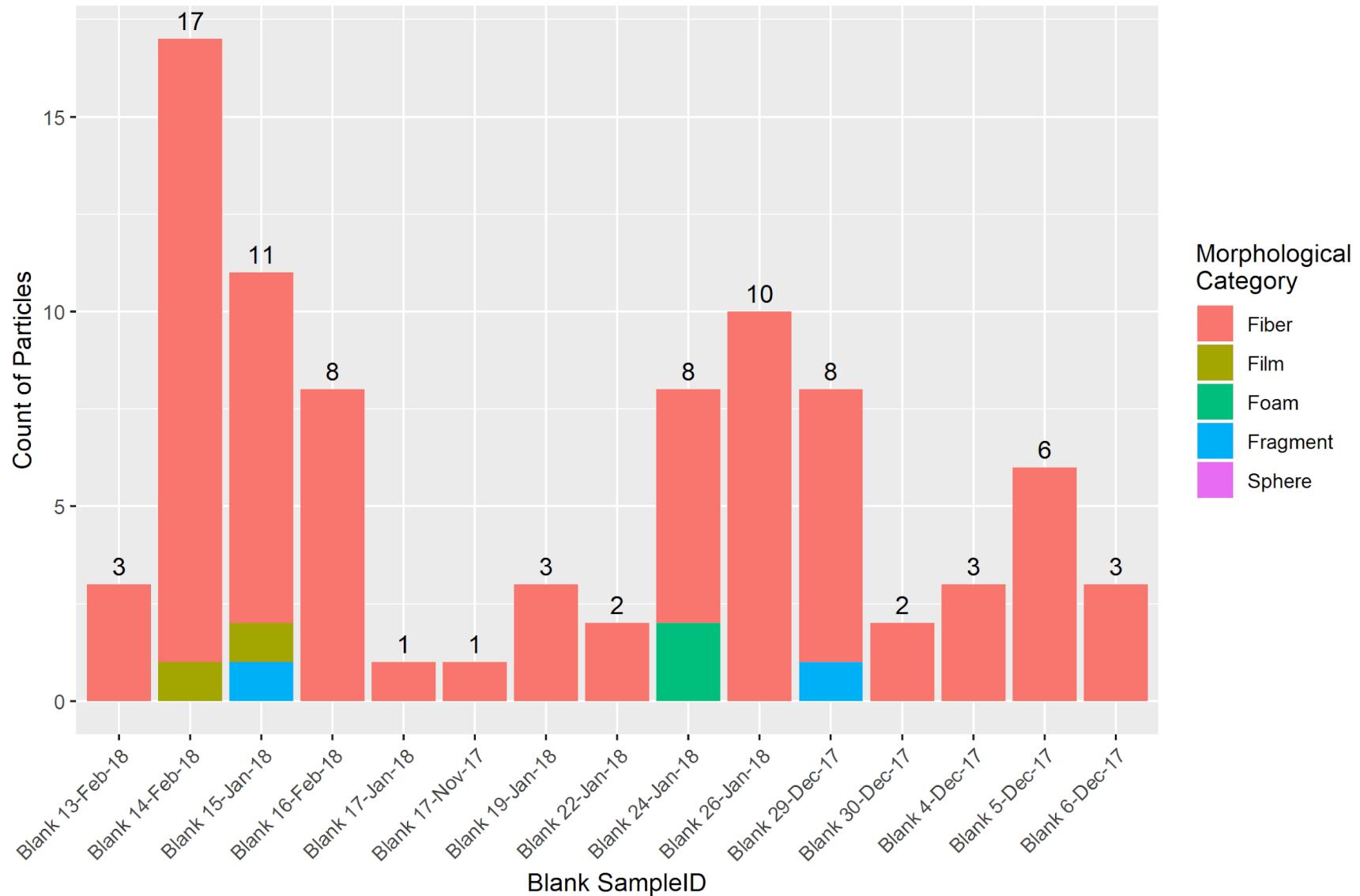


Figure 6.2. Microparticle counts in laboratory blanks. Numbers above bars represent total number of microparticles in each blank sample.

Particle occurrence and morphology

The contents of 152 fish guts were analyzed from all sites. All fish guts contained microparticles except for two fish from the Tomales Bay reference area, which did not contain any microparticles. A total of 1,919 microparticles were counted in the 152 fish (Table A-1). The number of microparticles in individual fish ranged from 0–57. The morphology of microparticles counted in all fish samples was 87% fibers, 10% fragments, and 3% films. Two foam particles (0.1% of total microparticles) and one sphere (0.05%) were identified in fish. The number of microfibers in individual fish ranged from 0–53, and 28% (43 out of 152) of fish guts had microfiber counts greater than 13.6 microfibers/fish, the conservative threshold for data qualification below which laboratory contamination may be a significant component of detected microfibers. Sixty-one fish (40%) contained 1–11 fragments, which are above the conservative threshold for data qualification of 0.8 fragments/fish. Twenty-five fish (16%) contained 1–12 films/fish, which are above the conservative threshold for data qualification of 0.8 films/fish. The non-fiber total microparticle counts (sum of fragments, films, foams, and spheres), ranged between 0–15 microparticles/fish, and 48% (73 out of 152) of fish guts contained at least one non-fiber microparticle/fish.

The most frequently detected microparticle sizes were less than 1 mm, and nearly all were smaller than 5 mm (Figure 6.3). Five hundred sixty-three microparticles in the fish were smaller than 0.35 mm, and 128 of those were smaller than 0.15 mm. Thirty-seven fish ingested 58 fibers that were greater than 5 mm in length, and the largest microparticle identified in the fish was a blue fiber 28 mm in length that was found in a topsmelt at site SOSL40 in the South Bay. For fragments, the most frequently detected microparticle size was 0.15–0.35 mm, followed by the 0.5–1 mm and 0–0.15 mm size classes. These microparticle counts represent all microparticles that visually looked like plastic, while only some of the microparticles were confirmed to be microplastics using Raman or FTIR spectroscopy, as described in the Composition section below.

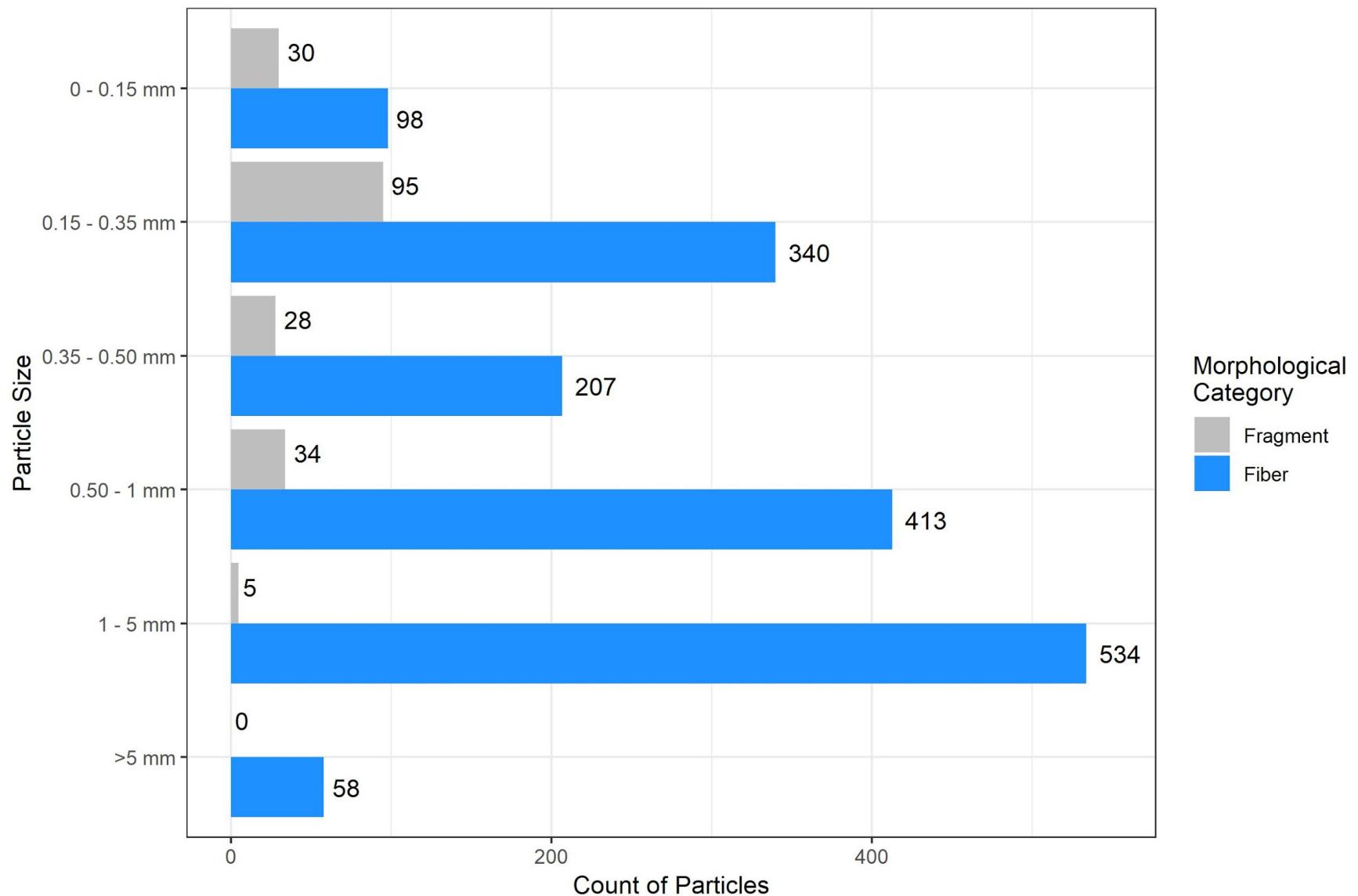


Figure 6.3. Microparticle sizes for fibers and fragments from fish samples. Particle size refers to the maximum dimension of the microparticle. Note: size bins are not evenly distributed.

Particle concentrations and fish size

It may be hypothesized that larger fish may have a higher burden of microparticles. However, particle counts in the individual fish did not correlate well with total fish length, even when analyzed by species and site (Figure 6.4).

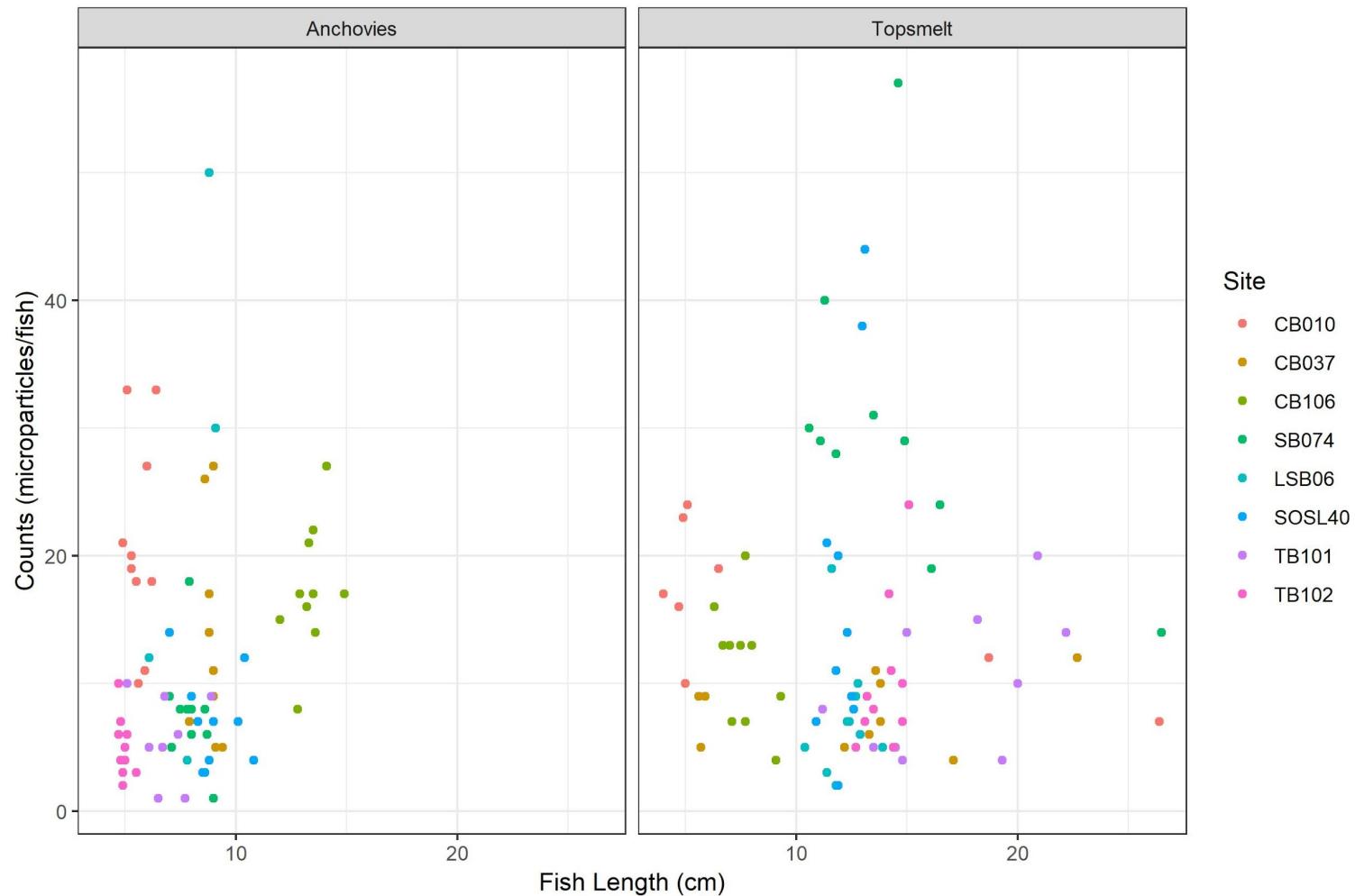


Figure 6.4. Fish length and microparticle counts for each species and sampling site.

Regional differences

Fish from San Francisco Bay had higher levels of total microparticles compared to the reference area, while fish from the reference area were not significantly different from blanks (Kruskal-Wallis test, $p = 3 \times 10^{-6}$; pairwise post-hoc test). Within the Bay, there was no regional difference in microparticle counts in fish from Central Bay compared to South Bay (Figure 6.5).

Particle counts in fish within the range measured in laboratory blanks should be treated with caution. While all prey fish from San Francisco Bay contained microparticles, 38% (43 out of 112 fish) contained total microparticle counts (all morphologies combined) above the threshold for data qualification of 14.9, below which laboratory contamination may be a significant component of detected particles. Of the fish samples from the reference area (Tomales Bay), only 10% (4 out of 40 fish) were above the conservative threshold for data qualification. The data qualification threshold for total particles may be considered particularly conservative because it does not account for the variation in blank contamination for different morphologies. When each morphology was considered separately, 67% of Bay fish contained microparticles (anchovies and topsmelt combined, all morphologies combined).

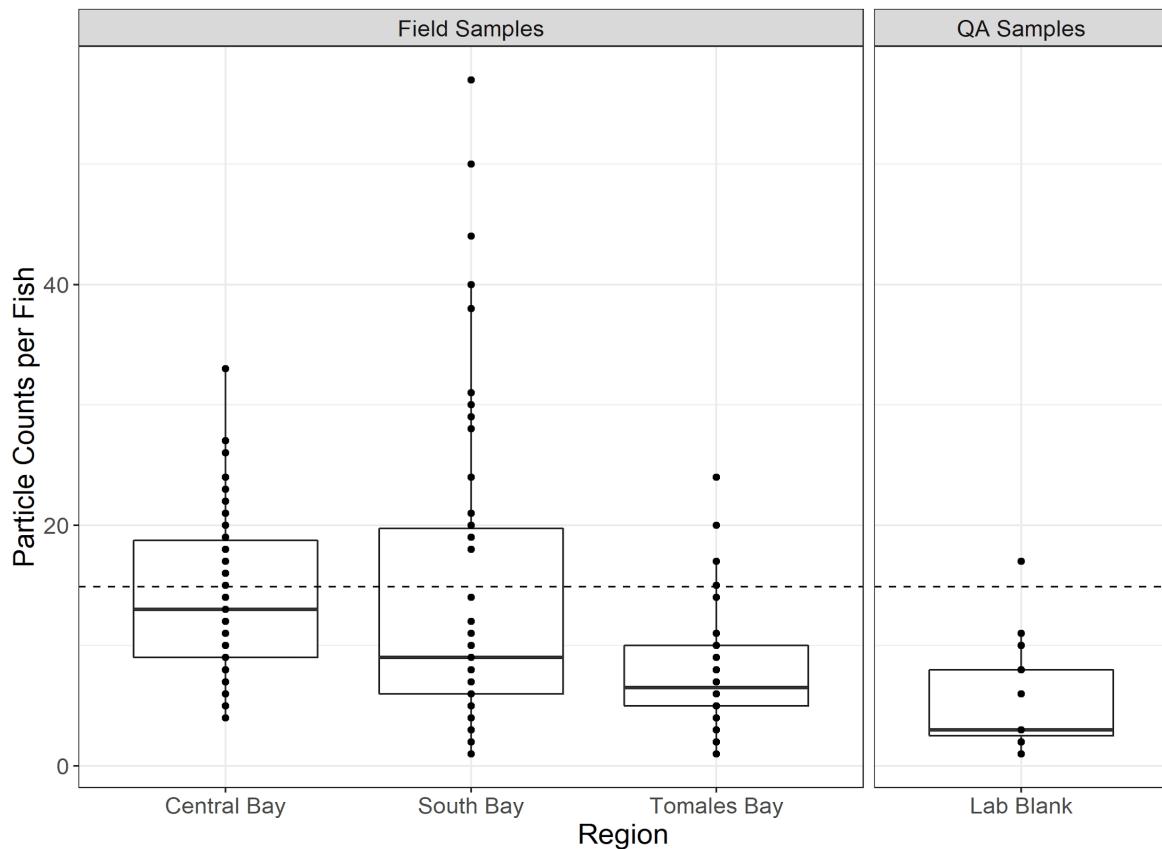


Figure 6.5. Number of microparticles/fish in each region. Points represent individual microparticles/fish; bold line = median, box = interquartile range (25th and 75th percentile), whiskers extend to minimum or maximum value, but no greater than 1.5 interquartile range (e.g., 25th percentile - 1.5 * interquartile range). The dotted line at 14.9 microparticles/sample represents the conservative data quality threshold below which laboratory contamination may be a significant component of total detected microparticles (all morphologies combined).

Species differences

There was no statistical difference in total microparticle counts between anchovies and topsmelt from San Francisco Bay (Mann-Whitney U Test, $p = 0.15$; Figure 6.6). The microparticle counts were dominated by microfiber counts. Most of the microfiber counts (74 out of 112 fish) were qualified because the fibers per fish were below the conservative threshold for data qualification of 13.6 fibers/sample.

Although non-fiber microparticle counts represented a small fraction of the total microparticle counts, topsmelt had higher levels of non-fiber microparticles compared to anchovies (Mann-Whitney U Test, $p = 0.001$). Sixty-nine percent of topsmelt (40 out of 58 fish) had one or more non-fiber microparticle, compared to 35% of anchovies (19 out of 54 fish), these non-fiber detection frequencies includes any fish with fragment, film, foam, or sphere microparticles counts above the conservative data quality threshold for each morphology.

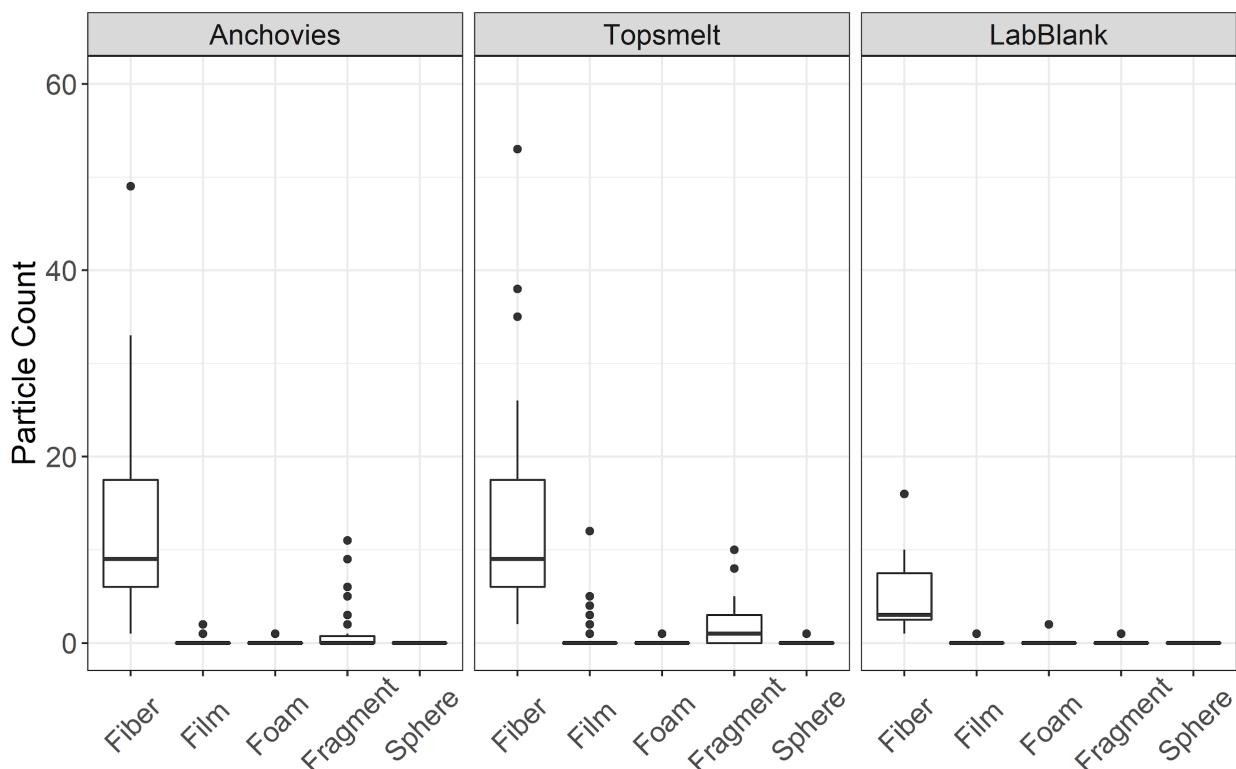


Figure 6.6. Number of microparticles/fish for each morphology for each species as compared with the laboratory blanks. Only data from the San Francisco Bay sites are represented (does not include Tomales Bay). Bold line = median; box = interquartile range (25th and 75th percentile); whiskers extend to minimum or maximum values, but no greater than 1.5 interquartile range (e.g., 25th percentile - 1.5 * interquartile range); points = individual sample points that lie outside the whisker range. Box and whiskers are not drawn for categories with insufficient counts (e.g., film, foam, sphere). The conservative thresholds for data qualification (average laboratory blank plus two times the standard deviation) for each morphology are 13.6 fibers/sample, 0.8 fragments/sample, 0.8 films/sample, and 1.2 foams/sample. Values below these thresholds should be treated with caution because they may be strongly influenced by contamination from processing and analysis.

Composition

Sixty-four percent of the 1,919 microparticles counted in all the fish samples were analyzed via spectroscopy (Figure 6.7; Table A-6.1). Fibers comprised 86% of the microparticles in San Francisco Bay fish samples (1,621 microparticles; Tables 6.A-1 and 6.A-5). A majority of the fibers that underwent spectroscopic analysis could not be identified and matched to a polymer type due to dye interference, and were therefore identified as anthropogenic unknown (Table A-6.5). The largest category of polymers for which the composition could be identified in San Francisco Bay fish was polyester fibers (77 fibers, 9%), followed by acrylic fibers (39 fibers, 5%), cellulose acetate fibers (18 fibers, 2%), and polypropylene fibers (8 fibers, 1%). Natural-based fibers comprised 11% of the fibers (98 fibers) identified in San Francisco Bay fish. This includes fibers that were specifically identified as cotton and wool, as well as anthropogenic cellulosic (dyed fibers made of cotton, rayon, or Lycocell), and cellulosic (similar fibers that were not dyed). In total, 23% of the microfibers analyzed were confirmed to be microplastic, while 60% were anthropogenic unknown and may or may not be microplastic.

Fragments comprised 11% of microparticles counted in San Francisco Bay fish samples (182 microparticles; Table A-6.5). Seventy percent of the fragments were analyzed by spectroscopy, but a majority of fragments analyzed could not be identified by polymer type, and were placed in the anthropogenic unknown or unknown categories (Table A-6.5). Small amounts of polyethylene (6 fragments, 5%) and polypropylene (5 fragments, 4%) were identified in fish samples. One rubber fragment (1%) and one polystyrene fragment (1%) were identified in fish samples. In total, 19% of the fragments analyzed were confirmed to be plastic.

Film composed 3% of the microparticles counted in Bay fish samples (46 microparticles; Table A-6.5). Eighty-nine percent of the film microparticles were analyzed by spectroscopy, and a majority (61%) of these could not be identified due to dye interference and were categorized as anthropogenic unknown. Twenty-four percent of film particles analyzed were plastic.

Two foam particles (one polypropylene and one unknown) were detected in all Bay fish samples. One acrylic sphere was detected in fish samples.

In total, 64% of the non-fiber microparticles (fragment, film, foam, sphere) were analyzed by spectroscopy, and 21% of microparticles analyzed were confirmed to be plastic (Table A-6.5).

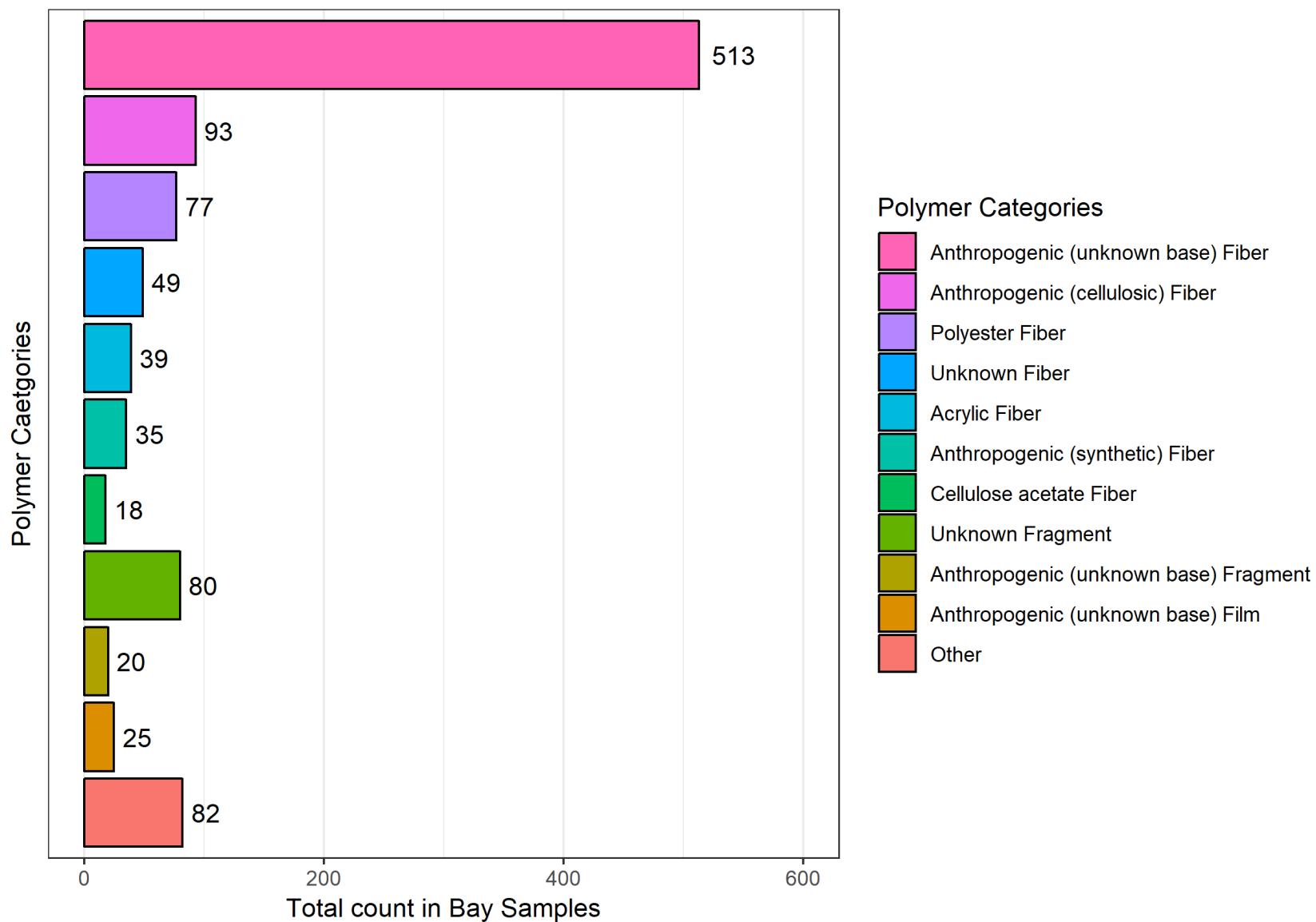


Figure 6.7. Top polymer categories identified by spectroscopy in San Francisco Bay anchovies and topsmelt. All other microparticle polymer-morphology categories not listed are grouped under “Other.”

Discussion

Microplastic levels were comparable to other studies

Microparticles were detected in all San Francisco Bay fish guts analyzed ($n = 112$); however, only 38% (43 out of 112 fish) were above the conservative threshold for data qualification (average of blanks plus two times the standard deviation, 14.9 microparticles/fish) for total microparticles. The distribution of microparticles/fish was skewed such that most fish had relatively low microparticles/fish, below the conservative threshold for data qualification, while a limited number of fish had high microparticle counts (Figure 6.5). Most of the data qualifications by morphology were due to the microfiber counts, because the conservative threshold for data qualification for fibers was relatively high (13.6 microfibers/fish). The variability in the number of fibers detected in the laboratory blanks (1–16, $n = 15$) makes the microfiber counts less definitive. Most of the laboratory blanks did not have non-fiber microparticles, which indicated the non-fiber microparticle counts were most likely less influenced by background contamination. We therefore estimated fibers and non-fiber microplastics separately, and estimated the portion that was microplastics based on the results of the spectroscopic analysis.

First, we estimated an upper and lower bound of the average number of microfibers in anchovies and topsmelt after blank correction. The upper-bound estimate was calculated after blank-correcting microfiber counts in both species of fish by the average of the laboratory blanks (5.3 microfibers/sample). The lower-bound estimate was calculated after blank-correcting microfiber counts using the conservative threshold for data qualification (13.6 microfibers/sample). Since not all microfibers are plastic, we then estimated a percentage of the microfibers in anchovies and topsmelt to be plastic based on the composition of fibers that were analyzed by spectroscopy (Tables A-6.3 and A-6.4). An upper-bound estimate of fiber microplastics/anchovy and fiber microplastics/topsmelt was calculated using the percentage of fibers confirmed to be plastic (21% for anchovies and 26% for topsmelt) and assuming 60% of the anthropogenic unknown fibers (62% and 57% of total fibers, respectively; Tables A-6.3 and A-6.4) was also plastic. The estimate of 60% of anthropogenic unknown fibers as plastic was based on the industry estimate that approximately 60% of textiles today are made from nylon and polyester (Almroth et al. 2018). The lower bound uses only the percentage of particles confirmed to be plastic.

We also estimated an upper and lower bound of the average number of non-fiber microparticles in anchovies and topsmelt. The upper-bound estimate was calculated by blank-subtracting particle counts of each morphology using the average of the blanks for each morphology, and summing the average of the resulting microparticle counts for each

morphology. The lower-bound estimate was calculated by blank-subtracting particle counts using the conservative threshold for data qualification for each morphology. The amount of microplastics was estimated from the microparticle averages based on the estimated percentage of plastics using a similar methodology as described above for fibers. The upper-bound estimate used the percentage of microparticles confirmed to be plastic and half of the percentage of particles identified as anthropogenic unknown for each morphology. The lower-bound estimate used only the percentage of particles confirmed to be plastic (Tables A-6.3 and A-6.4).

The resulting estimates of the average number of fiber microplastics in anchovies and topsmelt are 0.6–4.3 microplastics/anchovy and 0.9–4.5 microplastics/topsmelt (Table 6.3). The fiber microplastic estimates are blank-corrected as described above. The resulting estimates of the average number of non-fiber microplastics in anchovies is 0.2–0.4 microplastics/anchovy, and the estimate for average in topsmelt is 0.4–0.9 microplastics/topsmelt (Table 6.3).

The resulting estimates of the average number of fiber microplastics in anchovies and topsmelt were 0.6–4.3 microplastics/anchovy and 0.9–4.5 microplastics/topsmelt (Table 6.3). The resulting estimates of the average number of non-fiber microplastics in anchovies was 0.2–0.4 microplastics/anchovy, and the estimate for average in topsmelt was 0.4–0.9 microplastics/topsmelt (Table 6.3).

Table 6.3. Estimated upper- and lower-bound of average number of microplastics/fish for fibers and non-fiber particles based on results of spectroscopic analysis. Detection frequency of fiber and non-fiber microparticles/fish above the conservative threshold for data qualification. Non-fiber detection frequencies includes any fish with fragment, film, foam, or sphere microparticles counts above the conservative data quality threshold for each morphology.

	<i>Fibers</i>		<i>Non-fibers</i>	
	<i>Detection frequency (Microparticles)</i>	<i>Estimated average (microplastics/fish)</i>	<i>Detection frequency (Microparticles)</i>	<i>Estimated average (microplastics/fish)</i>
Anchovies	33% (18/54)	0.6–4.3	35% (19/54)	0.2–0.4
Topsmelt	34% (20/58)	0.9–4.5	69% (40/58)	0.4–0.9

Microparticle counts in fish from Tomales Bay were not statistically different from laboratory blanks, so estimates of microplastics/fish were not calculated.

Numerous studies have quantified the presence of microplastics in marine fish (Abbasi et al., 2018; Anastasopoulou et al., 2018; Avio et al., 2015; Baalkhuyur et al., 2018; Bellas et al., 2016; Boerger et al., 2010; Chagnon et al., 2018; Collard et al., 2017; Davison and Asch, 2011; Foekema et al., 2013; Güven et al., 2017; Hermsen et al., 2017; Hipfner et al., 2018; Jabeen et al., 2017; Lusher et al., 2017, 2013; Nelms et al., 2018; Ory et al., 2018, 2017; Peters et al., 2017; Roch and Brinker, 2017; Rochman et al., 2015; Romeo et al., 2015; Rummel et al., 2016; Vendel et al., 2017; Wagner et al., 2017). However, comparison



between studies is complicated by differences in methods, reporting, and fish species. There are currently no standardized methods for analyzing microplastics in fish, meaning each study is unique in target tissue (e.g., individual tissues vs. whole fish), sample pre-treatment methods (e.g., storage and handling, dissection, digestion), microparticle size ranges analyzed, identification methods (e.g., visual, spectroscopy), and blank collection and interpretation. Even without these differences, differences in fish species among studies may also affect microplastic counts, as each fish species has a different habitat and diet, potentially affecting microplastic exposure. Even within samples of a single species, fish age, sex, and size; sampling season; and location may also play a role in microplastic accumulation.

We compared microplastics in Bay fish with other studies with these methodological and species differences in mind. Table 6.4 includes only marine and estuarine species that do not eat other fish, and therefore have somewhat comparable diets. Microplastics have been shown to transfer up food chains (Mattsson et al., 2017; Nelms et al., 2018; Setälä et al., 2014; Tosetto et al., 2017), and we therefore expect piscivorous fish to have differing microplastic loads than prey fish. Foraging preferences have been shown to correlate with microplastic amount and morphology in fish digestive tracts, even between fish at similar trophic levels (Jabeen et al., 2017; Peters et al., 2017; Zheng et al., 2019). However, eliminating all fish with different diets from our study would leave no fish to compare. Furthermore, Table 6.4 only includes studies that digested the gut; many studies of microplastics in fish only rinse or scrape the gut lumen, which may underestimate totals if microparticles are entrenched in the tissue.

Table 6.4. Average microplastic counts and microparticle detection frequencies above the conservative data quality threshold from this study, and comparable studies around the world. Most other studies are unclear whether reported abundances and detection frequencies are microparticles or microplastics.

Ref.	Extraction method	Filter size (μm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish	Particle frequency/ description total fish
Present study	KOH digestion and filtration	10	Raman/ FTIR	San Francisco Bay	Topsmelt silverside <i>Atherinops affinis</i>	Marine; brackish; pelagic-neritic	plants, invertebrates, diatoms, and amphipods	0.9–4.5 fibers; 0.9 non-fibers	18/54; 20/54	mostly fibers; clear and white fibers were not counted
					Northern or Californian anchovy <i>Engraulis mordax</i>	Marine; pelagic-neritic	plankton, zooplankton, crustaceans, fish eggs, larvae	0.6–4.3 fibers; 0.4 non-fibers	19/58; 40/58	mostly fibers; clear and white fibers were not counted
Anastasopoulou et al. 2018	“macro”-litter items (> 1 mm) visually identified and removed, H_2O_2 digestion and filtration	visual only	Croatian Sea, Northern/ Middle Adriatic Sea	red mullet <i>Mullus barbatus</i>	Marine; demersal	polychaete worms, bivalve mollusks, and crustaceans	0.5 ± 0.8	8/25	mostly fragments	
				European pilchard <i>Sardina pilchardus</i>	Marine; freshwater; brackish; pelagic-neritic; oceanodromous	zooplankton and phytoplankton	0.9 ± 1.4	11/28	mostly fragments	
			Greece, NE Ionian Sea	European pilchard <i>Sardina pilchardus</i>	Marine; freshwater; brackish; pelagic-neritic; oceanodromous	zooplankton and phytoplankton	0.8 ± 1.2	17/30	mostly fragments	
			Slovenian Sea, Northern Adriatic Sea	common, Dover, or black sole <i>Solea solea</i>	Marine; brackish; demersal; oceanodromous	worms, mollusks, and small crustaceans	1.9 ± 2.7	13/20	mostly fibers	
				gilthead seabream <i>Sparus aurata</i>	Marine; brackish; demersal	shellfish, including mussels and oysters	7.3 ± 6.6	20/20	mostly fibers	

Ref.	Extraction method	Filter size (µm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish
Anastasopoulou et al. 2018	"macro"-litter items (> 1 mm) visually identified and removed, H ₂ O ₂ digestion and filtration		visual only	Slovenian Sea, Northern Adriatic Sea	golden grey mullet <i>Chelon auratus</i>	Marine; freshwater; brackish; pelagic-neritic; catadromous	small benthic organisms, detritus, and occasionally insects and plankton	9.5 ± 8.4	19/20 mostly fibers
Avio et al. 2015	filtration and H ₂ O ₂ digestion after density separation	8	FTIR	Central and North Adriatic Sea	European pilchard <i>Sardina pilchardus</i>	Marine; freshwater; brackish; pelagic-neritic; oceanodromous	zooplankton and phytoplankton	positive sample mean = 1.78 ± 0.7	19/99
					red mullet <i>Mullus barbatus</i>	Marine; demersal	small benthic crustaceans, worms and mollusks	positive sample mean = 1.57 ± 0.78	7/11
Baalkhuyur et al. 2018	NaOH digestion and filtration	200	FTIR	Saudi Arabian Red Sea	rosy dwarf monocle bream <i>Parascloopsis eriomma</i>	Marine; demersal	benthic invertebrates	not reported	3/5 avg size = 1.38 mm
					regal angelfish <i>Pygoplites diacanthus</i>	Marine; reef-associated	sponges and tunicates		0/5
					scissortail sergeant <i>Abudefduf sexfasciatus</i>	Marine; reef-associated	zooplankton and algae	0-1	1/5 1.2 mm

Ref.	Extraction method	Filter size (µm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish
Baalkhuyur et al. 2018	NaOH digestion and filtration	200	FTIR	Saudi Arabian Red Sea	Sohal surgeonfish <i>Acanthurus sohal</i>	Marine; reef-associated	algae	0/3	
					threespot dascyllus <i>Dascyllus trimaculatus</i>	Marine; reef-associated	algae, copepods, and other planktonic crustaceans	0/2	
					bluespine unicornfish <i>Naso unicornis</i>	Marine; reef-associated	algae	0/2	
					skinnycheek lanternfish <i>Benthosema pterotum</i>	Marine; benthopelagic	copepods and various crustacean larvae	1/10	2.58 mm
					Red mullet <i>Mullus barbatus</i>	Marine; demersal	small benthic crustaceans, worms and mollusks	24/128	mostly fibers and spheres
Bellas et al. 2016	NaOH digestion and filtration	visual only		Mediterranean Sea	Atlantic herring <i>Clupea harengus</i>	Marine; brackish; benthopelagic; oceanodromous	copepods, arrow worms, pelagic amphipods, mysids and krill	0/100	
					sprat <i>Sprattus sprattus</i>	Marine; brackish; pelagic-neritic; oceanodromous	planktonic crustaceans	0-2	1/100 Polymethyl-methacrylate spheres
Hermesen et al. 2017	KOH digestion and filtration	20	FTIR	North Sea					

Ref.	Extraction method	Filter size (µm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish
Jabeen et al. 2017 5	H ₂ O ₂ digestion, density separation and filtration	FTIR	fishery markets of Shanghai	Asian pencil halfbeak <i>Hyporhamphus intermedius</i>	Marine; freshwater; brackish; pelagic-neritic; amphidromous	zooplankton	3.7 ± 2.2	fibers and fragments	
				Japanese grenadier anchovy <i>Coilia ectenes</i>	Marine; freshwater; brackish; pelagic-neritic; anadromous	plankton	4.0 ± 1.8	fibers and fragments	
				silver sillago <i>Sillago sihama</i>	Marine; brackish; reef-associated; amphidromous	polychaete worms, shrimps and amphipods	2.8 ± 1.5	fibers and fragments	
				silver pomfret <i>Pampus cinereus</i>	Marine; benthopelagic; oceanodromous	zooplankton	3.0 ± 0.8	fibers and fragments	
				flathead grey mullet <i>Mugil cephalus</i>	Marine; freshwater; brackish; benthopelagic; catadromous	detritus, micro-algae and benthic organisms	3.7 ± 1.0	fibers, fragments, pellets	
				three-lined tongue sole <i>Cynoglossus abbreviatus</i>	Marine; demersal	mainly benthic invertebrates	6.9 ± 2.4	fibers, fragments, sheets, films	
				common carp <i>Cyprinus carpio</i>	Freshwater; brackish; benthopelagic	a variety of benthic organisms and plant material	2.5 ± 1.3	fibers	
				goldfish <i>Carassius auratus</i>	Freshwater; brackish; benthopelagic	plankton, benthic invertebrates, plant material and detritus	1.9 ± 1.0	fibers and pellets	

Ref.	Extraction method	Filter size (μm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish
Jabeen et al. 2017	H ₂ O ₂ digestion, density separation and filtration	5	FTIR	fishery markets of Shanghai	silver carp <i>Hypophthalmichthys molitrix</i>	Freshwater; brackish; benthopelagic; potamodromous	phytoplankton and zooplankton	3.8 ± 2.0	fibers, fragments, pellets
					Nile tilapia <i>Oreochromis niloticus</i>	Freshwater; brackish; benthopelagic; potamodromous	phytoplankton, benthic algae, insect larvae, aufwuchs and detritus	0/5	
Rochman et al. 2015	KOH digestion and visual separation	500	visual only	fish market in Makassar, Sulawesi, Indonesia	shortfin scad <i>Decapterus macrosoma</i>	Marine; reef-associated	zooplankton and small invertebrates	0–21 (average 2.5)	5/17 styrofoam, fragments
					streamlined spinefoot or rabbitfish <i>Siganus argenteus</i>	Marine; reef-associated	algae	0–1	1/2 fragments
					mottled spinefoot or rabbitfish <i>Siganus fuscescens</i>	Marine; brackish; reef-associated; oceanodromous	algae and seagrasses	0/2	
					white-spotted spinefoot or rabbitfish <i>Siganus canaliculatus</i>	Marine; brackish; reef-associated; oceanodromous	algae and seagrasses	0–1	1/3 monofilament
					oxeye scad <i>Selar boops</i>	Marine; reef-associated	planktonic and benthic invertebrates	0/7	

Ref.	Extraction method	Filter size (µm)	Plastic ID method	Study Location	Fish Species	Environmental Domain	Diet	Abundance (# particles/fish)	Particle detection frequency/ description total fish
Rochman et al. 2015	KOH digestion and visual separation	500	visual only	local fishermen, Half Moon Bay, California	Jack silverside or jacksmelt <i>Atherinopsis californiensis</i> Pacific or Californian anchovy <i>Engraulis mordax</i>	Marine; pelagic-neritic	phytoplankton euphausiids, copepods and decapod larvae	0–10 (average 1.6) 0–1 (average 0.3)	2/7 3/10

Chapter 6—Prey Fish

The detection frequency of microfibers and non-fiber microparticles in San Francisco Bay anchovies and topsmelt, taking into account only microparticle counts above the conservative data qualification threshold, appeared to be within the range reported in other studies. Studies that report high detection frequencies observed mostly fibers (Anastasopoulou et al., 2018; Jabbeen et al., 2017), and it is unclear whether blank contamination was measured or subtracted from reported counts. Jabeen et al. (2017) observed microparticles in the digestive tract of 100% of marine fish and 95.7% of freshwater fish they sampled. They also observed a high abundance of larger particles; 70.9% of marine fish and 43.5% of freshwater fish contained items that appeared to be plastic with dimensions between 5 and 25 mm. These findings may be due to high plastic loads in marine waters near the major city of Shanghai (Isobe et al., 2015) where these fish were caught. Although Jabeen et al. (2017) reported low blank contamination (0.25 ± 0.05 items/filter) and FTIR polymer identification of a subset of microparticles, it is unclear from their reporting whether blank contamination was subtracted from reported microplastics in fish and whether numbers account for the percentage of microparticles confirmed to be plastic. The detection frequencies found in our study are conservative because we only included values above a conservative threshold for data qualification.

Most other studies also reported an average of less than one microplastic/fish when fibers were not included, which is consistent with non-fiber microplastics estimates in San Francisco Bay. An

exception to this was Avio et al. (2015), who reported an average of greater than one microplastic/fish (1.6–1.8 microplastics/fish) in the Adriatic Sea, but the average was calculated only for fish with detectable particles (excluding non-detects). Upper-bound estimates of average non-fiber microplastics/fish in San Francisco Bay were similar when only such positive values were counted after blank correction. Also, it is not clear in other studies whether reported microparticle abundances account for particles that were confirmed to be plastic.

The abundance of fibers seen in San Francisco Bay was also within the range reported in other studies.

Anastasopoulou et al. (2018) reported higher average fiber abundances, including the abundances reported in gilthead seabream and golden grey mullet from the Slovenian Sea. However, it is unclear whether blank contamination was subtracted from reported counts. Fibers from the present study were likely undercounted



since white and clear fibers were not included. Recent reports of microplastics in fish from the Pearl River catchment in China indicated that transparent polyester fibers were the predominant type of microplastic debris in fish guts (Zheng et al., 2019). It is not possible to determine whether the fish from San Francisco Bay would have similar compositions of clear plastic fibers, had blank contamination been less significant.

Several of the comparative studies (Table 6.4) lacked polymer identification, which may lead to erroneous characterization of microplastics; it is also not clear in other studies that used spectroscopy for polymer identification how plastic confirmations were used to estimate microparticle and microplastic abundances and detection frequencies. Therefore, our method of estimating microplastics based on the percentage of microparticles confirmed to be plastic may be more conservative than other studies. Another consideration is that we used a smaller filter size (10 µm) compared with many other studies; we were able to count smaller microparticles, likely increasing total counts somewhat. Two of the studies with the lowest microparticle counts in fish used filters with pore sizes of 200 µm or greater (Baalkhuyur et al., 2018; Rochman et al., 2015). However, our quantification of smaller particles may not have had a large effect on counts, as only 7% of the particles in the present study (16% of fragments and 6% of fibers) were in the smallest (<150 µm) size class (Figure 6.3).

San Francisco Bay fish appeared to ingest mostly fibers and fragments, which is consistent with the microparticles found in fish digestive tracts in other studies around the world. This is likely due to the relatively high proportion of these morphologies compared to spheres, films, and foams in marine and estuarine water and sediment, including in the Bay.

Comparison between Bay species indicate differences

The two prey fish species chosen for this study were selected because of the differences in their foraging characteristics, which may lead to differences in microparticle ingestion. Topsmelt are thought to reside closer to Bay margins, shallow areas, and benthic areas, as compared to anchovies that generally live and feed in deeper Bay channels and throughout the water column.

While there was no significant difference in total microparticle counts between the two species, this was mainly driven by the variability in the fiber counts. There was a statistical difference in non-fiber microparticle counts between the two species; topsmelt had higher levels of non-fiber microparticles. The median number of microparticles in anchovies was zero compared to 1.5 microparticles/fish in topsmelt. Most of the microparticles could not be identified by polymer type due to interference of embedded dyes with the Raman spectroscopy. Topsmelt may be exposed to higher levels of microparticles and microplastics in sediment and margin areas, their preferred



habitat. However, due to methodological differences in how sediment and fish samples were analyzed, we could not evaluate whether topsmelt concentrations were correlated with sediment concentrations, and whether higher concentrations in topsmelt compared to anchovies may be explained by topsmelt preferences for areas with higher levels of microplastics.

Peters et al. (2017) reported that a benthic-foraging grunt species had lower microplastic ingestion rates than the other species measured. The authors suggested that the grunt was a more selective benthic invertebrate feeder and therefore less likely to ingest microplastics compared to a generalist. Lusher et al. (2013) did not find a significant difference in microplastic levels in sampled pelagic and demersal species. Likewise, in an extensive survey of 2,233 fish from 69 species in two Brazilian estuaries, Vendel et al. (2017) found that microplastic ingestion occurred irrespective of fish size, functional group, and feeding guild.

Particles detected in fish may potentially translocate and cause adverse impacts

There are mixed reports of microplastic effects on fish (Foley et al., 2018). Many studies report minimal effects, even when fish were exposed to relatively high doses of microplastics (Ašmonaitė et al., 2018a, 2018b; Caruso et al., 2018; Foley et al., 2018; Jacob et al., 2019; Jovanović et al., 2018; LeMoine et al., 2018; Malinich et al., 2018; Messinetti et al., 2019; Tosetto et al., 2017). However, other studies report a variety of adverse effects, including inflammation and oxidative stress (Brandts et al., 2018; Choi et al., 2018; Ding et al., 2018; Jin et al., 2018; Romano et al., 2018), microbiome changes (Jin et al., 2018), altered swimming and feeding behavior (Barboza et al., 2018; Critchell and Hoogenboom, 2018; Mattsson et al., 2017; Yin et al., 2018), altered reproductive success (Peixoto et al., 2019; Pitt et al., 2018), and decreased growth and body condition (Barboza et al., 2018; Critchell and Hoogenboom, 2018; Jabeen et al., 2018). The

discrepancy may be due to differences in species and microplastics (shape; size; polymer; virgin or weathered; associated contaminants—including additives, sorbed chemicals, and microorganisms), and whether the microplastic exposure occurred alone or in conjunction with exposure to other contaminants. Irregularly shaped microplastics (fragments and fibers), especially those that have experienced environmental weathering, are more likely to cause adverse effects than virgin particles (Choi et al., 2018; Jabeen et al., 2018). Microplastics also seem to increase the accumulation and toxicity of other environmental contaminants, including heavy metals (Barboza et al., 2018; Rainieri et al., 2018; Wen et al., 2018), persistent organic pollutants (Pannetier et al., 2019a, 2019b; Rainieri et al., 2018), and emerging contaminants (Chen et al., 2017; Zhang et al., 2019). Microplastics can also act as a vector for pathogens (Viršek et al., 2017). Smaller microparticles tend to cause more adverse effects (Critchell and Hoogenboom, 2018; Ding et al., 2018; Mattsson et al., 2017), possibly because they can more easily translocate from the gut into the bloodstream and other tissues (Ding et al., 2018; Mattsson et al., 2017; Messinetti et al., 2019), including maternal transfer to embryos (Pitt et al., 2018).

While most microplastics (greater than 90%) ingested by fish will be excreted (Lusher et al., 2017), they may bioaccumulate if they translocate across the gut and into the tissues of the animal (Browne et al., 2008). Reports vary as to what size of microplastics may translocate across the gut epithelium. Lusher et al. (2017) suggest that only microparticles smaller than 150 µm may translocate, and only microplastics smaller than 20 µm may penetrate organs. In another study on polyethylene microplastics, researchers found that 80% of exposed anchovies had microplastics in their livers, with sizes ranging from 125 to 438 µm (Collard et al., 2017). Additionally, Avio et al. (2015) found that microplastics between 200 and 600 µm translocated into the liver of mullets.

Based on the hypothesis that microparticles smaller than 150 µm may translocate, in this study, a total of 30 fragments (16% of total fragments) and 98 fibers (6% of total fibers) could potentially translocate (Figure 6.3). A more protective approach would use the higher estimate that microparticles up to 600 µm have been found to translocate. A total of 39% of fragments and 80% of fibers from this study were smaller than 500 µm. The smallest microparticle observed was a fragment 24 µm in width. These smaller microparticles were not targeted in Bay surface water samples (355 µm net size), sediment samples (125 µm sieve), or in wastewater and stormwater samples (125 µm sieve). Identifying very small microparticles in future studies will be important to understanding the potential impacts of microparticles on Bay fish.

Conclusions

This study sought to quantify the abundance of microplastics in prey fish, characterize the type and chemical composition of microparticles found in prey fish, identify sites or regions of particular concern, and compare the results with other studies.

While all prey fish from San Francisco Bay contained microparticles, only 38% had total microparticle counts greater than 14.9 microparticles/fish, a conservative data quality threshold below which laboratory contamination may be a significant component of detected microparticles. While there was no statistically significant difference in the abundance of fibers detected in Bay fish by species, non-fiber microparticles were more frequently detected, and at higher levels, in topsmelt compared to anchovies. Prey fish from San Francisco Bay had statistically higher total microparticle counts compared to the reference area with minimal urban influence (Tomales Bay).

The composition of microparticles counted in San Francisco Bay fish samples was 86% fibers, 11% fragments, and 3% films. While most microparticles could not be positively identified based on polymer type (unknown category), 23% of fibers that underwent spectroscopy were positively identified as plastic and 11% were natural-based. For non-fiber microparticles that underwent spectroscopy, 20% were confirmed to be plastic and 4% were natural-based.

The estimated averages of fiber and non-fiber microplastics per fish were calculated separately because the variability of fibers in the blank samples, as well as the exclusion of white and clear fibers from counts, makes the microfiber counts less definitive than the non-fiber microplastic counts. The estimated average number of non-fiber microplastic counts in the Bay fish was 0.2–0.9 microplastics/fish, and the estimated average of fiber microplastic counts was 0.6–4.5 microplastics/fish. Based on the incidence of microplastics in the prey fish studied, it is likely that higher trophic organisms such as sport fish and humans are exposed to microplastics through the food web.

The estimated range of average microplastics in Bay prey fish guts are within the ranges observed in most other studies. Some methodological differences made comparisons difficult; the present study used polymer identification to estimate the proportion of microparticles that were microplastics, while several other studies did not include polymer identification or did not clearly explain how polymer identification was used to estimate microplastic abundances. Additionally, the present study used a smaller filter size than most other studies, which may increase counts, while in contrast, the exclusion of white and clear fibers likely resulted in decreased counts.

Chapter 6—Prey Fish

The most frequently detected particle sizes were less than 1 mm and most were smaller than 2 mm. Fibers reached greater lengths than fragments, with the largest fiber identified in the fish being 28 mm in length. The smallest microparticles have the potential to translocate within the fish and cause adverse effects. Reported adverse impacts from microplastics include inflammation and oxidative stress, microbiome changes, altered swimming and feeding behavior, altered reproductive success, and decreased growth and body condition. Currently, there is a dearth of studies identifying impacts of microplastic exposure to organisms at ecologically relevant concentrations; therefore, we currently do not have ecotoxicological thresholds to evaluate whether fish in the Bay may be impacted by microplastic concentrations.

References

- Abbasi, S., Soltani, N., Keshavarzi, B., Moore, F., Turner, A., Hassanaghaei, M., 2018. Microplastics in different tissues of fish and prawn from the Musa Estuary, Persian Gulf. Chemosphere 205, 80–87. <https://doi.org/10.1016/j.chemosphere.2018.04.076>
- Anastasopoulou, A., Kovač Viršek, M., Bojanic Varezić, D., Digka, N., Fortibuoni, T., Koren, Š., Mandić, M., Mytilineou, C., Pešić, A., Ronchi, F., Šiljić, J., Torre, M., Tsangaridis, C., Tutman, P., 2018. Assessment on marine litter ingested by fish in the Adriatic and NE Ionian Sea macro-region (Mediterranean). Marine Pollution Bulletin 133, 841–851. <https://doi.org/10.1016/j.marpolbul.2018.06.050>
- Ašmonaitė, G., Larsson, K., Undeland, I., Sturve, J., Carney Almroth, B., 2018a. Size Matters: Ingestion of Relatively Large Microplastics contaminated with environmental pollutants posed little risk for fish health and fillet quality. Environ. Sci. Technol. 52, 14381–14391. <https://doi.org/10.1021/acs.est.8b04849>
- Ašmonaitė, G., Sundh, H., Asker, N., Carney Almroth, B., 2018b. Rainbow trout maintain intestinal transport and barrier functions following exposure to polystyrene microplastics. Environ. Sci. Technol. 52, 14392–14401. <https://doi.org/10.1021/acs.est.8b04848>
- Avio, C.G., Gorbi, S., Regoli, F., 2015. Experimental development of a new protocol for extraction and characterization of microplastics in fish tissues: First observations in commercial species from Adriatic Sea. Marine Environmental Research, Particles in the Oceans: Implication for a safe marine environment 111, 18–26. <https://doi.org/10.1016/j.marenvres.2015.06.014>
- Baalkhuyur, F.M., Bin Dohaish, E.-J.A., Elhalwagy, M.E.A., Alikunhi, N.M., AlSuwailem, A.M., Røstad, A., Coker, D.J., Berumen, M.L., Duarte, C.M., 2018. Microplastic in the gastrointestinal tract of fishes along the Saudi Arabian Red Sea coast. Marine Pollution Bulletin 131, 407–415. <https://doi.org/10.1016/j.marpolbul.2018.04.040>
- Barboza, L.G.A., Vieira, L.R., Branco, V., Figueiredo, N., Carvalho, F., Carvalho, C., Guilhermino, L., 2018. Microplastics cause neurotoxicity, oxidative damage and energy-related changes and interact with the bioaccumulation of mercury in the European seabass, *Dicentrarchus labrax* (Linnaeus, 1758). Aquatic Toxicology 195, 49–57. <https://doi.org/10.1016/j.aquatox.2017.12.008>
- Bellas, J., Martínez-Armental, J., Martínez-Cámarra, A., Besada, V., Martínez-Gómez, C., 2016. Ingestion of microplastics by demersal fish from the Spanish Atlantic and Mediterranean coasts. Marine Pollution Bulletin 109, 55–60. <https://doi.org/10.1016/j.marpolbul.2016.06.026>

Chapter 6—Prey Fish

Boerger, C.M., Lattin, G.L., Moore, S.L., Moore, C.J., 2010. Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. *Marine Pollution Bulletin* 60, 2275–2278.
<https://doi.org/10.1016/j.marpolbul.2010.08.007>

Brandts, I., Teles, M., Tvarijonaviciute, A., Pereira, M.L., Martins, M.A., Tort, L., Oliveira, M., 2018. Effects of polymethylmethacrylate nanoplastics on *Dicentrarchus labrax*. *Genomics* 110, 435–441. <https://doi.org/10.1016/j.ygeno.2018.10.006>

Caruso, G., Pedà, C., Cappello, S., Leonardi, M., La Ferla, R., Lo Giudice, A., Maricchiolo, G., Rizzo, C., Maimone, G., Rappazzo, A.C., Genovese, L., Romeo, T., 2018. Effects of microplastics on trophic parameters, abundance and metabolic activities of seawater and fish gut bacteria in mesocosm conditions. *Environ Sci Pollut Res* 25, 30067–30083. <https://doi.org/10.1007/s11356-018-2926-x>

Chagnon, C., Thiel, M., Antunes, J., Ferreira, J.L., Sobral, P., Ory, N.C., 2018. Plastic ingestion and trophic transfer between Easter Island flying fish (*Cheilopogon rapanouiensis*) and yellowfin tuna (*Thunnus albacares*) from Rapa Nui (Easter Island). *Environmental Pollution* 243, 127–133.
<https://doi.org/10.1016/j.envpol.2018.08.042>

Chen, Q., Yin, D., Jia, Y., Schiwy, S., Legradi, J., Yang, S., Hollert, H., 2017. Enhanced uptake of BPA in the presence of nanoplastics can lead to neurotoxic effects in adult zebrafish. *Science of The Total Environment* 609, 1312–1321. <https://doi.org/10.1016/j.scitotenv.2017.07.144>

Choi, J.S., Jung, Y.-J., Hong, N.-H., Hong, S.H., Park, J.-W., 2018. Toxicological effects of irregularly shaped and spherical microplastics in a marine teleost, the sheepshead minnow (*Cyprinodon variegatus*). *Marine Pollution Bulletin* 129, 231–240.
<https://doi.org/10.1016/j.marpolbul.2018.02.039>

Collard, F., Gilbert, B., Compère, P., Eppe, G., Das, K., Jauniaux, T., Parmentier, E., 2017. Microplastics in livers of European anchovies (*Engraulis encrasicolus*, L.). *Environmental Pollution* 229, 1000–1005. <https://doi.org/10.1016/j.envpol.2017.07.089>

Critchell, K., Hoogenboom, M.O., 2018. Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (*Acanthochromis polyacanthus*). *PLoS ONE* 13, e0193308.
<https://doi.org/10.1371/journal.pone.0193308>

Davison, P., Asch, R., 2011. Plastic ingestion by mesopelagic fishes in the North Pacific Subtropical Gyre. *Mar. Ecol. Prog. Ser.* 432, 173–180. <https://doi.org/10.3354/meps09142>

Dehaut, A., Cassone, A.-L., Frère, L., Hermabessiere, L., Himber, C., Rinnert, E., Rivière, G., Lambert, C., Soudant, P., Huvet, A., Duflos, G., Paul-Pont, I., 2016. Microplastics in seafood: Benchmark

Chapter 6—Prey Fish

protocol for their extraction and characterization. Environmental Pollution 215, 223–233.
<https://doi.org/10.1016/j.envpol.2016.05.018>

Ding, J., Zhang, S., Razanajatovo, R.M., Zou, H., Zhu, W., 2018. Accumulation, tissue distribution, and biochemical effects of polystyrene microplastics in the freshwater fish red tilapia (*Oreochromis niloticus*). Environmental Pollution 238, 1–9.
<https://doi.org/10.1016/j.envpol.2018.03.001>

Foekema, E.M., De Gruijter, C., Mergja, M.T., van Franeker, J.A., Murk, A.J., Koelmans, A.A., 2013. Plastic in North Sea fish. Environ. Sci. Technol. 47, 8818–8824.
<https://doi.org/10.1021/es400931b>

Foley, C.J., Feiner, Z.S., Malinich, T.D., Höök, T.O., 2018. A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. Science of The Total Environment 631–632, 550–559. <https://doi.org/10.1016/j.scitotenv.2018.03.046>

Froese, R., Pauly, D., 2019. FishBase [WWW Document]. URL www.fishbase.org

Greenfield, B.K., Allen, R.M., 2013. Polychlorinated biphenyl spatial patterns in San Francisco Bay forage fish. Chemosphere 90, 1693–1703. <https://doi.org/10.1016/j.chemosphere.2012.09.066>

Güven, O., Gökdağ, K., Jovanović, B., Kideyş, A.E., 2017. Microplastic litter composition of the Turkish territorial waters of the Mediterranean Sea, and its occurrence in the gastrointestinal tract of fish. Environmental Pollution 223, 286–294.
<https://doi.org/10.1016/j.envpol.2017.01.025>

Hermsen, E., Pompe, R., Besseling, E., Koelmans, A.A., 2017. Detection of low numbers of microplastics in North Sea fish using strict quality assurance criteria. Marine Pollution Bulletin 122, 253–258. <https://doi.org/10.1016/j.marpolbul.2017.06.051>

Hipfner, J.M., Galbraith, M., Tucker, S., Studholme, K.R., Domalik, A.D., Pearson, S.F., Good, T.P., Ross, P.S., Hodum, P., 2018. Two forage fishes as potential conduits for the vertical transfer of microfibres in Northeastern Pacific Ocean food webs. Environmental Pollution 239, 215–222.
<https://doi.org/10.1016/j.envpol.2018.04.009>

Isobe, A., Uchida, K., Tokai, T., Iwasaki, S., 2015. East Asian seas: A hot spot of pelagic microplastics. Marine Pollution Bulletin 101, 618–623. <https://doi.org/10.1016/j.marpolbul.2015.10.042>

Jabeen, K., Li, B., Chen, Q., Su, L., Wu, C., Hollert, H., Shi, H., 2018. Effects of virgin microplastics on goldfish (*Carassius auratus*). Chemosphere 213, 323–332.
<https://doi.org/10.1016/j.chemosphere.2018.09.031>

Chapter 6—Prey Fish

- Jabeen, K., Su, L., Li, J., Yang, D., Tong, C., Mu, J., Shi, H., 2017. Microplastics and mesoplastics in fish from coastal and fresh waters of China. *Environmental Pollution* 221, 141–149. <https://doi.org/10.1016/j.envpol.2016.11.055>
- Jacob, H., Gilson, A., Lanctôt, C., Besson, M., Metian, M., Lecchini, D., 2019. No Effect of Polystyrene Microplastics on Foraging Activity and Survival in a Post-larvae Coral-Reef Fish, *Acanthurus triostegus*. *Bull Environ Contam Toxicol* 102, 457–461. <https://doi.org/10.1007/s00128-019-02587-0>
- Jin, Y., Xia, J., Pan, Z., Yang, J., Wang, W., Fu, Z., 2018. Polystyrene microplastics induce microbiota dysbiosis and inflammation in the gut of adult zebrafish. *Environmental Pollution* 235, 322–329. <https://doi.org/10.1016/j.envpol.2017.12.088>
- Jovanović, B., Gökdağ, K., Güven, O., Emre, Y., Whitley, E.M., Kideys, A.E., 2018. Virgin microplastics are not causing imminent harm to fish after dietary exposure. *Marine Pollution Bulletin* 130, 123–131. <https://doi.org/10.1016/j.marpolbul.2018.03.016>
- Kimmerer, W., 2015. Baylands Ecosystem Habitat Goals Science Update: Appendix 3.1 – Case Study Northern Anchovy (*Engraulis mordax*).
- LeMoine, C.M.R., Kelleher, B.M., Lagarde, R., Northam, C., Elebute, O.O., Cassone, B.J., 2018. Transcriptional effects of polyethylene microplastics ingestion in developing zebrafish (*Danio rerio*). *Environmental Pollution* 243, 591–600. <https://doi.org/10.1016/j.envpol.2018.08.084>
- Lusher, A., Hollman, P.C.H., Mendoza-Hill, J., 2017. Microplastics in fisheries and aquaculture: Status of knowledge on their occurrence and implications for aquatic organisms and food safety, FAO fisheries and aquaculture technical paper. Food and Agriculture Organization of the United Nations, Rome.
- Lusher, A.L., McHugh, M., Thompson, R.C., 2013. Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. *Marine Pollution Bulletin* 67, 94–99. <https://doi.org/10.1016/j.marpolbul.2012.11.028>
- Malinich, T.D., Chou, N., Sepúlveda, M.S., Höök, T.O., 2018. No evidence of microplastic impacts on consumption or growth of larval *Pimephales promelas*: No microplastic impacts on consumption or growth of larvae. *Environ Toxicol Chem* 37, 2912–2918. <https://doi.org/10.1002/etc.4257>
- Mattsson, K., Johnson, E.V., Malmendal, A., Linse, S., Hansson, L.-A., Cedervall, T., 2017. Brain damage and behavioural disorders in fish induced by plastic nanoparticles delivered through the food chain. *Sci Rep* 7, 11452. <https://doi.org/10.1038/s41598-017-10813-0>

Chapter 6—Prey Fish

Messinetti, S., Mercurio, S., Scarì, G., Pennati, A., Pennati, R., 2019. Ingested microscopic plastics translocate from the gut cavity of juveniles of the ascidian *Ciona intestinalis*. The European Zoological Journal 86, 189–195. <https://doi.org/10.1080/24750263.2019.1616837>

Nelms, S.E., Galloway, T.S., Godley, B.J., Jarvis, D.S., Lindeque, P.K., 2018. Investigating microplastic trophic transfer in marine top predators. Environmental Pollution 238, 999–1007. <https://doi.org/10.1016/j.envpol.2018.02.016>

Ory, N., Chagnon, C., Felix, F., Fernández, C., Ferreira, J.L., Gallardo, C., Garcés Ordóñez, O., Henostroza, A., Laaz, E., Mizraji, R., Mojica, H., Murillo Haro, V., Ossa Medina, L., Preciado, M., Sobral, P., Urbina, M.A., Thiel, M., 2018. Low prevalence of microplastic contamination in planktivorous fish species from the southeast Pacific Ocean. Marine Pollution Bulletin 127, 211–216. <https://doi.org/10.1016/j.marpolbul.2017.12.016>

Ory, N.C., Sobral, P., Ferreira, J.L., Thiel, M., 2017. Amberstripe scad *Decapterus muroadsi* (*Carangidae*) fish ingest blue microplastics resembling their copepod prey along the coast of Rapa Nui (Easter Island) in the South Pacific subtropical gyre. Science of The Total Environment 586, 430–437. <https://doi.org/10.1016/j.scitotenv.2017.01.175>

Pannetier, P., Cachot, J., Clérandeau, C., Faure, F., Van Arkel, K., de Alencastro, L.F., Levasseur, C., Sciacca, F., Bourgeois, J.-P., Morin, B., 2019a. Toxicity assessment of pollutants sorbed on environmental sample microplastics collected on beaches: Part I-adverse effects on fish cell line. Environmental Pollution 248, 1088–1097. <https://doi.org/10.1016/j.envpol.2018.12.091>

Pannetier, P., Morin, B., Clérandeau, C., Laurent, J., Chapelle, C., Cachot, J., 2019b. Toxicity assessment of pollutants sorbed on environmental microplastics collected on beaches: Part II-adverse effects on Japanese medaka early life stages. Environmental Pollution 248, 1098–1107. <https://doi.org/10.1016/j.envpol.2018.10.129>

Peixoto, D., Amorim, J., Pinheiro, C., Oliva-Teles, L., Varó, I., de Medeiros Rocha, R., Vieira, M.N., 2019. Uptake and effects of different concentrations of spherical polymer microparticles on *Artemia franciscana*. Ecotoxicology and Environmental Safety 176, 211–218. <https://doi.org/10.1016/j.ecoenv.2019.03.100>

Peters, C.A., Thomas, P.A., Rieper, K.B., Bratton, S.P., 2017. Foraging preferences influence microplastic ingestion by six marine fish species from the Texas Gulf Coast. Marine Pollution Bulletin 124, 82–88. <https://doi.org/10.1016/j.marpolbul.2017.06.080>

Chapter 6—Prey Fish

Pitt, J.A., Trevisan, R., Massarsky, A., Kozal, J.S., Levin, E.D., Di Giulio, R.T., 2018. Maternal transfer of nanoplastics to offspring in zebrafish (*Danio rerio*): A case study with nanopolystyrene. *Science of The Total Environment* 643, 324–334. <https://doi.org/10.1016/j.scitotenv.2018.06.186>

Rainieri, S., Conlledo, N., Larsen, B.K., Granby, K., Barranco, A., 2018. Combined effects of microplastics and chemical contaminants on the organ toxicity of zebrafish (*Danio rerio*). *Environmental Research* 162, 135–143. <https://doi.org/10.1016/j.envres.2017.12.019>

Roch, S., Brinker, A., 2017. Rapid and efficient method for the detection of microplastic in the gastrointestinal tract of fishes. *Environ. Sci. Technol.* 51, 4522–4530. <https://doi.org/10.1021/acs.est.7b00364>

Rochman, C.M., Tahir, A., Williams, S.L., Baxa, D.V., Lam, R., Miller, J.T., Teh, F.-C., Werorilangi, S., Teh, S.J., 2015. Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. *Sci Rep* 5, 14340. <https://doi.org/10.1038/srep14340>

Romano, N., Ashikin, M., Teh, J.C., Syukri, F., Karami, A., 2018. Effects of pristine polyvinyl chloride fragments on whole body histology and protease activity in silver barb *Barbodes gonionotus* fry. *Environmental Pollution* 237, 1106–1111. <https://doi.org/10.1016/j.envpol.2017.11.040>

Romeo, T., Pietro, B., Pedà, C., Consoli, P., Andaloro, F., Fossi, M.C., 2015. First evidence of presence of plastic debris in stomach of large pelagic fish in the Mediterranean Sea. *Marine Pollution Bulletin* 95, 358–361. <https://doi.org/10.1016/j.marpolbul.2015.04.048>

Rummel, C.D., Löder, M.G.J., Fricke, N.F., Lang, T., Griebeler, E.-M., Janke, M., Gerdts, G., 2016. Plastic ingestion by pelagic and demersal fish from the North Sea and Baltic Sea. *Marine Pollution Bulletin* 102, 134–141. <https://doi.org/10.1016/j.marpolbul.2015.11.043>

Sedlak, M.D., Greig, D.J., 2012. Perfluoroalkyl compounds (PFCs) in wildlife from an urban estuary. *J. Environ. Monit.* 14, 146–154. <https://doi.org/10.1039/C1EM10609K>

Setälä, O., Fleming-Lehtinen, V., Lehtiniemi, M., 2014. Ingestion and transfer of microplastics in the planktonic food web. *Environmental Pollution* 185, 77–83. <https://doi.org/10.1016/j.envpol.2013.10.013>

Tosetto, L., Williamson, J.E., Brown, C., 2017. Trophic transfer of microplastics does not affect fish personality. *Animal Behaviour* 123, 159–167. <https://doi.org/10.1016/j.anbehav.2016.10.035>

Vendel, A.L., Bessa, F., Alves, V.E.N., Amorim, A.L.A., Patrício, J., Palma, A.R.T., 2017. Widespread microplastic ingestion by fish assemblages in tropical estuaries subjected to anthropogenic

Chapter 6—Prey Fish

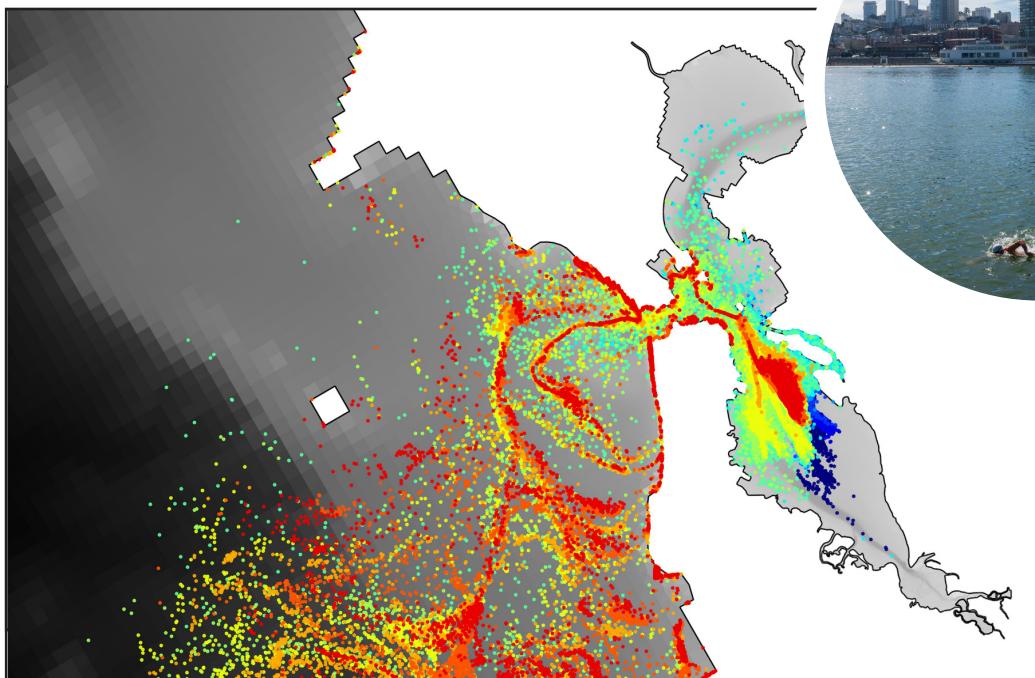
- pressures. *Marine Pollution Bulletin* 117, 448–455.
<https://doi.org/10.1016/j.marpolbul.2017.01.081>
- Viršek, M.K., Lovšin, M.N., Koren, Š., Kržan, A., Peterlin, M., 2017. Microplastics as a vector for the transport of the bacterial fish pathogen species *Aeromonas salmonicida*. *Marine Pollution Bulletin* 125, 301–309. <https://doi.org/10.1016/j.marpolbul.2017.08.024>
- Wagner, J., Wang, Z.-M., Ghosal, S., Rochman, C., Gassel, M., Wall, S., 2017. Novel method for the extraction and identification of microplastics in ocean trawl and fish gut matrices. *Anal. Methods* 9, 1479–1490. <https://doi.org/10.1039/C6AY02396G>
- Wen, B., Jin, S.-R., Chen, Z.-Z., Gao, J.-Z., Liu, Y.-N., Liu, J.-H., Feng, X.-S., 2018. Single and combined effects of microplastics and cadmium on the cadmium accumulation, antioxidant defence and innate immunity of the discus fish (*Symphysodon aequifasciatus*). *Environmental Pollution* 243, 462–471. <https://doi.org/10.1016/j.envpol.2018.09.029>
- Yin, L., Chen, B., Xia, B., Shi, X., Qu, K., 2018. Polystyrene microplastics alter the behavior, energy reserve and nutritional composition of marine jacopever (*Sebastes schlegelii*). *Journal of Hazardous Materials* 360, 97–105. <https://doi.org/10.1016/j.jhazmat.2018.07.110>
- Zhang, S., Ding, J., Razanajatovo, R.M., Jiang, H., Zou, H., Zhu, W., 2019. Interactive effects of polystyrene microplastics and roxithromycin on bioaccumulation and biochemical status in the freshwater fish red tilapia (*Oreochromis niloticus*). *Science of The Total Environment* 648, 1431–1439. <https://doi.org/10.1016/j.scitotenv.2018.08.266>
- Zheng, K., Fan, Y., Zhu, Z., Chen, G., Tang, C., Peng, X., 2019. Occurrence and species-specific distribution of plastic debris in wild freshwater fish from the Pearl River Catchment, China. *Environmental Toxicology and Chemistry etc.* 4437. <https://doi.org/10.1002/etc.4437>

CHAPTER

7

TRANSPORT OF MICROPARTICLES AND MICROPLASTICS IN SAN FRANCISCO BAY AND MARINE SANCTUARIES

by Rusty Holleman



Highlights

- ◆ This study developed and applied numerical models to estimate the dispersal and fate of microparticles and microplastics in San Francisco Bay and the adjacent National Marine Sanctuaries. The three-dimensional hydrodynamic model is unique in its spatial coverage from small-scale sloughs and mud flats within the Bay, to shelf-scale dynamics in the coastal ocean.
- ◆ The model incorporated estimated microparticle loads from stormwater and wastewater, and simulated particle trajectories throughout the Bay and into the coastal ocean. The rising and settling characteristics of particles were estimated based on laboratory measurements of composition, shape, and size.
- ◆ Export of particles to the coastal ocean was found to be highly sensitive to particle buoyancy, and even minimal sinking rates led to retention of particles within the Bay.
- ◆ Both buoyant and non-buoyant particles entering the Bay via confined sloughs and river channels were predicted to become beached or trapped before reaching the open Bay. While this fate may occur in reality, it also points to the limits of the models and a lack of information on particle–shoreline interaction at small scales.
- ◆ The model and the manta trawl particle abundance data are in good agreement, showing that the average abundance of particles is higher inside the Bay than in the coastal ocean. The difference is more pronounced in the model, and the presence of a coastal plume is not readily apparent in the trawl data, possibly due to the transient nature of the coastal plume.
- ◆ In general, the model and field data showed greater abundance of microparticles in wet weather than dry weather, particularly in South Bay. Several notable discrepancies may be due, in part, to the limitations of the manta trawl data in characterizing fibers, and secondly, the location of some field sites in highly localized convergence zones too small for the model to predict.
- ◆ Good agreement was also observed between the model and field data with both the predicted near-bed abundances and measured sediment concentrations showing the greatest level of microparticles in Lower South Bay, a region which is both slowly flushed and strongly influenced by wastewater and urban stormwater discharges.
- ◆ The model evaluated the transport of microparticles from the Bay to the coastal ocean. An important finding was that denser, settling particles are retained in the Bay, and only 20% of passive, neutrally buoyant particles make the journey from South Bay through the Golden Gate. If the model accurately describes conditions in the Bay, these findings are of concern for Bay benthos and their predators.

- ◆ Even after substantial averaging in space and time the model predictions showed persistent spatial gradients, which suggests that comprehensive spatial sampling would require a large number of stations.
- ◆ From both the model results and field data, we concluded that microparticles originating in San Francisco Bay do, on occasion, reach the majority of the nearby National Marine Sanctuaries. However, only buoyant particles are likely to travel any notable distance beyond the Golden Gate. The buoyant particles are efficiently transported in the freshwater plume leaving the Bay, often taking them northward along the coast, or dispersing them south and west by regional winds and coastal currents.

Objectives

Transport of microparticles and microplastics in San Francisco Bay and adjacent coastal waters is an essential component of understanding the fate of particles in the system, and the export of microparticles to the National Marine Sanctuaries and the global ocean. Most microplastic transport modeling to date has focused on global-scale circulation and the transport of surface-bound, buoyant particles (e.g., Lebreton et al., 2018). Past studies, including Sutton et al. (2016) as well as the present study (Chapters 2 Stormwater and 3 Wastewater), show that particles entering the Bay exhibit a wide variety of characteristics and likely have a wide range of settling and rising velocities. Studies of sediment transport dynamics suggest that transport and fate of material in an estuarine setting is highly dependent on settling and rising velocities (Williams et al., 2004), and only a fully three-dimensional analysis of transport can capture the breadth of relevant mechanisms (Scheu et al., 2015).

Through mechanistic numerical modeling and analysis of field data, this portion of the San Francisco Bay Microplastics Project addressed the following objectives.

- 1. Characterize physical behavior of microparticles and microplastics based on the characteristics of sampled particles.** Particle size, shape, and composition can be used to estimate how quickly particles may rise to the surface or settle to the bed once in the open waters of the Bay. Characterizing this physical behavior will improve our understanding of the processes relevant to transport and fate of these particles, an element of the third management question (MQ3) articulated in the RMP Microplastic Monitoring and Science Strategy for San Francisco Bay (Sutton and Sedlak, 2017).
- 2. Estimate ambient microparticle concentrations.** Transport models can be used to estimate particle concentrations throughout the Bay. Where field samples overlap with model predictions, the field samples can be used to evaluate the skill of the model. In other places and times, the model predictions can identify potential hot spots worth investigating in the future. This is relevant to MQ1, How much microplastic pollution is there in the Bay and in the surrounding ocean?
- 3. Estimate how quickly particles are exported from the Bay and where they might go.** Export of microparticles from the Bay is highly dependent on where they enter the Bay and the physical characteristics of the particles. Together, these factors determine which particles may reach the National Marine Sanctuaries, as opposed to depositing or beaching within the Bay. This modeling effort informs an element of MQ3, concerning whether the Bay is a net source of microplastics to the ocean.

Methods

Estimating settling and rising velocities

An essential aspect of microparticle transport is the rate at which individual particles rise or fall in the water column. This is analogous to the settling velocity of grains of sediment, though in the case of microparticles and microplastics the effect may be either a downward (i.e., settling), or upward (i.e., rising) motion, depending on the density of the particle. For the present study, we have applied the methods of Wäldschlagger and Schüttrumpf (2019, hereafter WS19) to relate particle measurements collected in the laboratory to rising and settling velocities. WS19 obtained samples of microplastics with varying sizes, densities, and shapes. Each sample was placed in an upright water-filled cylinder and allowed to rise or fall under its own buoyancy. They then developed semi-empirical relationships between particle characteristics and the resulting rising or settling velocity.

The relevant particle characteristics for the parameterization of WS19 are:

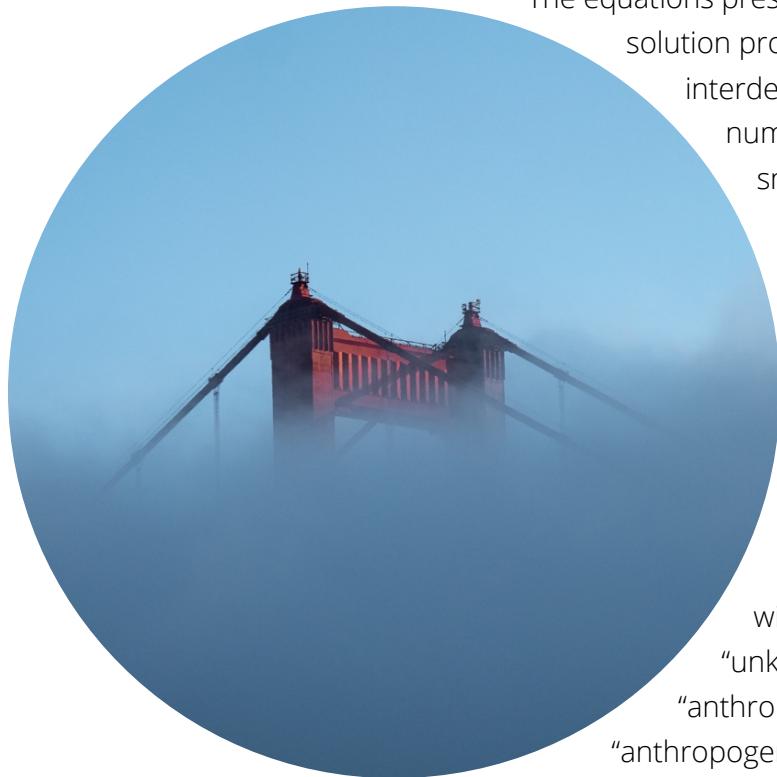
- ◆ Bulk density;
- ◆ Corey Shape Factor, defined as $c/\sqrt{a \cdot b}$ where a is the longest dimension of the particle, b the intermediate dimension, and c the shortest dimension; and
- ◆ Powers Roundness, a qualitative smoothness scale from 1 (rough, angular) to 6 (smooth, round).

Bulk density for particles was estimated based on the Raman and FTIR spectroscopy composition data. For 25 of the top 30 most commonly identified materials in the stormwater and wastewater datasets, a nominal bulk density was determined from general literature (Table 7.1). In some cases, additional information allowed a more specific estimate, such as interpreting rubber as tire rubber. In the case of paint, spectroscopy did not generally resolve the type of paint. Most paints dry to a density heavier than water, and the density used is representative of oil-based paint. All particles categorized as foam were given a minimal density of 0.10 g/cm^3 to account for included gas bubbles in the expanded plastic.

Table 7.1. Material density

Material	Density (g/cm³)
rubber	1.15
polyethylene	0.94
cellulosic (anthropogenic or natural), organic natural material	1.50
polyester (fiber)	1.38
cotton	1.50
cellulose acetate	1.30
polypropylene	0.90
acrylic	1.19
nylon	1.15
ethylene vinyl acetate copolymer	0.94
polyvinyl chloride	1.39
glass	2.60
Polyethylene terephthalate (non-fiber)	1.38
Polyurethane (non-foam)	1.10
wool	1.29
Polyvinyl butyral	1.08
Polystyrene (non-foam)	1.03
paint	1.40
Methyl vinyl ether copolymers	1.05
Acrylonitrile butadiene styrene	1.07
Polyethylene co-acrylic acid	0.96
Polyethylene/polypropylene copolymer	0.92
Polytetrafluoroethylene	2.20
all foam	0.10

Laboratory data included two dimensions, length and width, for most particles. Particle category was used to estimate the third (minor) dimension. For fibers, WS19 uses only the diameter of the fiber, taken to be the smaller of the reported length and width dimensions. Films were given a minor dimension of 0.05 mm, with the reported length and width used as the major and intermediate dimensions. Fiber bundles were given a minor dimension equal to the mean width of individually counted fibers. Fragments, spheres, and foams were assumed to have a minor dimension equal to the intermediate dimension. WS19 found the Powers Roundness to have only a small predictive power compared to density, size, and shape factor. Nevertheless, we included a rough estimate of the roundness, giving spheres a smooth value of 6, and all other particles a moderate value of 3.



The equations presented in WS19 required an iterative solution procedure because of the nonlinear interdependence between the Reynolds number (a measure of turbulent vs.

smooth flow) and the settling velocity.

A simple iterative solver for the settling velocity was found to quickly converge in all cases.

Typical ambient water density was set to a constant 1.025 g/cm³.

For the purposes of estimating settling and rising velocity, particles that were labeled “rubbery” and with a plastic type of “not identified,” “unknown potentially rubber,”

“anthropogenic synthetic,” “unknown,” and “anthropogenic unknown” were assumed to be rubber and given a corresponding density. Particles with

these plastics types but not noted as rubbery were omitted from the rising and settling velocity calculations. As described below, particles for which a rising or settling velocity could not be calculated were assumed to follow a distribution of velocities defined by the particles for which velocity estimates were possible.

Hydrodynamic model

In order to adequately capture the three-dimensional currents of San Francisco Bay and the adjacent National Marine Sanctuaries, we developed a three-dimensional hydrodynamic model of the region. This model departs from many San Francisco Bay applications in that it must seamlessly allow transport between the Bay and the coastal environment, capturing the tide- and river-driven dynamics within the Bay, as well as inertial- and wind-driven currents in the coastal ocean. To this end, we developed the model using the SUNTANS hydrodynamic model (Fringer et al., 2006), which has been successfully used in previous San Francisco Bay model applications (Chua and Fringer, 2011; Holleman and Stacey, 2014), as well as coastal domains (Fringer et al., 2006). This model platform divides the vertical dimension of the water column into layers ranging from 0.4 m thick near the surface to 460 m thick in the deepest regions of the model domain. This arrangement enables the model to efficiently capture currents across the wide range of depths in the domain, from intertidal mud flats to 4000 m depths off the shelf.

Figure 7.1 shows the domain and computational grid of the hydrodynamic model. Within the Bay the grid is taken from a previous model application (Holleman et al., 2017), with horizontal resolution ranging from 15 m to 800 m. The coastal portion of the domain is a coast-aligned rectilinear grid with a resolution of 2000 m. Freshwater boundary conditions include:

- ◆ Tidal flows from the Sacramento-San Joaquin Delta, with 15-minute data pulled from USGS flow stations for the time period modeled;
- ◆ Discharges from the 12 largest wastewater treatment plants, using seasonal climatologies of discharge from previous years; and
- ◆ Stormwater flows from Coyote Creek, Alameda Flood Control Channel, Napa River, and Guadalupe River, using 15-minute data from USGS gauging stations.

The ocean boundary is forced with a combination of water level and three-dimensional fluxes, following the methodology of Rayson et al. (2018). The portion of the ocean boundary within 10 km of the shoreline is forced with spatially varying tidal water level extracted from the West Coast model from the Oregon Tidal Prediction Software (OTPS) suite. The remainder of the ocean boundary is forced with a combination of two-dimensional (depth-averaged) tidal fluxes from the same OTPS model and non-tidal three-dimensional fluxes from the global circulation model HYCOM. Boundary conditions for salinity and temperature are also taken from HYCOM. Wind forcing is derived from the data set of King (2019) and the regional COAMPS model. The combination of water-level boundaries and flux boundaries avoids issues with numerical instability in deep water (flux boundaries), while also avoiding long-term drift of the free surface (via water-level boundaries). Testing with only the flux boundaries showed that over the span of 15 days the water level drift was on the order of 0.2 m, indicating that errors in the fluxes were relatively small and easily corrected by imposing water level on a short section of the boundary.

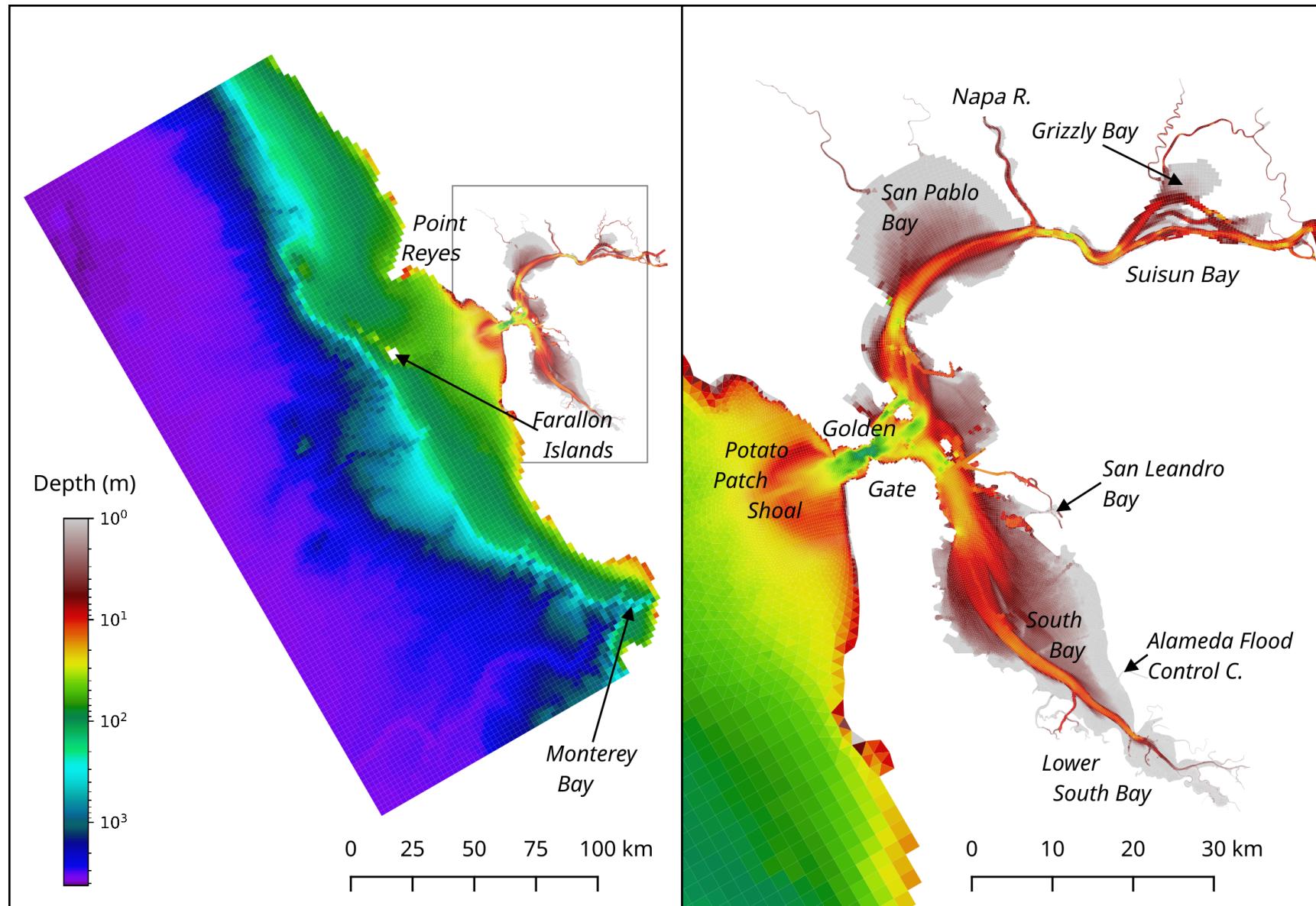


Figure 7.1. Hydrodynamic model domain, with bathymetry shown on the computational grid.

The hydrodynamic model was run from June 10, 2017, through July 5, 2018. This period allows two months of model spin-up before the first surface water samples were collected and covers the transitions from the dry season to wet and back. The model is parallelized with MPI (message passing interface), and on 16 cores runs at approximately 90x real-time.

The model was calibrated for tidal water level at NOAA gauges within the Bay and at Point Reyes, generally achieving tidal amplitude within 5% of observed and phase lags within 20 minutes. The predicted salinity field was compared to monthly observations along the spine of the Bay for both depth-averaged salinity and vertical salinity gradient. Separate calibration and validation periods were not run. Calibration was achieved by adjusting the method for setting bathymetry on the computational grid and selectively refining the grid where salinity gradients indicated poor transport resolution.

Particle tracking model

A particle tracking model was used to simulate the transport of microparticles in the Bay and coastal ocean. While these simulations could in part be carried out using a scalar transport model, the particle tracking model provides more control over the analysis, including post-hoc adjustment of source concentrations and maximum age of particles. The FISH-PTM model (Ketefian et al., 2016) was chosen for its speed, flexibility, track record of successful application in San Francisco Bay, and compatibility with SUNTANS hydrodynamic data.

Particles were introduced in the simulation at locations corresponding to each of the eight sampled wastewater treatment plant discharges, and at four significant stormwater-driven tributaries. The wastewater locations correspond to

- ◆ Central Contra Costa Sanitary District (CCCSD),
- ◆ East Bay Dischargers Authority (EBDA),
- ◆ East Bay Municipal Utility District (EBMUD),
- ◆ Fairfield-Suisun Sewer District,
- ◆ Palo Alto Regional Water Quality Control Plant,
- ◆ San José/Santa Clara Wastewater Facility,
- ◆ San Francisco Public Utilities Commission (SFPUC), and
- ◆ Sunnyvale Water Pollution Control Plant.

The stormwater entry points correspond to Coyote Creek, Alameda Flood Control Channel, Napa River, and Guadalupe River.

Chapter 7 – Transport Model

Seven rising and settling velocities were used in order to approximately span the range of particle rising and settling rates estimated from the stormwater and effluent data. The simulated rates were:

- ◆ rising at 50 mm/s, 5 mm/s, and 0.5 mm/s;
- ◆ passive (0 mm/s); and
- ◆ settling at 50 mm/s, 5 mm/s, and 0.5 mm/s.

In the case of settling velocities, the velocity was prescribed down to within 0.5 m of the sediment bed, below which the particle was passive, and in the case of rising velocities, the velocity was prescribed up to within 0.5 m of the surface. This zone of passive behavior was necessary to avoid simulation issues with particles erroneously traveling through the bed or the water's surface. In the case of the water surface, one would expect that highly buoyant particles would become highly concentrated in the top millimeters or centimeters of the water column (motivating the use of a manta trawl for surface sampling). A consequence of this buffer region of passive behavior is that model particles are not concentrated at the surface beyond the 0.5 m thickness of the buffer layer, and instead will remain evenly distributed within the top 0.5 m. Similarly, dense particles will not concentrate further once they enter the bottom 0.5 m of the water column. For this reason, model-data comparisons rely on abundance levels in particles/m², integrating over the top or bottom 0.5 m of the water column. This approach decreases the sensitivity of the model-data comparison to small changes or errors in vertical distribution.

For all particle release locations (the eight wastewater discharges and four stormwater sites), particles were released at a constant rate of five particles per hour. By decoupling the release rate from the estimated particle load, a single time-consuming particle simulation can be rescaled after the fact to represent different assumptions on loads. The scaling approach is described below.

To satisfy the goal of simulating representative conditions spanning the full year of the hydrodynamic simulation, 12 separate particle tracking simulation periods were run. For each run, particles were released for 30 days, and then simulated for an additional 30 days with no additional particle releases. This allowed greater flexibility in distributing the simulation workload and supported analysis of particle lifetimes up to 60 days. In practice, particle tracking results were averaged over 14-day intervals, which reduced the maximum particle age in the analysis to 44 days (the sum is less than 60 due to variability in the length of a month). Each particle tracking simulation ran at approximately 180x real-time, for a total of 670 hours of computation.

Particle analysis

Since the particle releases were constant in time and across release locations (five particles per hour per location), additional processing was required to account for differences in concentration and flow across locations and through time.

All release locations were assumed to have a constant concentration of microparticles in each rising/settling velocity class throughout time. These concentrations were estimated for stormwater as a whole by considering the combined samples from all stormwater sampling sites. Wastewater data from each of the eight plants were kept separate, resulting in a separate concentration profile for each plant.

Particles in samples come from a population distribution of varying sizes, shape categories, and composition. Particles in blank samples do not necessarily have the same population distribution, and wastewater (Chapter 3) and stormwater (Chapter 2) data suggest that the distribution of particle morphology in blanks was distinct from samples. For this reason, as in previous chapters, blank counts were analyzed on a per-category basis. Due to the limited number and size of the blank samples, we chose not to further divide particle counts into groups finer than category. Unlike previous chapters, the modeling approach depends on the distribution of rising and settling velocities. A straightforward blank subtraction for each category would still result in the need to choose *which* particles to remove, because the particles in each category are still distinct with respect to rising or settling velocities. These constraints led to a different approach for handling blanks in the model. Rather than subtracting a blank level for each category from the field samples, we applied a weighting factor (between zero and one) to particles in each category, which accounts for the fraction of particles estimated to be from sample contamination.

This blank strategy departs from the approach taken in previous chapters. Previous chapters strive to be transparent (communicating the full counts) but conservative (qualifying counts that are close to a calculated threshold based on blank levels). The modeling is ultimately a comparison of loads (stormwater and wastewater) and ambient levels (surface water and sediment). To make this comparison as meaningful as possible, we have used the blank samples to remove the contamination bias as much as possible.

Counts for each particle category were averaged across the blanks (including both field and laboratory blanks), yielding an estimate, $B(\text{category})$, of the blank contamination on a per-sample, per-category basis. For each particle category, the particle count, $S(\text{category})$, was averaged across all field samples. A fractional “weight” was then defined as

$$W_{\text{storm}}(\text{category}) = \frac{S(\text{category}) - B(\text{category})}{S(\text{category})}$$

This weight was applied to all particles in the field samples to get an adjusted particle count. For example, the four stormwater blank samples had an average of 37 fibers per blank. The field samples had an average of 320 fibers per sample. This led to a weight, $W_{\text{storm}}(\text{fiber}) = 0.885$. This can be interpreted as 88.5% of the sampled fibers are real, and the remaining 11.5% are due to contamination. Since each particle is distinct (with specific dimensions and composition), we could not simply omit 11.5% of the fibers (and face an arbitrary choice of which specific fibers to omit). Instead, we included all particles but when summing particle counts each sampled fiber was counted as only 0.885 fibers instead of 1.0.

Stormwater and wastewater blanks were handled similarly, with the difference that the weights and adjustments were applied separately for each wastewater plant, whereas the stormwater samples were combined. In some cases, the average field count was below the average blank count, in which case the weight was set to zero to reflect that the field data was indistinguishable from the blank data.

Flows for each release location were extracted from the hydrodynamic data. As such, stormwater flows come from USGS gauges. Wastewater discharge flow rates were taken from the same dataset as Holleman et al. (2017). While that dataset does not include reported discharges for 2017, the extrapolated baseline flows and seasonal climatology were assumed to be sufficiently accurate for wastewater discharges.

With the combination of flow and concentration, the particle tracking outputs can be scaled to an estimate of particle concentration, normalizing the constant five particles per hour release rate to the reality of time- and source-varying loads of microparticles.

Results

Rising and settling velocities

Of the 15,748 particles counted in the stormwater and wastewater field datasets, sufficient spectroscopic and morphology data were available to estimate a rising/settling velocity for 1,821 particles. Figure 7.2 shows the overall distribution of rising/settling velocities for the stormwater and wastewater datasets (normalized within each pathway). The large peak in settling particles in the stormwater was driven by the prevalence of (presumed) rubber fragments, which are both denser than water and have relatively little drag to slow their descent.

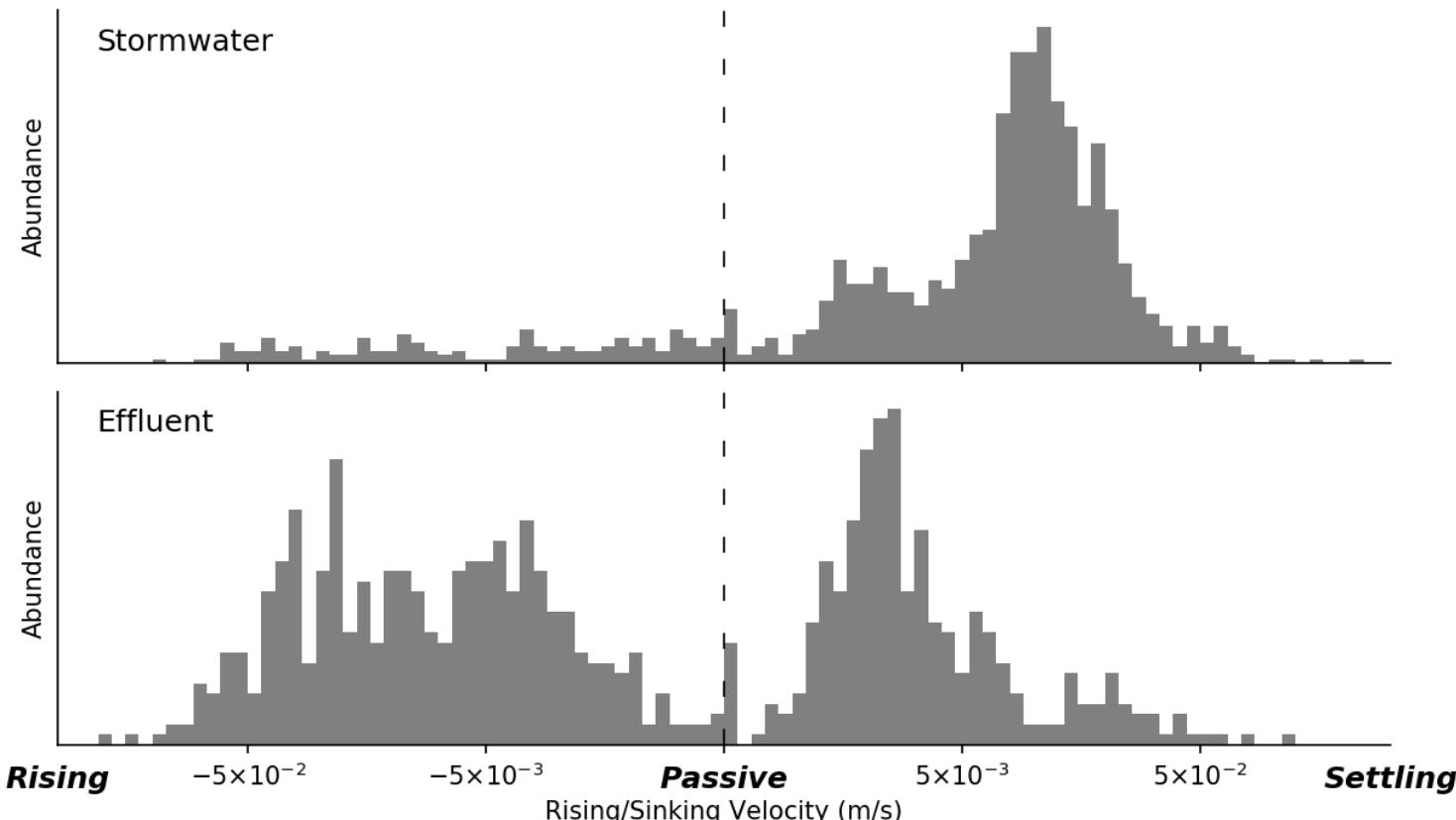


Figure 7.2. Rising and settling velocities, stratified by pathway. Distributions are normalized within each group; absolute counts were significantly higher for stormwater than for effluent (wastewater).

Whether a specific particle is expected to rise or fall depends on the density of the material (down to a particle size of approximately 1 μm , below which particles can remain in suspension indefinitely as a colloid). Figure 7.3 shows the range of rising/settling velocities for common material types. For most materials, all particles of that material fall on one side or the other of the neutral divide in the middle of the histogram. The same material may occur in both foam and non-foam categories (e.g., polyurethane), in which case the foam particles are plotted separately. Non-foam polyurethane particles are expected to sink, while expanded polyurethane foam is expected to rise. The density assumptions above also allow a rising velocity to be calculated for particles that are identified as foam but with unknown composition. We assume size and shape are the dominant factors determining rising velocities for these particles, independent of the material type.

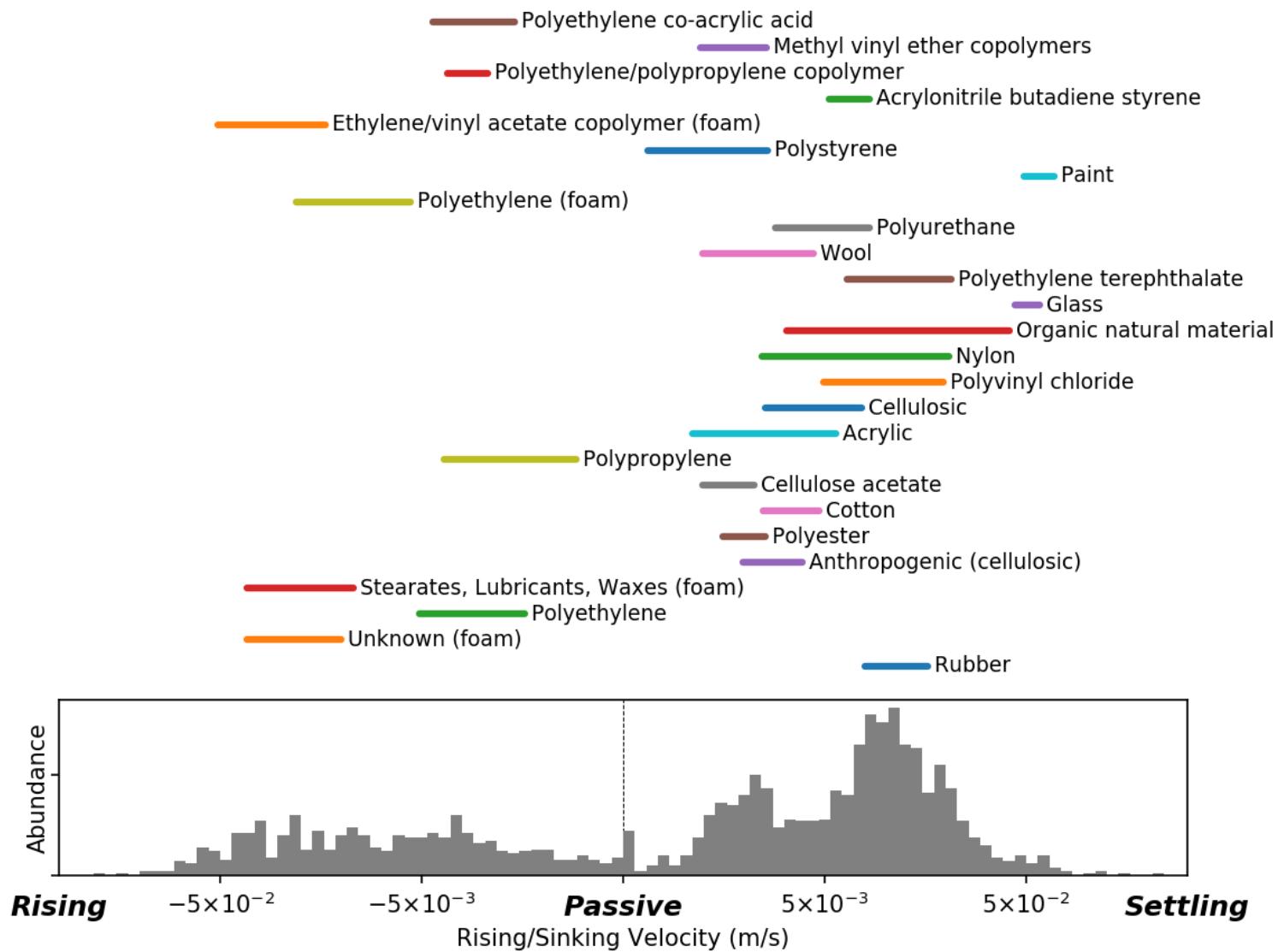


Figure 7.3. Range of rising/settling velocities by particle type. The histogram summarizes rising and settling velocities for all quantified particles in the stormwater and effluent (wastewater) data sets (absolute counts, not normalized for flow rates or sample volumes). Horizontal bars span the interquartile range and are ordered from least common (polyethylene co-acrylic acid) at the top to most common (rubber) at the bottom. Foams are a separated category due to the different density used for those particles. Uncommon particle types are omitted.

Particle shape is also an important factor in determining the estimated rising/settling velocity. Figure 7.4 shows the range of rising/settling velocities stratified by particle category. Fragments have relatively little drag, and correspondingly have a very wide range of velocities. In contrast, films and fibers have significant drag, such that they tend towards smaller, closer to passive rising/settling velocities. While spheres are low drag shapes, they appear with both buoyant and dense, sinking material types, leading to a broad range of velocities. Foams are clearly biased towards fast rising velocities due to the trapped gases, which give them light bulk densities.

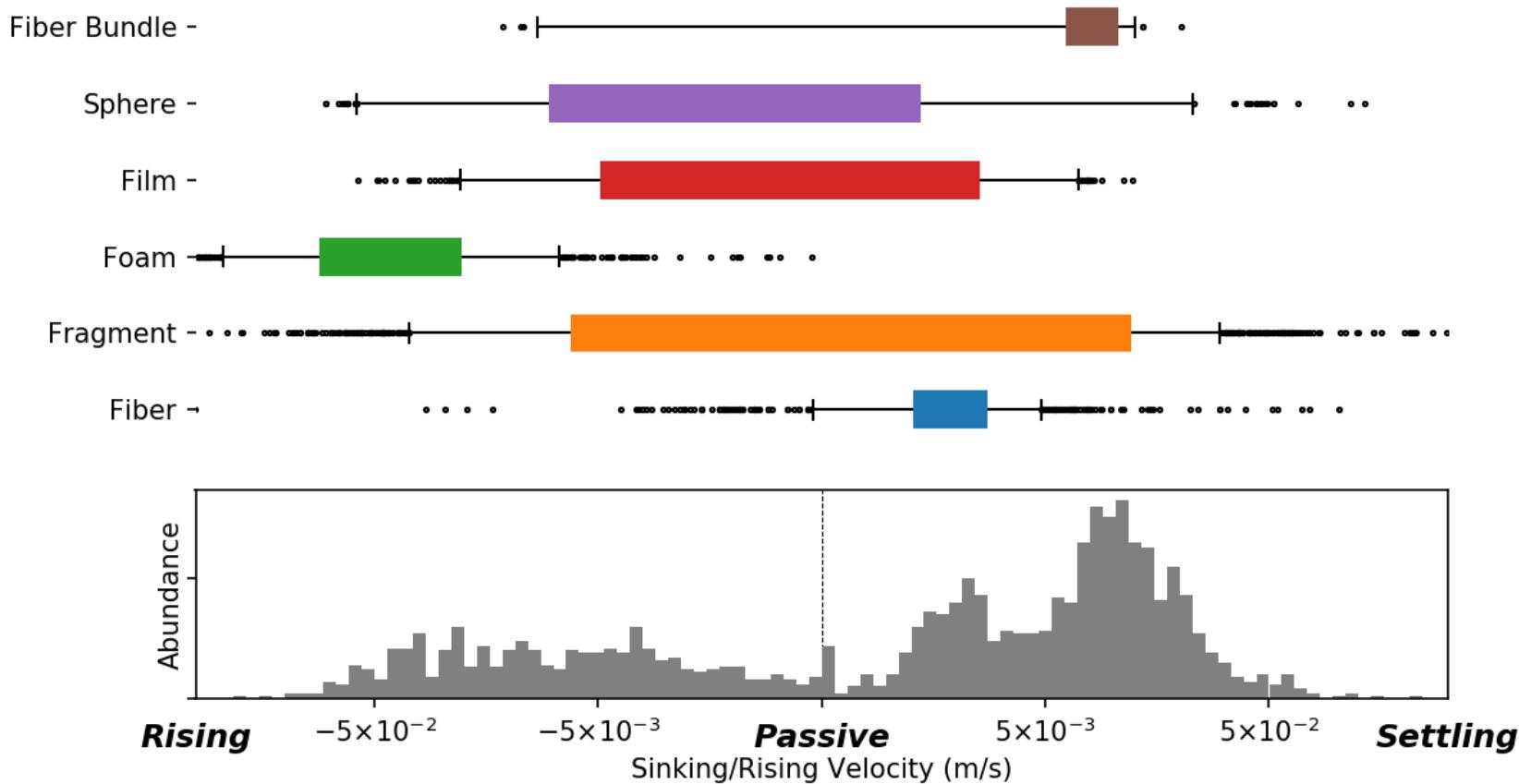


Figure 7.4. Rising/settling velocities stratified across particle categories. Each horizontal bar shows the interquartile range, with whiskers out to the 5th and 95th percentiles. The histogram, as in Figure 7.3, summarizes rising and settling velocities for all quantified particles in the stormwater and effluent (wastewater) data sets (absolute counts, not normalized for flow rates or sample volumes).

Concentration in discharges to the Bay

For the purposes of particle tracking, all wastewater and stormwater particle counts were binned by rising/settling velocity. The wide distribution of velocities motivated a similarly wide range of bins: ± 50 mm/s, ± 5 mm/s, ± 0.5 mm/s, and passive (0 mm/s). Particles in the sample data were grouped by station (the eight wastewater sites were kept distinct and the four stormwater sites were combined), binned by rising/settling velocity, and normalized by sample volume to get a concentration. Resulting concentrations are shown in Table 7.2.

Table 7.2. Microparticle concentrations in discharges to the Bay.

Discharge	Concentration (microparticles/L)						
	Rising velocity (mm/s)			Passive	Settling velocity (mm/s)		
	50	5	0.5	0	0.5	5	50
CCCS	0.0175	0.0325	0.0106	0.0000	0.0098	0.0055	0.0021
EBDA	0.0067	0.0060	0.0058	0.0000	0.0088	0.0033	0.0011
EBMUD	0.0152	0.0498	0.0093	0.0000	0.0100	0.0090	0.0012
Fairfield-Suisun	0.0000	0.0013	0.0003	0.0000	0.0015	0.0016	0.0002
Palo Alto	0.0000	0.0004	0.0001	0.0000	0.0026	0.0029	0.0001
SFPUC	0.0280	0.0925	0.0261	0.0000	0.0202	0.0136	0.0009
San Jose	0.0041	0.0110	0.0023	0.0008	0.0021	0.0022	0.0003
Sunnyvale	0.0010	0.0003	0.0007	0.0000	0.0053	0.0051	0.0000
Stormwater	0.2451	0.5137	0.5434	0.0165	1.0966	6.1955	0.5679

For reasons of practicality and availability of daily flow data, only four stormwater discharge locations were included in the hydrodynamic model. These four tributaries make up 33% of the overall stormwater flows entering the Bay downstream of the Carquinez Strait. At present, there is not a robust estimate of microplastic and microparticle loads in the large freshwater flows originating from the Sacramento-San Joaquin Delta. Due to this uncertainty, Delta flows were not considered in the particle tracking model. The stormwater concentrations in Table 7.2 were further scaled by a factor of 1/0.33 when analyzing the particle tracking output to adjust for the ratio of modeled to physical flows for the region, which effectively assumes that currently unmonitored watersheds deliver similar concentrations. Similarly, wastewater concentrations were scaled by a factor of 1/0.70, since these eight discharges account for approximately 70% of the effluent discharges Bay-wide.

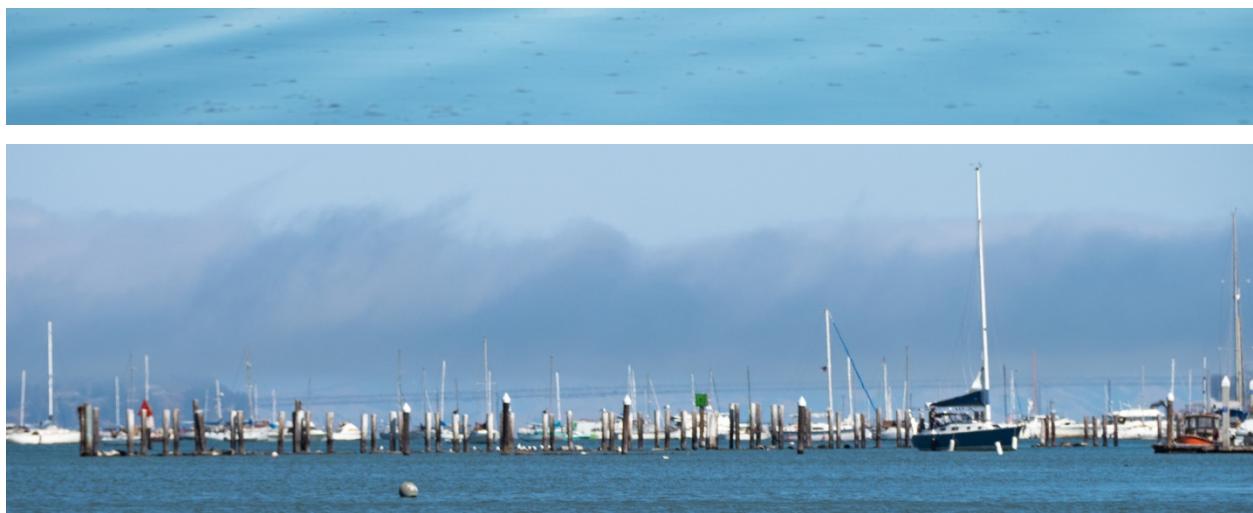
As mentioned above, the particle tracking model employed a constant release rate of five particles per hour for each rising/settling velocity and each discharge. The “true” particle

loading rate is the product of the concentrations in Table 7.2 and the time-varying flow rates from the hydrodynamic model. This difference was accounted for in a post-processing step by scaling the raw particle outputs by the ratio of the true particle loading rate to the constant loading rate of the model.

Spatial and temporal particle distributions

Particle tracking results were converted from discrete particle locations to counts per area, averaged over 15-day intervals, and spatially smoothed. Tidal amplitudes vary on a two-week cycle (the spring-neap cycle), and the averaged 15-day interval was chosen to approximate this period in order to evenly sample all parts of the biweekly cycle. Spatial smoothing is necessary due to the limited number of particles and large model domain. The smoothing amounts to applying a diffusion step, where the diffusion rate scales with the grid resolution. This results in less smoothing inside the Bay, where the grid has a finer resolution and particle counts are higher, than in the open ocean.

The analysis also assumes an upper bound on the particle age (results are shown below for upper bounds of 15 and 44 days). In the case of buoyant and passive particles, these upper bounds provide an approximate means for assessing sensitivity to particle lifetime and the potential for microparticles in the environment to accumulate microbial colonies and lose their buoyancy (Chen et al., 2019; Fazey and Ryan, 2016). In the case of settling particles, the upper bound on particle age is a rough proxy for deposition to sediment. While sediment models, including particle-based sediment models, typically parameterize deposition directly, it remains one of the more difficult aspects of calibrating sediment models and a major source of uncertainty. For deposition of microparticles with densities much closer to water and morphologies distinct from the sediment particles making up the bed, there are even fewer data. For these reasons, we chose not to model deposition, and instead characterized the distribution of near-bed concentrations over the finite lifetime of each numerical particle.



Given these limitations of the analysis, we do not expect the estimates of near-bed particle abundance to quantitatively agree with observed microparticle concentrations in the sediment. Specifically, without also modeling deposition and erosion of sediment mass, it is not possible to directly calculate a particle concentration in sediment, even if deposition were included in the simulation of the microparticles. Instead, the near-bed data presented below should be interpreted qualitatively as an indication of regions of higher or lower propensity for accumulating particles.

In the figures below, “near-surface” includes particles within the top 0.50 m of the surface of the water column, and “near-bed” includes particles within 0.50 m of the bed. Given these definitions, shallow regions (e.g., the shoals of South Bay) may count some particles in both the near-surface and near-bed categories.

DISTRIBUTION IN THE BAY

Figure 7.5 shows the near-surface microparticle distribution for dry conditions. This analysis period was shortly after the August 2017 Bay sampling campaign. The combination of minimal stormwater flows (which would deliver a disproportionate number of settling particles), and the surface-focused analysis led to very few settling particles in the distributions. Passive particles were also uncommon in the discharge concentrations (Table 7.2), which was reflected in the general lack of these particles in most parts of the Bay. Long residence time in South Bay and Lower South Bay, coupled with local discharges from wastewater and urban stormwater, led to the highest predicted abundance appearing in South Bay and Lower South Bay. The simulated CCCSD effluent also played a significant role in delivering buoyant particles to North Bay, while the effect of Fairfield-Suisun effluent was lessened by both its smaller discharge flows and the potential for particle trapping within the sloughs of Suisun Marsh.

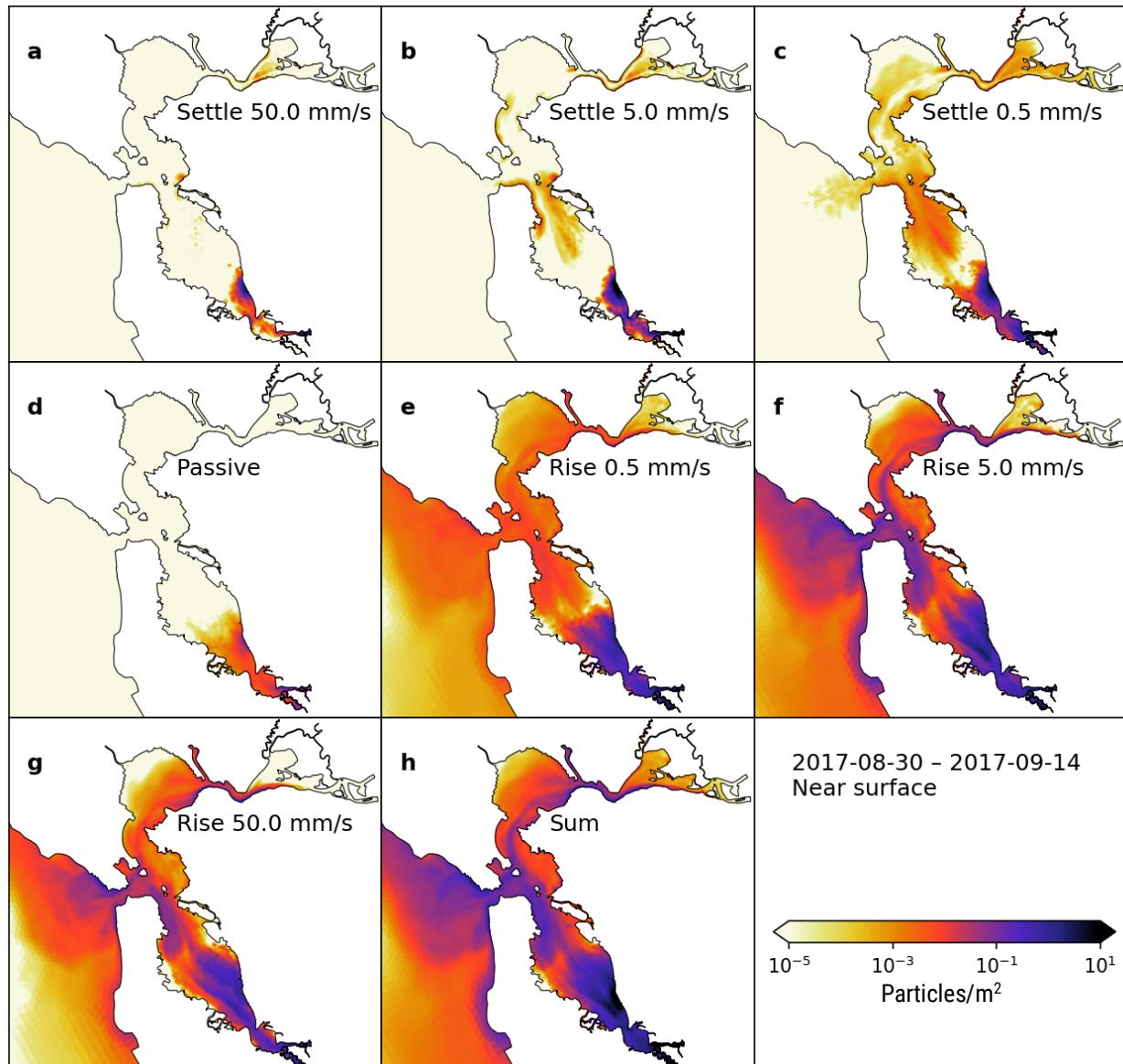


Figure 7.5. Predicted surface particle abundance during dry conditions. Only particles up to 15 days old are shown.

The distribution of particles near the bed for dry weather conditions is shown in Figure 7.6. Localized high counts of settling particles were found in South Bay and Lower South Bay, especially near the Alameda Flood Control Channel on the eastern shore of South Bay, even for the relatively low dry-season flows. As expected, the abundance of buoyant particles was generally lower than in the surface water distribution, though in the broad, shallow shoals many particles were both near the bed and near the surface and were thus included in both figures. Where Figure 7.5 clearly shows particles at the water's surface exiting the Bay, the more compact distribution of near-bed abundance in Figure 7.6 demonstrates that at depth, there was very little export. This vertical variation was a direct effect of the estuarine circulation

through the Golden Gate, in which surface flows are fresher and transport material out of the Bay, while near-bed flows are saltier and transport material into the Bay (Largier, 1996).

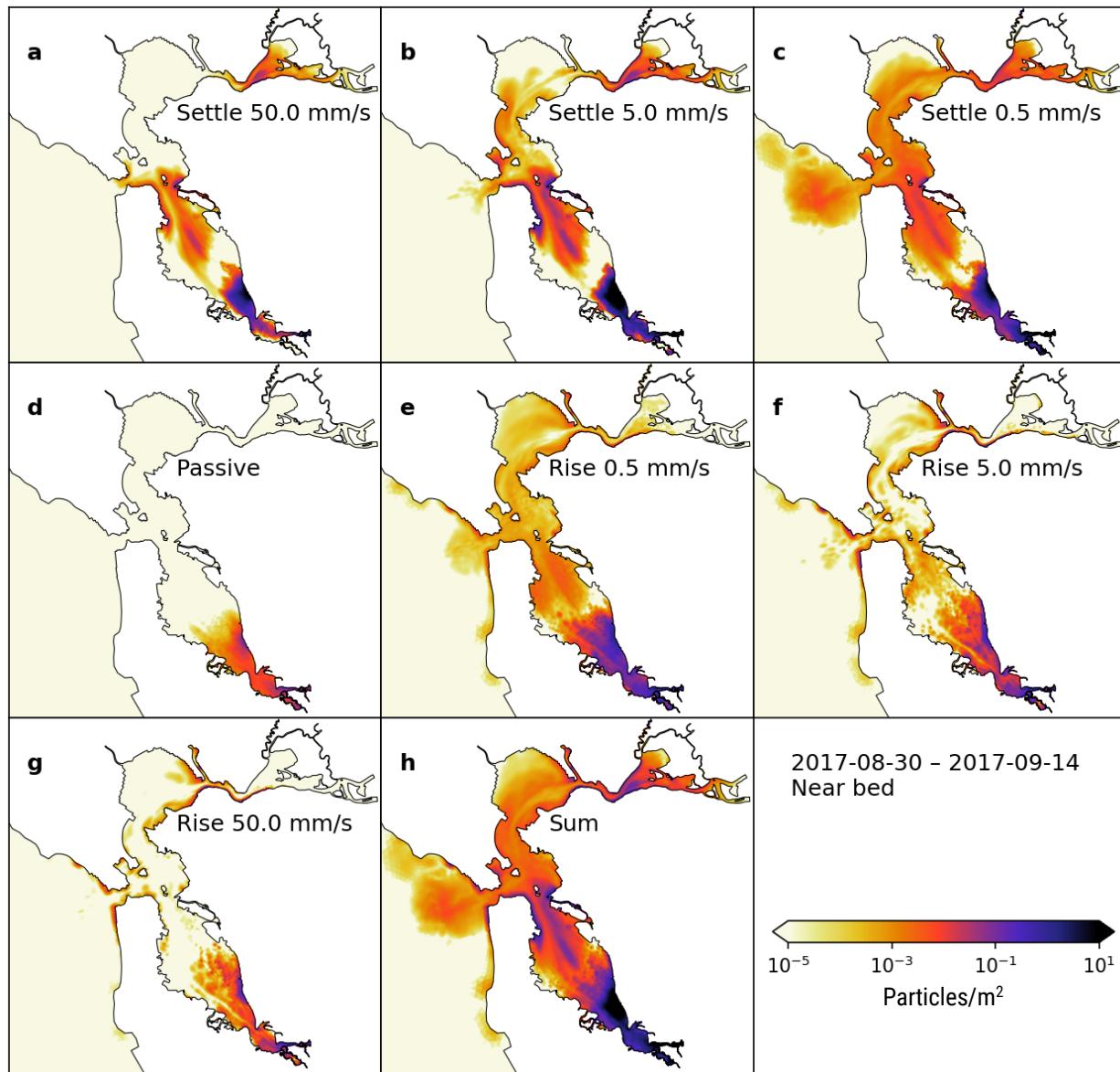


Figure 7.6. Predicted near-bed particle abundance during dry conditions. Only particles up to 15 days old are shown.

Analogous to Figures 7.5 and 7.6, Figures 7.7 and 7.8 show wet-weather particle distributions at the surface and bed, respectively. The clearest differences in near-surface abundance were in San Pablo Bay, where Napa River flows contributed to increased abundance of buoyant particles in the north. The increased Delta outflows pushed particles out of Suisun Bay, leading to an overall decrease in abundance there. Near-bed abundances (Figure 7.8) were larger than their dry-weather counterparts, consistent with denser particles delivered via stormwater. Abundance within the short reach of the Napa River itself was particularly high, likely due to a

model artifact that resulted in trapping particles in this confined portion of the domain (discussed below).

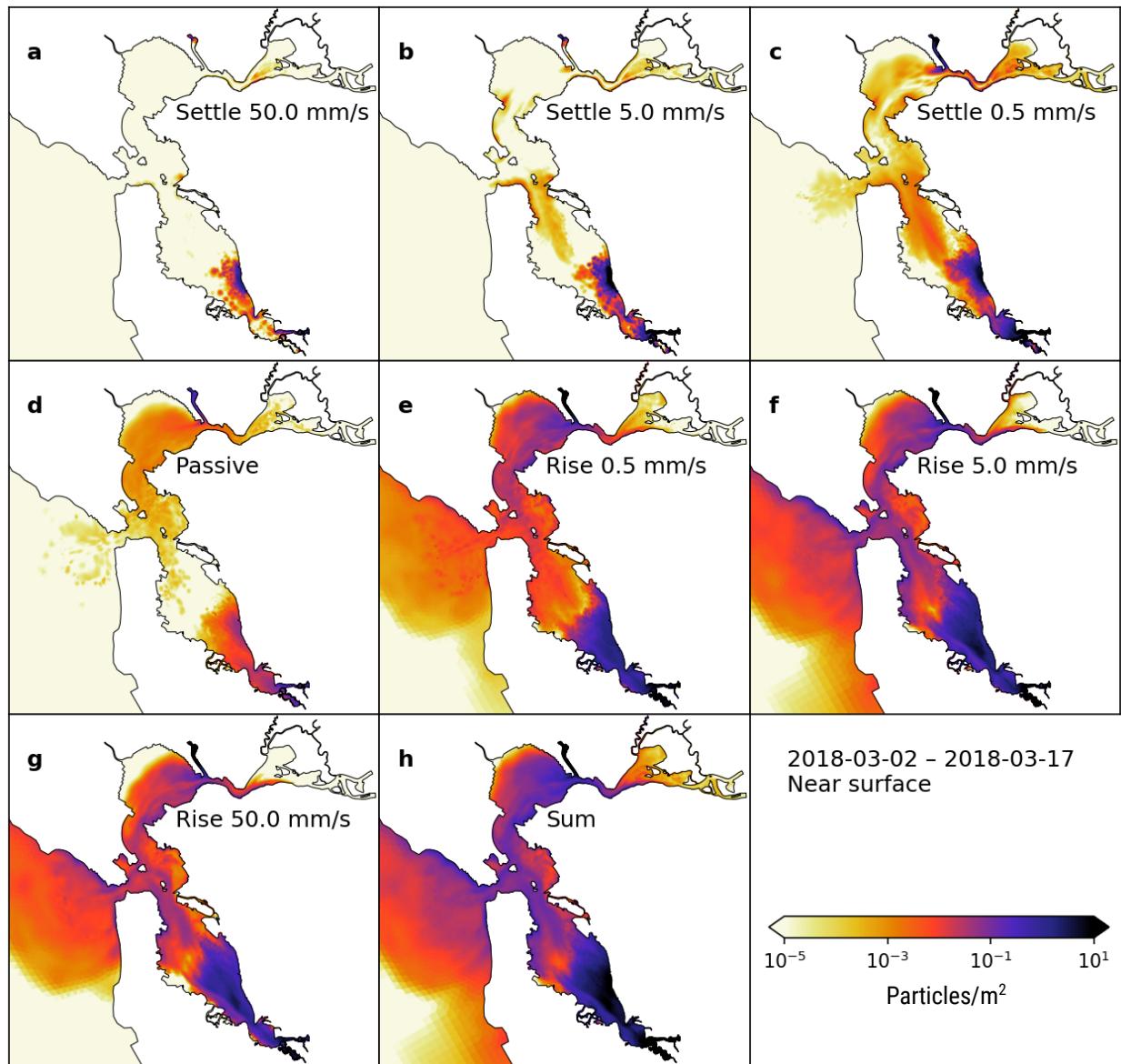


Figure 7.7. Predicted near surface particle abundance under wet-weather conditions. Only particles less than 15 days old are shown.

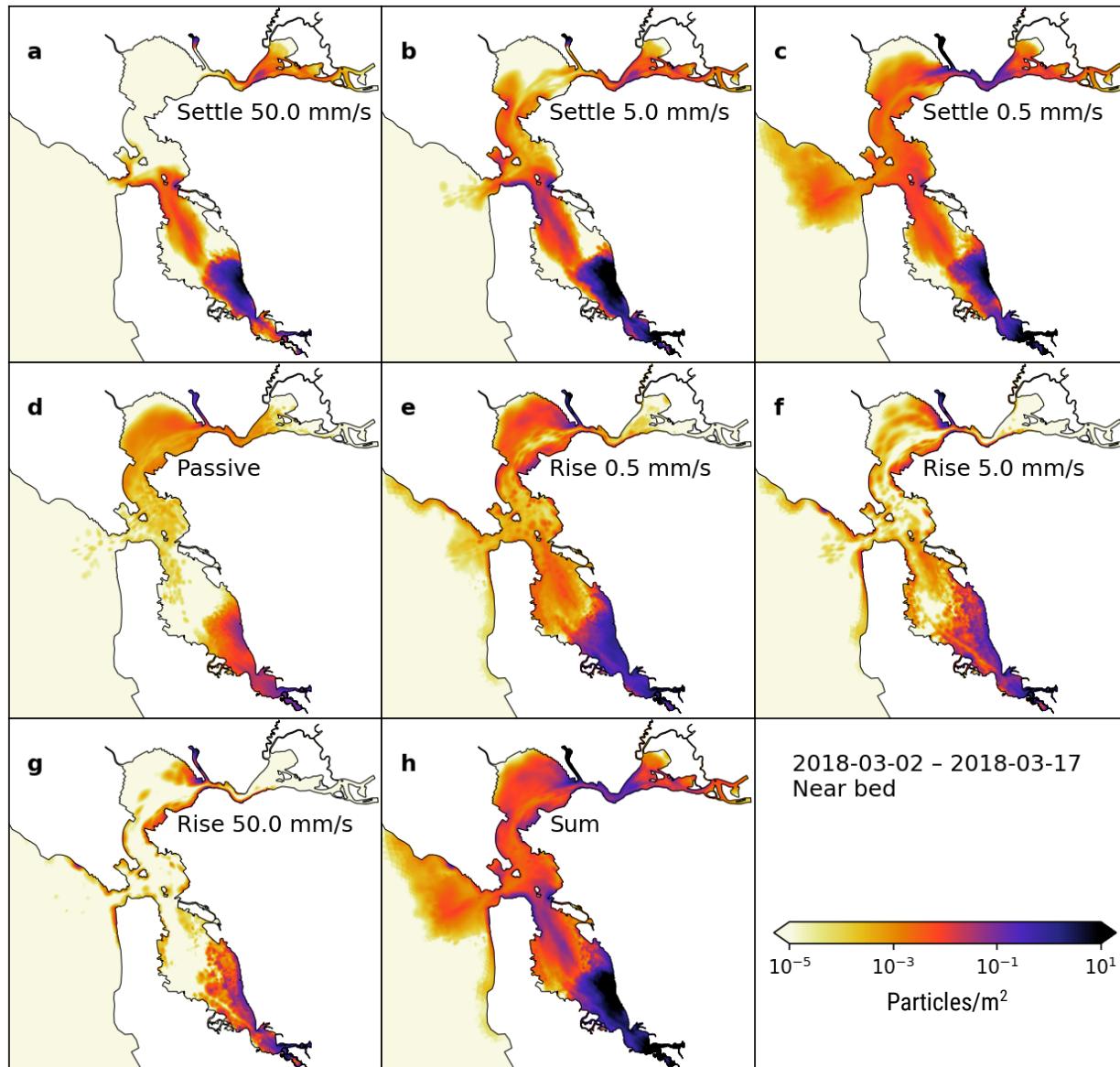


Figure 7.8. Predicted near-bed particle abundance during wet-weather conditions. Only particles less than 15 days old are shown.

DISTRIBUTION IN THE NATIONAL MARINE SANCTUARIES AND COASTAL OCEAN

Figures 7.9–7.12 show a larger-scale view of particle distribution extending to the coastal ocean. As above, dry- and wet-weather conditions are shown, although near-bed abundance is only shown for dry conditions. Very few of the particles that exited the Bay wound up near the bed in the coastal ocean, leading to negligible levels regardless of season.

While the previous figures were limited to model particles with a maximum age of 15 days, Figures 7.9–7.11 include particles up to 44 days old. This extended period is intended to show where particles might go if they avoid microbial colonization, trapping, beaching, or deposition. However, the 44-day period is long enough for many particles to grow a sufficient biofilm to

become negatively buoyant and sink (Chen et al., 2019; Fazey and Ryan, 2016), such that limiting the analysis to a maximum age of 15 days (i.e., Figure 7.12) may be more realistic.

Figure 7.9 clearly shows the degree to which settling particles were retained within the Bay. Likewise, while some passive particles were barely visible beyond the Golden Gate, the release concentrations of passive particles were sufficiently low that abundance in the coastal ocean was still near zero. Meanwhile, buoyant particles were both present in discharges and effectively transported out of the Bay, leading to broad particle distributions for rising particles (Figure 7.9 panels e–g). The corresponding near-bed abundance (Figure 7.10) generally showed no significant accumulation of particles seaward of the Golden Gate. The one minor exception here are the slowest-settling particles (Figure 7.10 panel c). Here a small tongue of exported particles was visible, snaking up the coast around Point Reyes. Tidally driven turbulent mixing inside the Bay moves these slowly settling particles to higher elevations in the water column than the fast-settling particles. With the aid of this turbulent mixing, the particles were able, in small numbers, to be transported out of the Bay. Once out of the Bay and its energetic mixing, these particles sank towards the bed and experienced little further transport.

Moving into wet-weather conditions, Figure 7.11 shows the near-surface particle distribution in the first half of March 2018. The average abundance in the coastal ocean appeared to be higher in the wet season, though the difference was small compared to the spatial heterogeneity present in both figures. The most striking difference was the more organized plumes, one hugging the shoreline to the north, and one meandering to the south. Given the lack of robust model validation in the far-flung ocean regions of the domain, more information is needed before concluding that these patterns were physically realistic or simply a quirk of the transport model.

As a final comparison in the coastal portion in the domain, Figure 7.12 shows particle abundances similar to Figure 7.11, but limited to particles less than 15 days old. Over this shorter window of particle ages, the south-going plume has now “aged out,” and the younger subset of particles was confined to the north-going, shore-bound plume. This suggests that the distribution of particles in the surface of the coastal ocean was driven by Northern Hemisphere river plume dynamics as well as the transient effects of regional winds and the California coastal current (Largier, 2016).

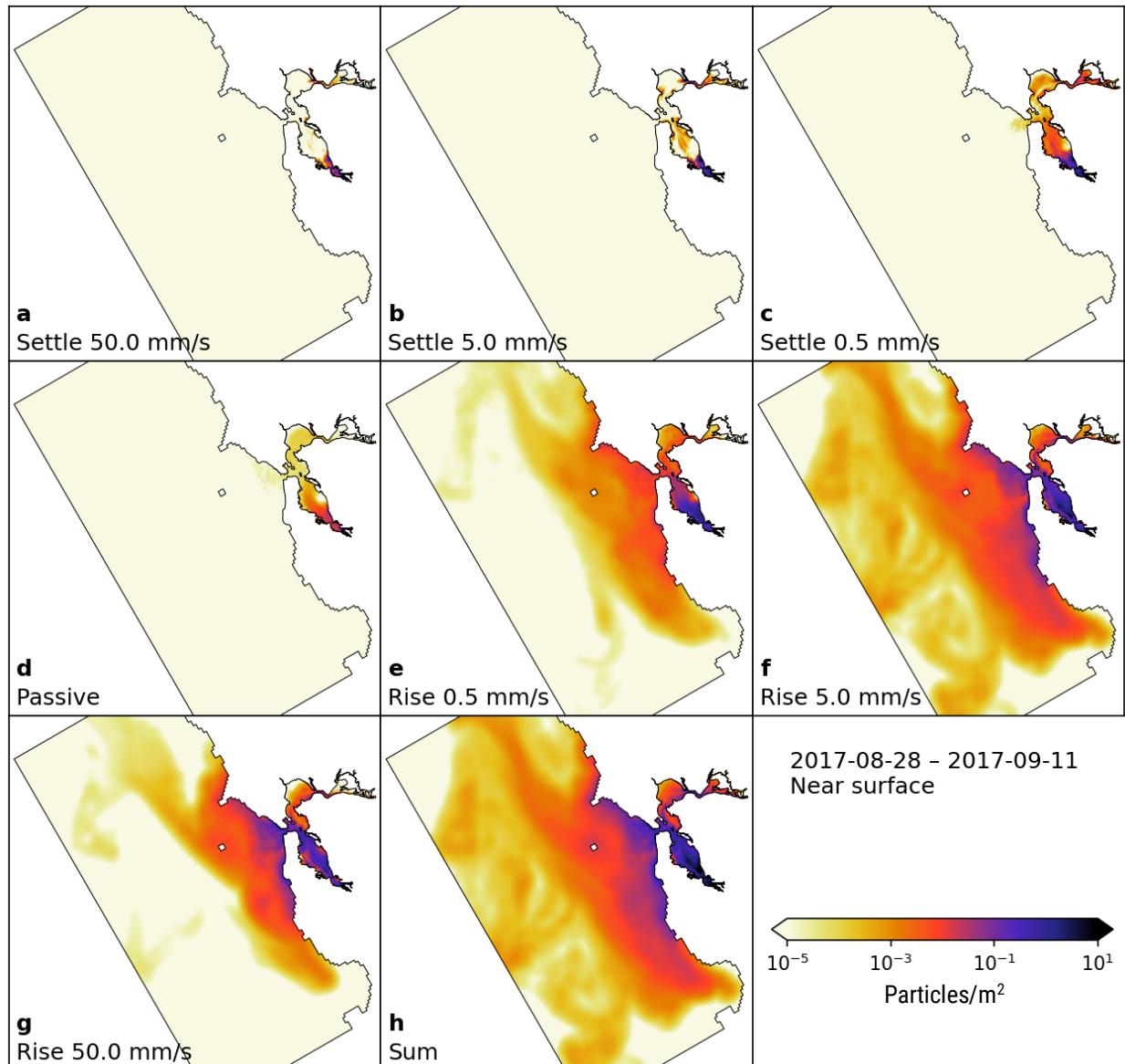


Figure 7.9. Predicted near-surface microparticle abundance under dry conditions. Particles up to 44 days old are included.

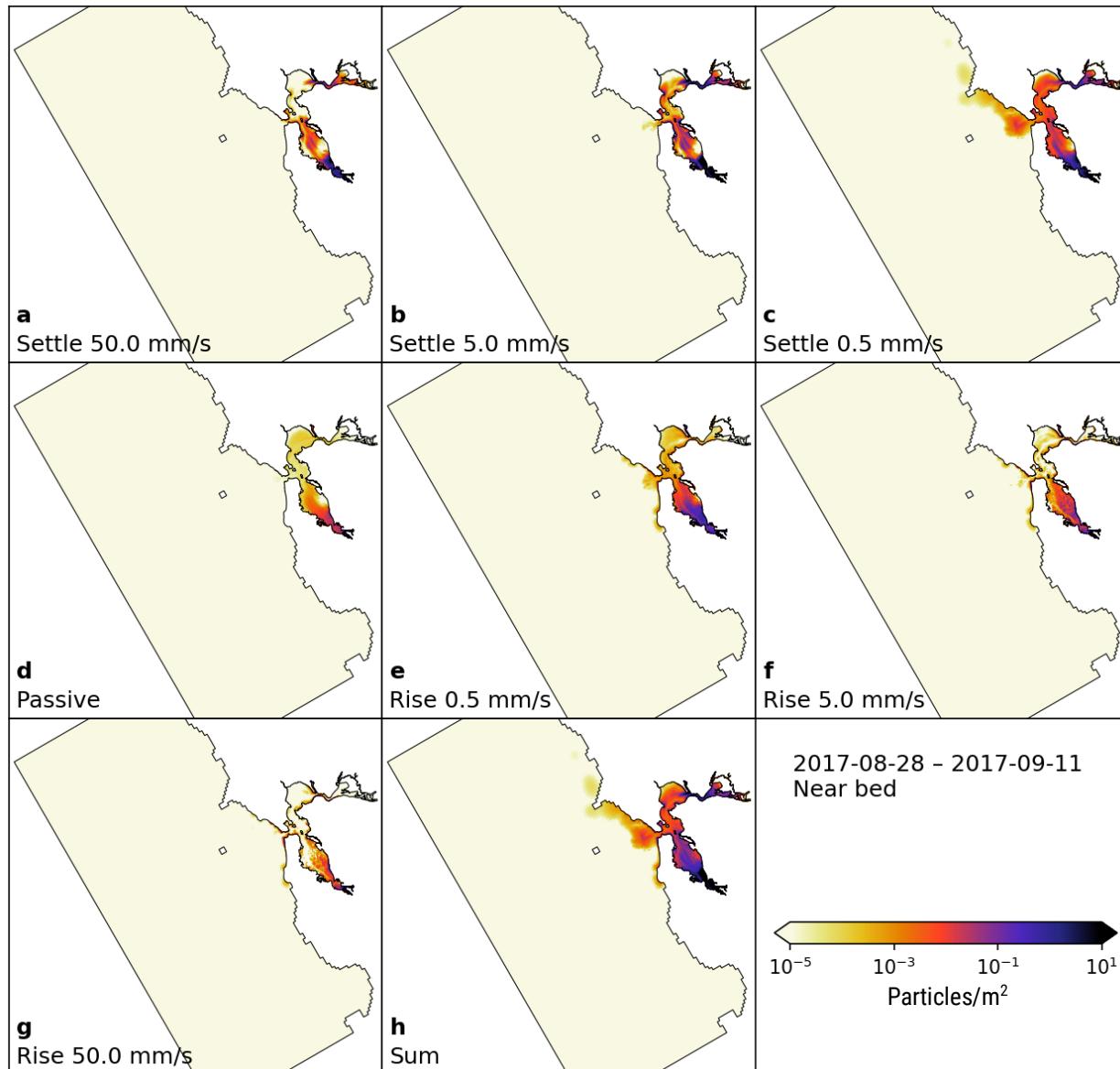


Figure 7.10. Predicted near-bed microparticle abundance under dry conditions. Particles up to 44 days old are included.

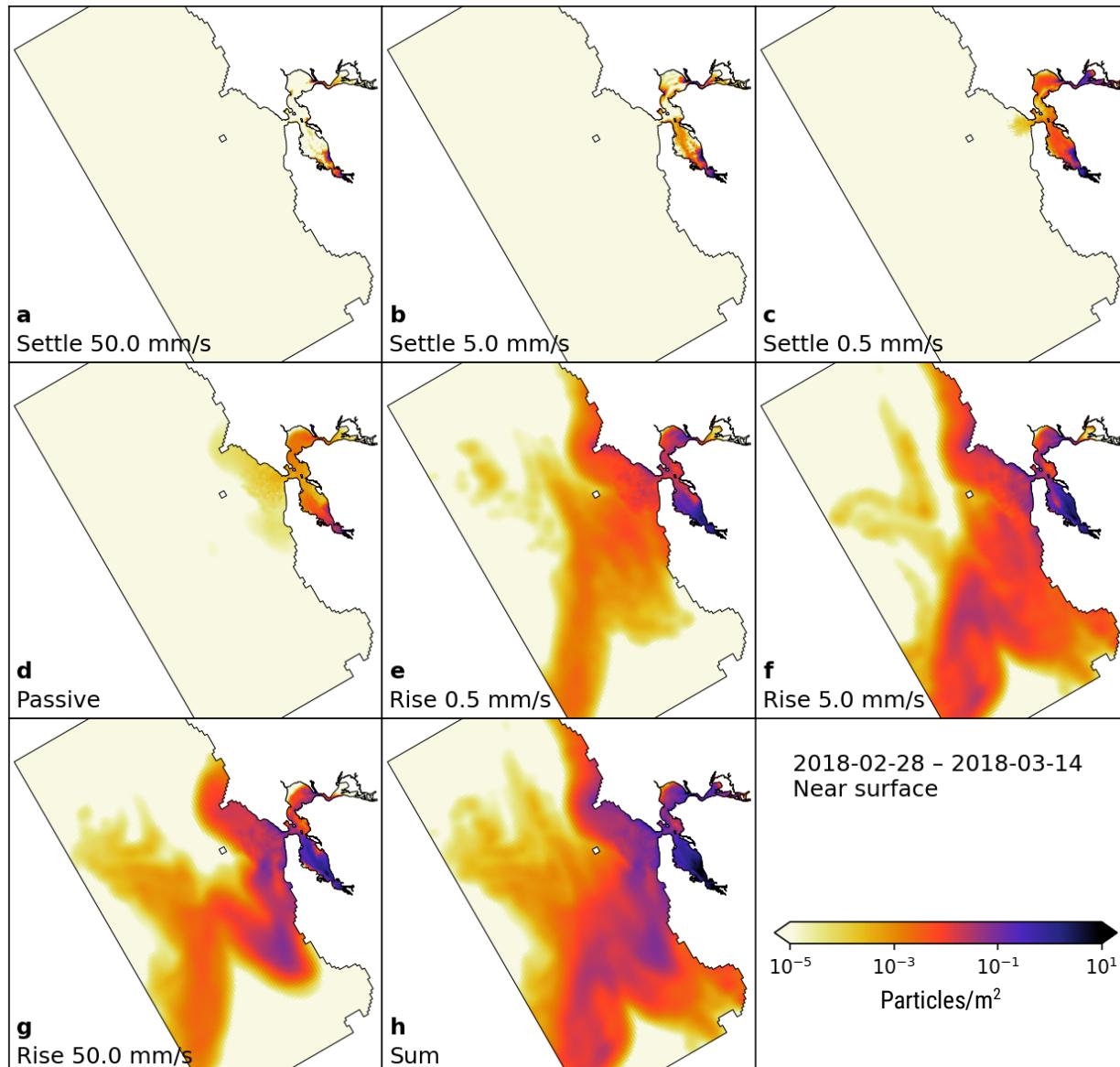


Figure 7.11. Predicted near-surface microparticle abundance under wet conditions. Particles up to 44 days old are included.

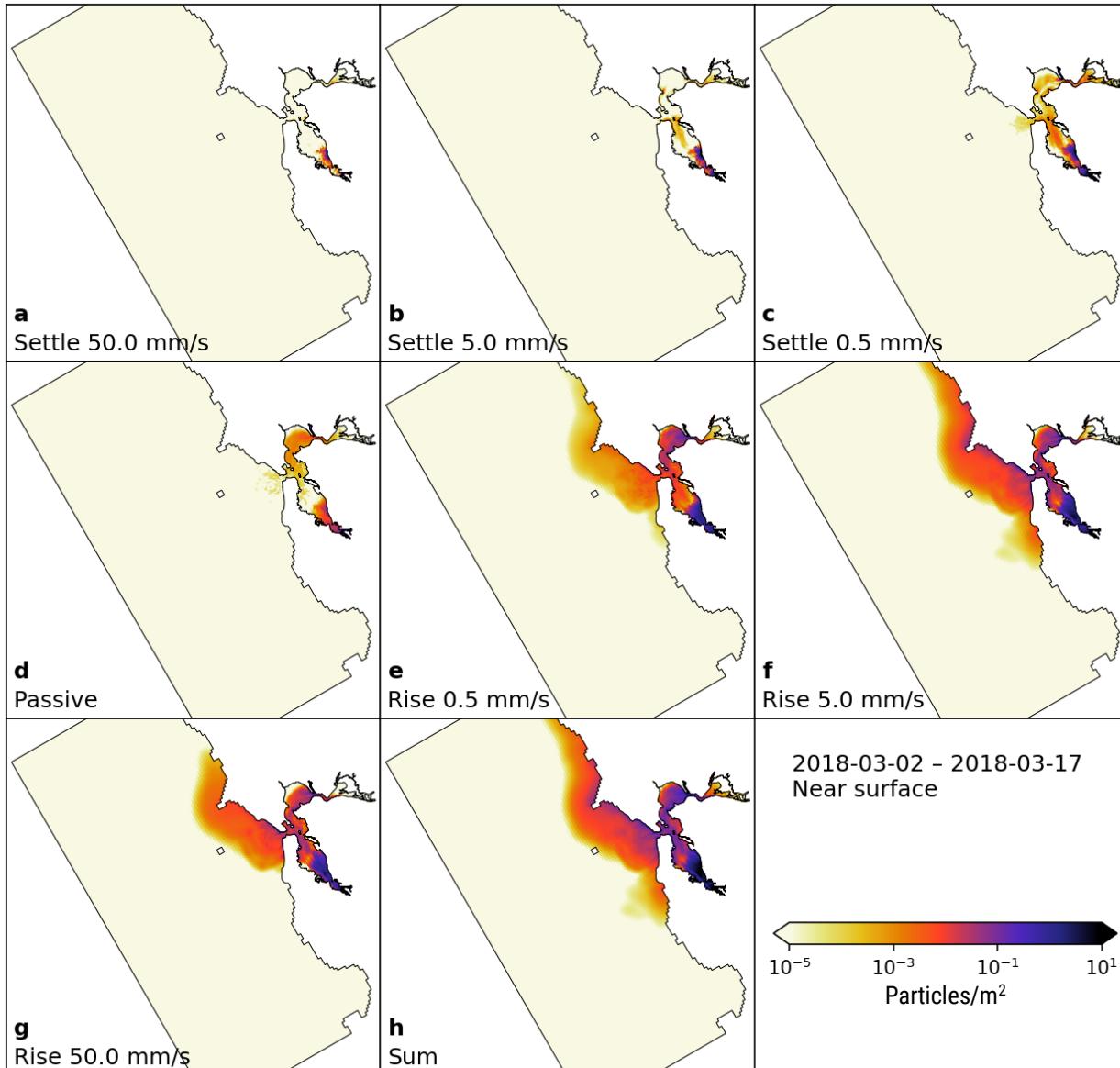


Figure 7.12. Predicted near-surface microparticle abundance under wet conditions. Particles up to 15 days old are included.

Comparison to manta trawls

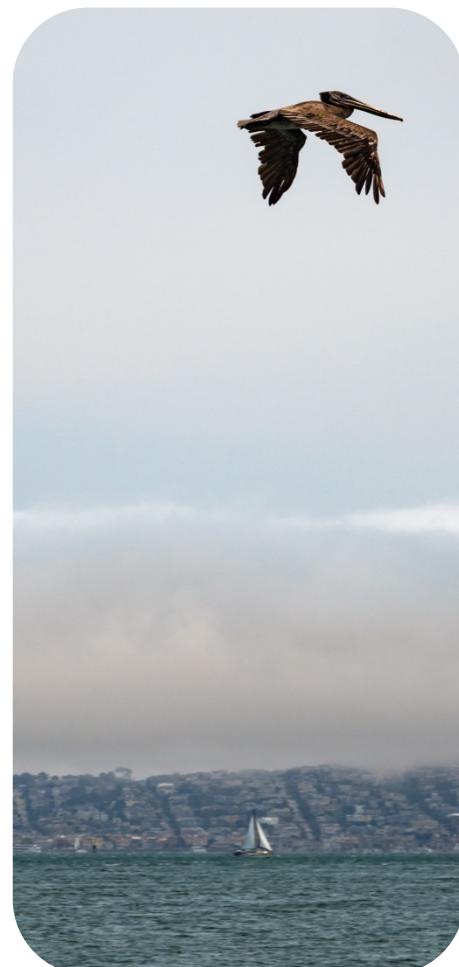
Manta trawl surface water data were aggregated into total particle counts per unit area for each trawl. As in the stormwater and effluent data, particle counts for each category were adjusted by the ratio of the average blank count to the sample count. When field duplicates were available, they were averaged into a single particle density estimate (after blank adjustment for each sample). As described in Chapter 4 Surface Water, trawl samples were limited to size classes greater than 355 µm, while the model predictions are based on stormwater and wastewater samples that include size classes down to 125 µm. In order to keep the model as informative as

Chapter 7—Transport Model

possible, all size classes from the stormwater and wastewater data have been retained, although this introduces a bias in the model–data comparisons presented below.

Figure 7.13 shows predicted surface abundance in dry weather conditions with the trawl data overlaid. Both trawl and model data in Figure 7.13 include all particle categories, and the figure omits trawl samples for which fibers were not analyzed. Both the model and the trawl data were, on average, greater inside the Bay than in the coastal ocean, though the difference was more pronounced in the model. The presence of a coastal plume was not readily apparent in the trawl data. Abundance near San Bruno Shoal (the boundary between Central Bay and South Bay) was less than abundances either north or south of the Shoal in both the trawl data and the model. The greatest predicted abundance values occurred in South Bay, on the eastern shoal. Here the nearest trawl sample appeared to be more reflective of the channel water than the shoal water. San Pablo Bay had two trawl samples, and while both samples had higher abundances of microparticles than in the model, the gradient between the channel and the shoal was reasonably predicted. The lack of 125 μm particles in the trawl data was expected to lead to model results being biased high relative to the trawl data. However, any such discrepancy was obscured by the broad variability of the data.

One pattern that was perhaps non-intuitive was that both the trawl data and the model showed high abundance in much of Central Bay compared to counts directly to the south or north. Considering the proximity of Central Bay to the coastal ocean, and the lower average abundance in the coastal ocean, one might expect Central Bay to have particle counts somewhere between the in-Bay and coastal numbers. This is commonly observed for dissolved constituents (Largier, 1996; Smith and Hollibaugh, 2006), but does not necessarily hold true for buoyant particles. As the water depth increases, the ratio of surface area to water volume decreases, such that the same particle concentration (depth-averaged, as particles per unit volume) will translate to a greater surface abundance. In other words, particles transported at or near the surface become more concentrated (per unit area) in deep regions because there is less surface area for the particles to occupy. Regional winds may also be responsible for the accumulation of particles near Potato Patch Shoal and the Golden Gate.



Chapter 7—Transport Model

Figure 7.13 includes only sites where all particle categories, including fibers, were counted. Figure 7.14 shows a similar comparison but with fibers omitted from both the model loads and surface water samples (and including the samples omitted from Figure 7.13). The larger number of samples mostly exhibit similar patterns. The additional sample in Grizzly Bay (the northwestern lobe of Suisun Bay) had low abundance in both the model and data, suggesting that the stark north–south gradient here could be realistic. The coastal samples roughly confirmed the predicted north-going plume during this period, though the trawl samples indicated this plume reached farther offshore than the model predictions. Several notable discrepancies between the model and data are worth mentioning. A high count sample in San Leandro Bay was not captured by the model, potentially due to poorly resolved hydrodynamics relative to the size of the sample area. The southeastern-most sample in Lower South Bay had lower abundance than the neighboring trawl samples, while the model predicted increasing abundance as one moves south towards several discharges. Finally, a trawl sample in the Golden Gate had one of the lowest counts of all samples, yet the model indicated relatively high abundance throughout this region. Overall, there were many places where the model captured important gradients.

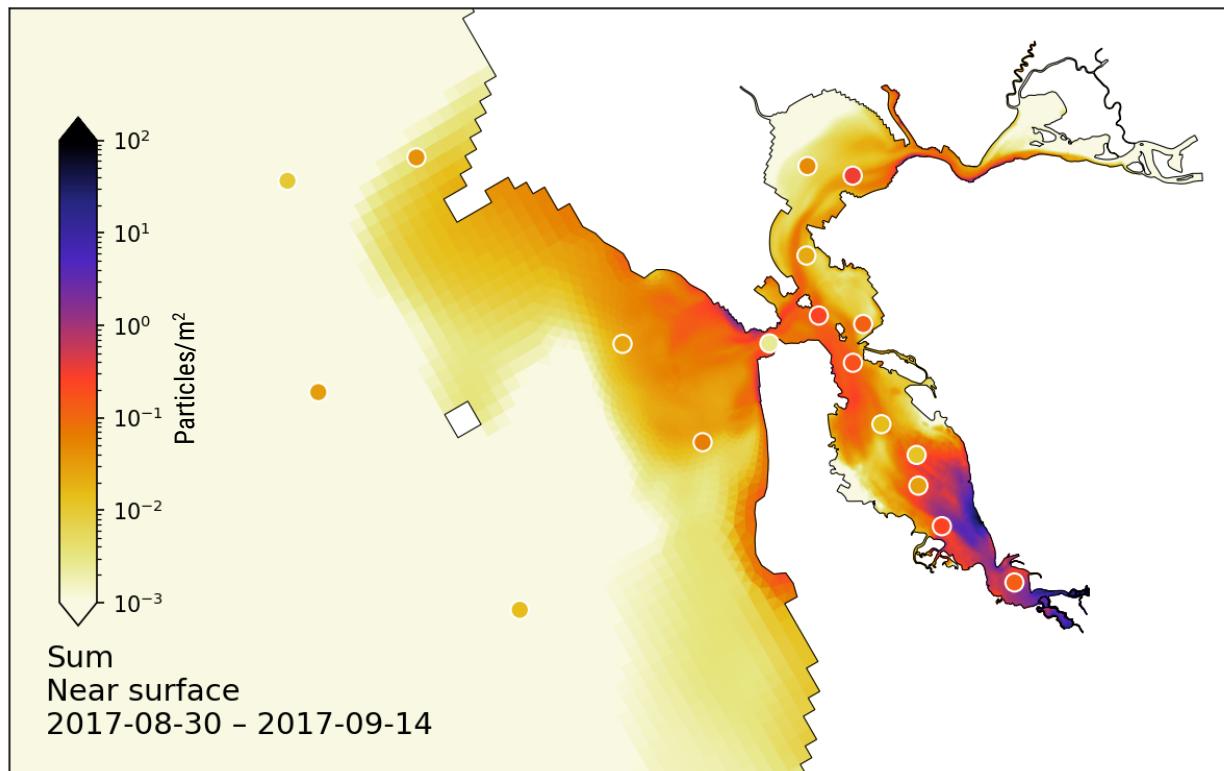


Figure 7.13. Total particle abundance (fiber and non-fiber) from manta trawls compared to predicted surface abundance for dry conditions. Only particles up to 15 days old are included in the model. Manta trawl samples for which fibers were not counted were omitted (See Chapter 4 Surface Water). Note that the color scale range differs from previous plots.

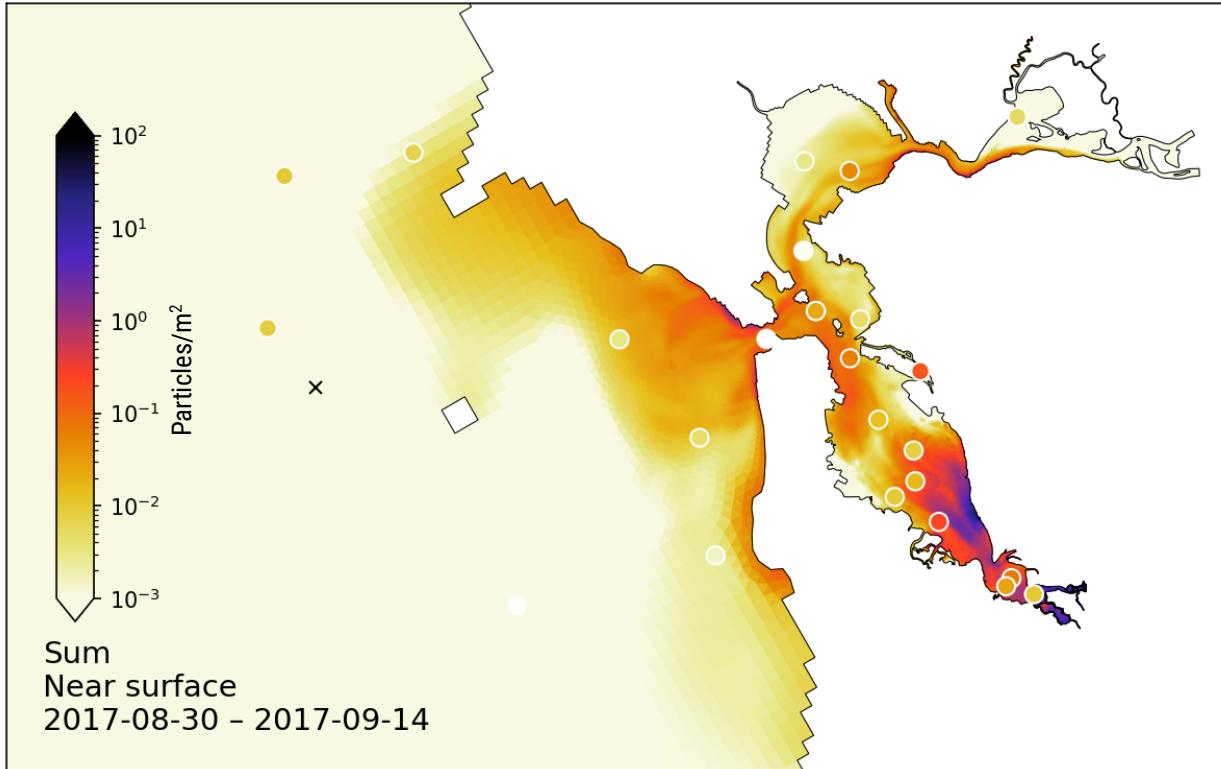


Figure 7.14. Non-fiber particle abundance from manta trawls compared to predicted non-fiber surface abundance for dry conditions. Manta trawl samples for which particle counts were below the mean blank counts across all categories are shown as an \times . Only particles up to 15 days old were included in the model.

A trawl–model comparison for wet weather conditions is shown in Figure 7.15, again only for samples in which fibers were among the quantified categories. Both datasets showed greater abundance in wet weather than dry weather, particularly in South Bay. Within the limited set of samples for which fibers were counted, the predictions were generally in line with observations, with the exception of one coastal sample that was severely under-predicted. Without more samples, it is difficult to discern whether this was due to the freshwater plume extending farther offshore, the sample reflecting a localized convergence, or if the coastal predictions have more fundamental errors.

Figure 7.16 shows the same time period as Figure 7.15, but now with fibers omitted from the trawl samples as well as the model predictions. As in previous figures, abundance within the Bay is greater than abundance in the coastal ocean in both the predictions and trawl samples. Predicted abundance in Lower South Bay was much greater than observed. This discrepancy may arise from the poor distribution of stormwater inputs in the hydrodynamic model. While realistic stormwater discharges were distributed around the Bay margins, the modeled discharges were concentrated at just four locations, two of which were in Lower South Bay. Another notable outlier was the high-abundance sample near the Central Bay–South Bay boundary. This specific trawl happened to sample across a frontal feature (see Chapter 4 Surface Water), and while the model

Chapter 7—Transport Model

was able to capture some frontal features (such as the linear features in Figure 7.19), this level of small-scale heterogeneity was averaged away in the model data analysis and was unlikely to be captured precisely in the first place.

In the coastal ocean, the observations were relatively consistent, with one exception of a low-abundance sample just south of the Golden Gate. In comparison, the modeled values were heterogeneous, with a predicted plume overlapping three of the eight coastal sample sites. Within the plume, levels were comparable between the model and data, while the predicted values were lower outside the plume. This divergence may be due to the hydrodynamic model not adequately capturing the location of the freshwater plume, for example missing an offshore wind event that temporarily pushed the plume westward. The fact that the observations showed greater abundance farther from shore also suggests that these surface-bound particles could be persisting for longer than 15 days, and the 15-day cutoff of Figure 7.16 is too short.

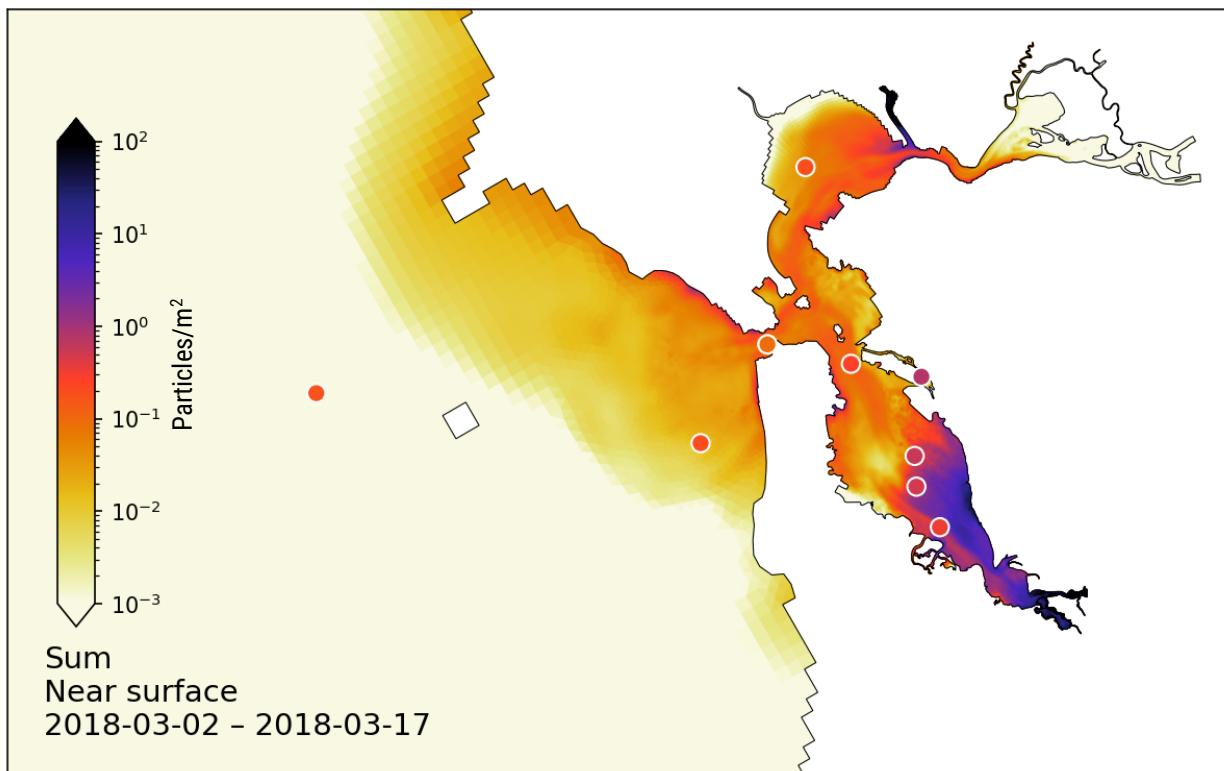


Figure 7.15. Particle counts (fiber and non-fiber) from manta trawls compared to predicted near-surface abundance for wet conditions. The mean particle count from field and laboratory blanks was subtracted from field samples on a per-category basis. Field duplicates were combined. Only particles up to 15 days old were included in the model.

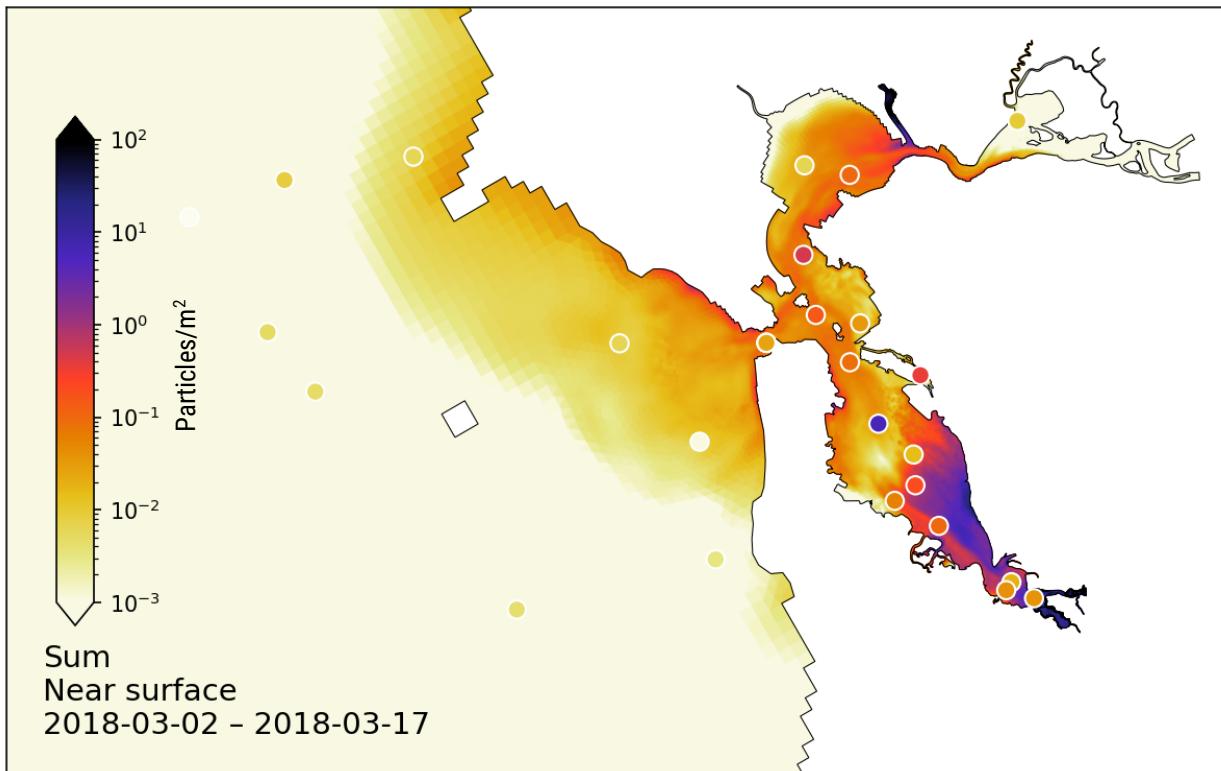


Figure 7.16. Non-fiber particle counts from manta trawls compared to predicted near-surface abundance of non-fiber particles for wet conditions. The mean particle count from field and laboratory blanks was subtracted from field samples on a per-category basis. Only particles up to 15 days old were included in the model.

Comparison to sediment data

The near-bed outputs extracted from the model were intended to provide a qualitative basis for comparison to microparticle counts in sediment samples. Because sediment and microparticle accumulation integrate over potentially long periods, the model output was averaged over the full period of the model run, approximately one year. For the purposes of general comparison, the sediment data were aggregated within each sub-embayment and are reported as a single concentration in units of microparticles/g dry weight (dw) of sediment. Sediment sample concentrations were adjusted to account for the average counts in the blanks following the same approach as above. For each category, blank counts were averaged over the field blank and laboratory blanks. Particle counts in each field sample were then scaled by the ratio $(\text{sample} - \text{blank}) / \text{sample}$.

Figures 7.17 and 7.18 show the average predicted near-bed microparticle abundance along with measured sediment sample data. Figure 7.17 limits the model output to simulated particles up to 15 days old, while Figure 7.18 includes particles up to 44 days old. In both cases, model output is reported as particles per area and is integrated over the lowest 0.5 m of the water column. There was good agreement between the field data and predicted distribution in placing the highest levels in Lower South Bay. Overall, the model-data agreement was significantly better with the 15-

day cutoff than with the 44-day cutoff, suggesting that particles transported near or along the bed were deposited on time scales closer to 15 days than 44 days. The predicted distributions showed significant gradients near shorelines and in close proximity to stormwater sources, in particular Napa River and Alameda Flood Control Channel.

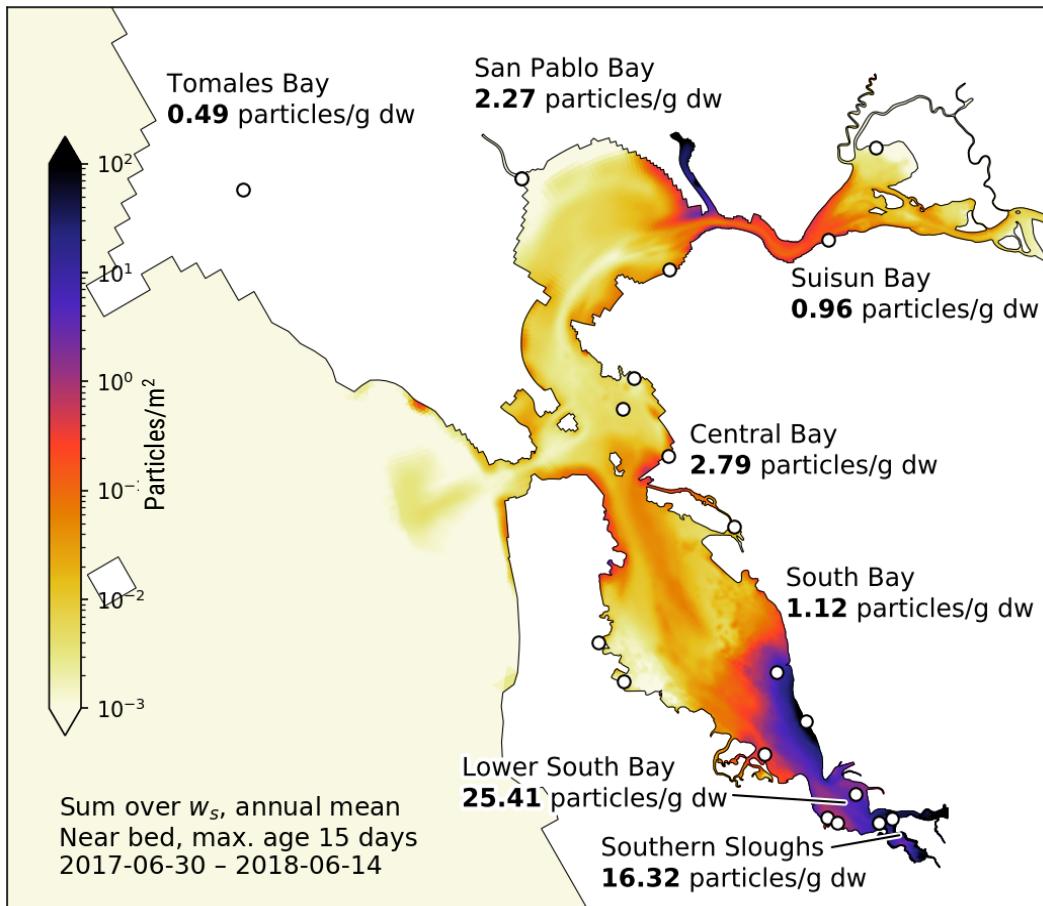


Figure 7.17. Predicted average near-bed microparticle abundance (color shading), truncated at 15 days since release, and summarized sediment data (text values). Observed sediment concentrations are particle counts per unit mass, while predictions are counts per unit area. The locations of individual sediment samples are shown as empty circles.

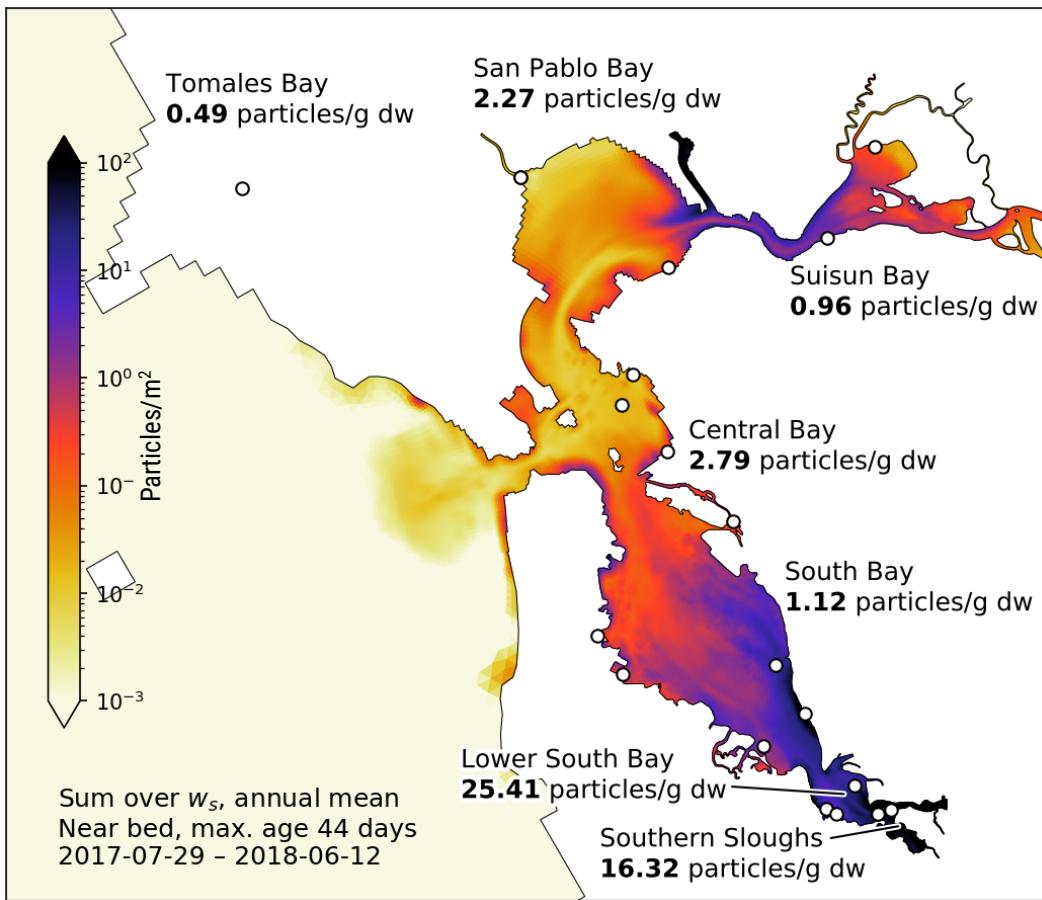


Figure 7.18. Predicted average near-bed microparticle abundance (color shading), truncated at 44 days since release, and summarized sediment data (text values). Sediment values represent particle count per unit mass. The mean of laboratory and field blanks was subtracted from sample concentrations on a per-category basis. The locations of individual sediment samples are shown as empty circles.

Time for export

A key question about transport of microparticles in San Francisco Bay and the coastal ocean is whether particles are retained in the Bay or exported to the coastal ocean, and how quickly this transport occurs. A series of particle releases were simulated under the conditions of July 2017 to explore the time scales of export as a function of rising/settling velocity. For this analysis, a single release location was used, roughly in the center of South San Francisco Bay, and a wider selection of rising/settling velocities were used compared to the per-discharge simulations discussed above. Figure 7.19 shows the distribution of particles at the end of the 30-day simulation, with particles colored by rising/settling velocity.

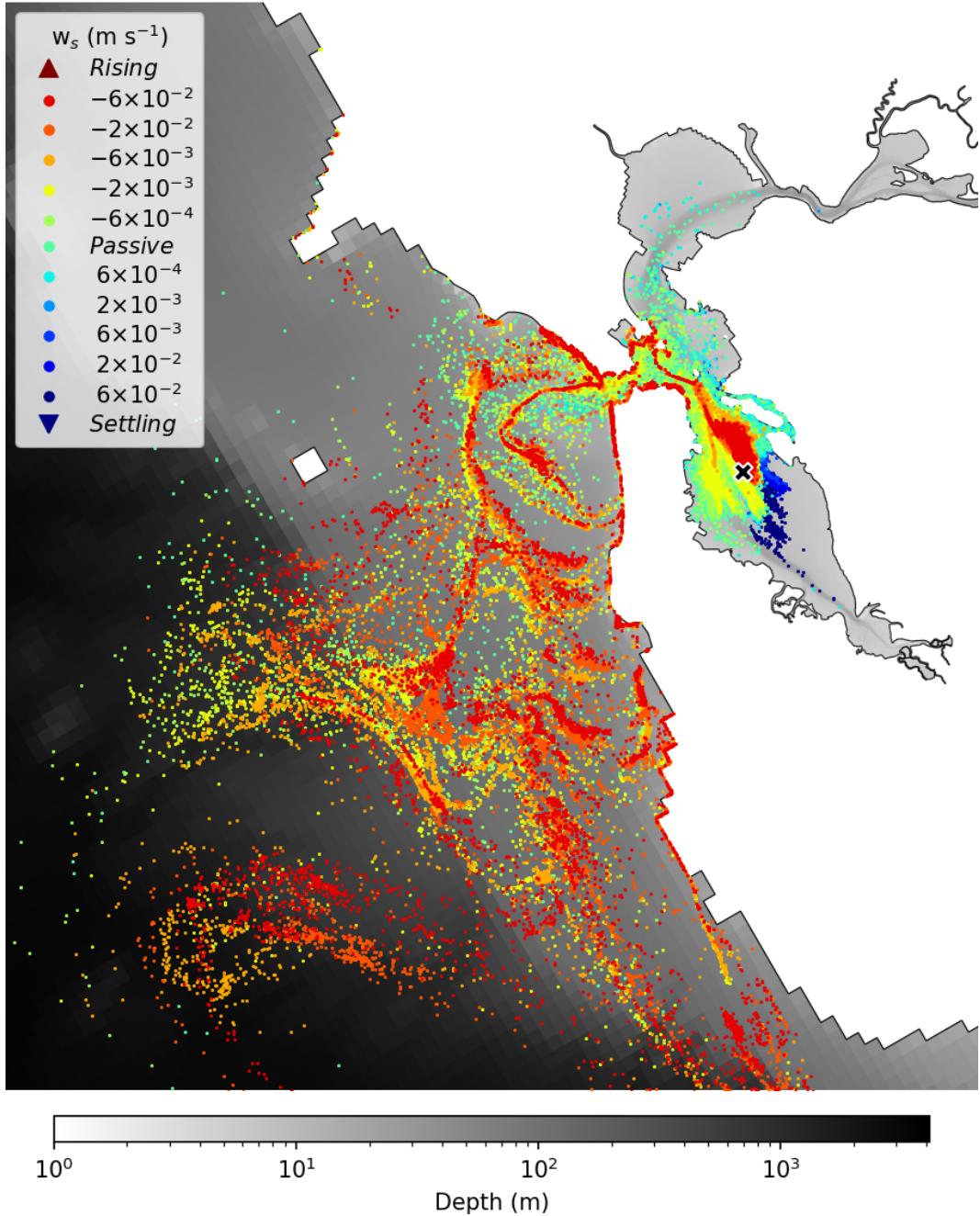


Figure 7.19. Particle distribution after 30 days, South Bay release at **X**. w_s represents a rising velocity when negative and a settling velocity when positive.

Analyzing each rising or settling velocity separately, we compared the elapsed time since particles were released into South Bay with the fraction of particles that left the Bay via the Golden Gate. Figure 7.20 shows the relationship between exit and elapsed time for eleven different rising/settling velocities. Note that essentially all settling particles were retained in the Bay, and only 20% of passive particles made the journey from South Bay through the Golden Gate, whereas 40–100% of buoyant particles exited the Bay within 30 days.

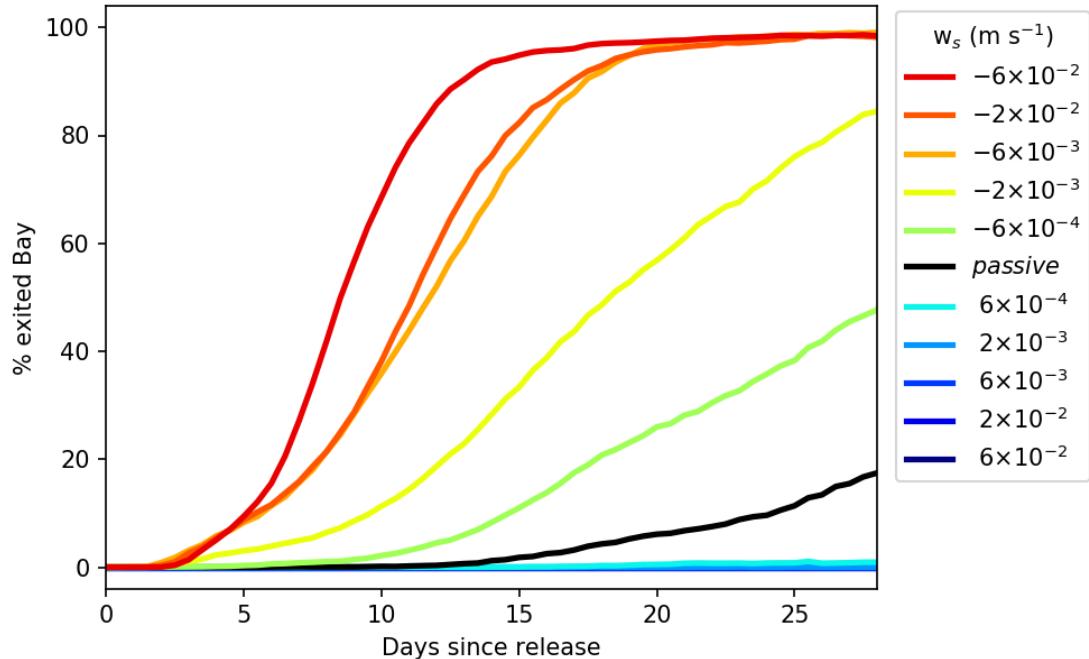


Figure 7.20. Transport time from South Bay through the Golden Gate. w_s represents a rising velocity when negative and a settling velocity when positive.

Discussion

Overall the model brackets the range of observed concentrations and captures broad patterns between Bay and coastal abundances. Even with the model output averaged over 15 days and spatially smoothed, many spatial gradients persist and suggest that comprehensive spatial sampling would require a large number of sampling sites.

Strengths and weaknesses of the model

The coupled hydrodynamic and particle tracking model proved to be a powerful and flexible platform for the analysis of buoyant and settling particles across estuarine and coastal spatial scales. Interactions between the buoyancy of modeled particles and the three-dimensional circulation in the system were well represented, and the distinct fates of different particle types demonstrated the importance of these interactions. The unstructured grid allowed seamless transport between the Bay and the coastal ocean, avoiding any artifacts of a transition between two models or two distinct grids within the same model. While the circulation patterns in the coastal portion of the domain have not been validated, coupling with large scale currents (HYCOM), tides (OTPS), and winds (COAMPS) generated realistic circulation patterns, qualitatively similar to patterns observed in satellite remote sensing (e.g., Largier, 2016).

The hydrodynamic model is calibrated to tides and salinity within the Bay, but several potentially important aspects of the model have not been calibrated. A parameterization of wind drag on surface currents is included in the hydrodynamic model (per Large and Pond, 1981), but this parameterization has not been specifically calibrated, and there is no direct parameterization of surface waves. In addition, given the importance of stormwater loads in the overall microparticle load to the Bay, the hydrodynamic model does not incorporate enough distinct stormwater discharge sites. While a simple scaling was applied in order to reach a reasonable total number of particles entering the system via stormwater, the spatial and temporal distribution of the loads were not accurate due to the limited number of discharges. With additional hydrologic modeling, ungauged watersheds could be included to better distribute stormwater flows.

While the use of a particle-tracking model allows for flexible post-processing of the model output, it also comes with some limitations. A direct tradeoff for this flexibility is that concentration and abundance estimates derived from particle distributions require some degree of smoothing in time and/or space. This limits the meaningful model outputs to averaging periods of days to weeks and a spatial resolution roughly an order of magnitude coarser than the hydrodynamic model grid. Applying a particle tracking model to the output of a z-layer hydrodynamic model also brings the possibility of particles becoming stuck on the bed and accumulating at discontinuities in the bed layers. There is also the possibility of particles getting stuck in particularly small, confined portions of the grid where local velocity gradients are barely resolved and may include

dead zones. This likely leads to many particles accumulating in the confined irregular channels around the Bay. These same reaches may accumulate microparticles in reality, but until there are better observational data at the scale of creeks and sloughs, including beaching efficiency across variable shoreline types, it will be difficult to assess how much of an issue this is in the model.

The difference in size classes in the load data (greater than 125 µm) and trawl data (greater than 355 µm) was expected to introduce bias into the comparisons. Any such bias, however, was obscured by the large variability in predictions and observations, spanning one to two orders of magnitude.

Insights on accumulation zones

The model results suggest several accumulation zones and related mechanisms.

ESTUARINE CONVERGENCE

A common physical phenomenon in estuaries is a convergence of near-bed currents near the transition point between brackish water and freshwater. Landward of this point, river flows carry material downstream, while seaward of this point, intruding, dense saltwater carries material landward. Settling particles tends to accumulate where these two currents meet. These convergence zones are similar in principle to the convergence zones driven by Langmuir circulation discussed in Chapter 4 Surface Water, but differ in location (estuarine convergence is at the bed, while Langmuir cells are at the surface) and scale (estuarine convergence is often a zone spanning kilometers, while Langmuir cells are typically meters wide and hundreds of meters long).

More commonly discussed in the context of sediment transport and termed *estuarine turbidity maxima*, there are two potential regions in the Bay where near-bed currents converge and are likely to accumulate settling particles. In San Francisco Bay, the more distinct convergence region is in North Bay, falling in Carquinez Strait and the western portion of Suisun Bay during wet conditions (MacWilliams et al., 2015). This point is often codified as "X2," denoting the location where the average near-bed salinity is 2 ppt. While in theory a similar region of convergence would exist in Lower South Bay, the much smaller freshwater flows in Lower South Bay allow saltwater to intrude deeper into the slough and channel network, where local sources and circulation patterns obscure any local convergence.

SOUTH BAY SHOALS

Driven by Alameda Flood Control Channel discharge, prevailing winds, and the long residence time of the shoals, predicted particle abundances were consistently higher here than to the north or in the adjacent channel. While the model does have some bias here



due to the under-distributed nature of the stormwater inputs, hydrodynamically it is still expected that particle concentrations would be elevated relative to the overall mean.

COASTAL PLUME

In the absence of winds, the Coriolis effect dictates that a buoyant river plume in the Northern Hemisphere turns to the right. This effect is most clearly seen in Figure 7.12. This right-turning plume behavior is sometimes upset by transient wind events (visible in Figure 7.11, where a wider variety of ocean conditions have been included in the longer averaging window) and summer-time upwelling winds (such as in Figure 7.9), which tend to push the plume downwind and offshore. These wind-driven behaviors, a product of passing weather systems, are naturally more chaotic than the north-traveling plume.

Loads and fate

While both data and predictions span orders of magnitude, the general overlap of the respective ranges suggest that major loads or sinks were not absent from the model. Bay–ocean gradients in the trawl data, placing higher concentrations within the Bay, suggest that the coastal ocean is not a significant source of buoyant particles to the Bay. However, the model over-predicted this gradient, and specifically under-predicted the spatial extent of buoyant particles in the coastal ocean. This leaves open the possibility that San Francisco Bay does not represent the only supply of buoyant particles to this region of the coastal ocean.

Predictions of near-bed abundance and the time scales of particle export (Figure 7.20) show that settling particles are effectively retained in the Bay. These results are consistent with the conceptual model of estuarine circulation, in which freshwater exits the Bay near the water surface (exporting buoyant particles), while saltwater enters the Bay near the bed (potentially importing settling particles), and suggest that settling particles are more likely to be imported into the Bay than buoyant particles. Furthermore, the transport range of settling particles was generally short relative to that of buoyant particles, and suggests that long-range transport of settling particles is unlikely.

Conclusions

To our knowledge, this is the first three-dimensional, microplastic transport model to span estuarine and coastal scales. The model successfully reproduced broad spatial patterns observed in manta trawl surface water and sediment samples, and demonstrated the potential for physical processes to lead to variations in particle abundance at finer scales. A particle tracking approach to the modeling allowed flexible analysis of model outputs after the fact. This avoided the need to have all field data in hand before modeling could commence, and enabled multiple analyses (e.g., with and without fibers) with a single simulation. Even though the particle tracking approach required some degree of averaging in time and space, spatial gradients were still evident in much of the model domain for both wet and dry seasons. While it remains an open challenge to predict microparticle abundance across the spatial scales and heterogeneity of San Francisco Bay, the present modeling provides a solid foundation for designing future monitoring, interpreting field data, and refining future mechanistic modeling efforts.

The degree of spatial heterogeneity in the model outputs led to a valuable observation that microparticles were not evenly distributed in surface waters, even after the homogenizing effects of turbulent mixing and tidal stirring. This persistent heterogeneity was a direct consequence of the buoyant or sinking nature of each particle. The distributions of new (less than 15 day old) buoyant and passive particles reflected gradients in the relative concentrations from different source waters, with wind-driven accumulation in some leeward shoals. Sinking particles primarily accumulated near their entry points, and were confined broadly within the Bay by the landward component of freshwater/saltwater exchange flows.

Likewise, the fate of microparticles, in terms of retention or export from an estuarine setting like San Francisco Bay, was largely dependent on their buoyancy. This dependence makes laboratory analysis, whether by density separation or by spectroscopic identification, essential in understanding the fate of microparticles entering estuarine waters.

From both the model results and field data, we conclude that microparticles originating in San Francisco Bay do, on occasion, reach the majority of the nearby National Marine Sanctuaries. However, only buoyant particles are likely to travel any significant distance beyond the Golden Gate. Buoyant particles are efficiently transported in the freshwater plume leaving the Bay, often taking them northward along the coast, or dispersing them south and west by regional winds.

References

- Chen, X., Xiong, X., Jiang, X., Shi, H., and Wu, C. 2019. Sinking of floating plastic debris caused by biofilm development in a freshwater lake. *Chemosphere*. 222:856–864.
- Chua, V. and Fringer, O.B., 2011. Sensitivity analysis of three-dimensional salinity simulations in North San Francisco Bay using the unstructured-grid SUNTANS model. *Ocean Modeling*. 39 (3–4) 332–350.
- Fazey, F., and Ryan, P., 2016. Biofouling on buoyant marine plastics: An experimental study into the effect of size on surface longevity. *Environmental Pollution* 210:354–360.
- Fringer, O.B., Gerritsen, M., and Street, R.L. 2006. An unstructured-grid, finite-volume, nonhydrostatic, parallel coastal ocean simulator. *Ocean Modeling*. 15 (3–4), 139–173.
- Holleman, R.C., and Stacey, M.T. 2014. Coupling of sea level rise, tidal amplification and inundation. *Journal of Physical Oceanography*. (44) 1439–1455.
- Holleman, R.C., Nuss, E., and Senn, D. 2017. San Francisco Bay Interim Model Validation Report. SFEI Contribution No. 850. San Francisco Estuary Institute, Richmond, CA.
- Ketefian, GS, Gross, ES., Stelling, GS., 2016. Accurate and consistent particle tracking on unstructured grids. *International Journal for Numerical Methods in Fluids*, 80(11), 648–665.
- King, A., 2019. Wind over San Francisco Bay and the Sacramento-San Joaquin River Delta: Forcing for Hydrodynamic Models. SFEI Contribution No. 937. San Francisco Estuary Institute, Richmond, CA.
- Large, W.G. and Pond, S., 1981. Open Ocean Momentum Flux Measurements in Moderate to Strong Winds. *Journal of Physical Oceanography*. (11), 324–336.
- Largier, J., 1996. Hydrodynamic exchange between San Francisco Bay and the ocean: the role of ocean circulation and stratification. In: *San Francisco Bay: The Ecosystem*. J.T. Hollibaugh (editor). Pacific Division AAAS, 69–104.
- Largier, J., 2016. The Zone of Impact of San Francisco Bay Outflow: Plume Patterns and Nearshore Distribution of Pollutants in the Gulf of Farallones. Bodega Marine Laboratory. Available online: www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/amendments/es_tuarineNNE/science_plan/Final%20Report%20Largier.pdf [Verified August, 2019].

Chapter 7—Transport Model

Lebreton, L., Slat, B., Ferrari, F., Sainte-Rose, B., Aitken, J., Marthouse, R., Hajbane, S., Cunsolo, S., Schwarz, A., Levivier, A., Noble, K., Debeljak, P., Maral, H., Schöneich-Argent, R., Brambini, R., and Reisser, J., 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. *Scientific Reports.* 10.1038/s41598-018-22939-w.

MacWilliams, M.L., Bever, A.J., Gross, E.S. Ketefian, G.S., and Kimmerer, W.J., 2015. Three-dimensional modeling of hydrodynamics and salinity in the San Francisco Estuary: An evaluation of model accuracy, X2, and the low-salinity zone. *San Francisco Estuary and Watershed Science* (13)1.

Rayson, M.D., Ivey, G.N., Jones, N.L., and Fringer, O.B., 2018. Resolving high-frequency internal waves generated at an isolated coral atoll using an unstructured grid ocean model. *Ocean Modelling.* 122, 67–84.

Scheu, K., Fong, D., Monismith, S., and Fringer, O., 2015. Sediment transport dynamics near a river inflow in a large alpine lake. *Limnology & Oceanography* 60: 1195–1211.

Smith, S., and Hollibaugh, J., 2006. Water, salt and nutrient exchanges in San Francisco Bay. *Limnology & Oceanography* 51(1, part 2), 504–517.

Sutton, R., Mason, S., Stanek, S., Willis-Norton, E., Wren, I., and Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. *Marine Pollution Bulletin.* 109(1), 230–235.

Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay. SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA.

WäldschLAGGER, K., and Schüttrumpf, H., 2019. Effects of particle properties on the settling and rise velocities of microplastics in freshwater under laboratory conditions. *Environmental Science & Technology.* 53: 1958–1966.

Williams, N., Walling, D., and Leeks, G., 2004. The settling behaviour of fine sediment particles: Some preliminary results from LISST instruments. *Sediment Transfer through the Fluvial System. Symposium Proceedings, Moscow. IAHS Publication 288.*

CHAPTER

8

Microparticles and Microplastics in San Francisco Bay:

**CONCEPTUAL MODEL,
CROSS-MATRIX SYNTHESIS,
& DIRECTIONS FOR FUTURE RESEARCH**

by Rebecca Stutton



Highlights

- ◆ A refined conceptual model of major pathways of microplastic pollution for San Francisco Bay is presented alongside a comprehensive review of likely sources to urban stormwater runoff and treated wastewater discharges.
- ◆ Comparison of urban stormwater and treated wastewater indicated that beyond the large differences in estimated loads to the Bay, there were also considerable differences in relative proportions of different polymers, and more limited differences based on morphology. The large contribution of black, rubbery fragments was a dominant feature in urban stormwater samples. Meanwhile, wastewater samples indicated influence of multiple sources, including plastics used in textiles (acrylic and polyester fibers), as well as microbeads used in personal care products and fragments of single-use items (polyethylene).
- ◆ Comparison of surface water and sediment samples likewise indicated that polymer type was generally the most influential variable in determining whether different types of microplastics were preferentially concentrated in one matrix or the other. Buoyant polymers were more likely to be found in surface water, while denser particles were often found in sediment. In addition, the foam morphology was more often observed in surface waters. Transport model predictions similarly suggest that the movement and fate of microplastics is heavily dependent on polymer type, with denser particles destined to be trapped within the Bay, while more buoyant particles can be transported to the marine sanctuaries.
- ◆ This study synthesis indicated identification of specific plastic polymers is essential for pinpointing potential sources of microplastics, as well as predicting the movement of these particles within and through estuarine ecosystems.
- ◆ Key data gaps for San Francisco Bay remain, including additional information on the sources and pathways of microplastics, the exposure of a greater diversity of Bay aquatic organisms and associated risk for adverse impacts, more comprehensive information resulting from essential improvements in methodology, and the effects of current and future solutions implemented to reduce microplastic pollution.

Objectives

This multifaceted study of San Francisco Bay provides the first-ever comprehensive examination of microparticles and microplastics in an estuary, generating data that allow for unique, cross-matrix evaluations that advance our understanding of the sources and pathways of these contaminants, as well as their movement and fate in the environment. In this report, we have distinguished between microparticles, which are small particles (less than 5 mm) that are visually identified as potentially plastic, and microplastics, which have been confirmed to be plastic through Raman or FTIR spectroscopy.

As presented in the study's Sampling and Analysis Plan (Sedlak et al. 2017), specific cross-matrix hypotheses include:

1. Concentrations of microplastics in stormwater and wastewater will be comparable; however, the composition of the microplastics will be different.
2. Water and sediment from the same location will contain different types of microplastics (e.g., morphology and chemical composition).

Exploration of these hypotheses and further synthesis of study findings has provided critical insights, allowing refinement of conceptual models of microplastic sources and pathways in the Bay Area, as well as the movement and fate of these contaminants in San Francisco Bay and the adjacent ocean environment. This data synthesis can inform a broader response to the RMP microplastics Management Questions 1 and 3, regarding the levels of microplastics present in the Bay and marine sanctuaries, as well as the sources, pathways, loadings, and processes leading to this pollution (Sutton and Sedlak, 2017). The synthesis of microplastic findings also highlights critical policy-relevant data gaps that merit additional investigation.



Methods

To evaluate hypotheses regarding the composition of urban stormwater runoff vs. wastewater and surface water vs. sediment, principal component analysis (PCA) was carried out using the eigen decomposition method (Lê et al., 2008).

Stormwater samples were limited to those from watersheds with urban land uses greater than 60%, to assure a focus on urban stormwater runoff. This land-use threshold excludes three sites: Rodeo Creek, Refugio Creek, and Coyote Creek (Chapter 2 Stormwater). Wastewater samples were labeled with respect to treatment type: secondary or tertiary (the latter including dual media filtration; Chapter 3 Wastewater). Urban stormwater and wastewater samples were first evaluated for broad differences in particle morphology and/or color for all microparticles observed in each sample. Visualizations of the first two principal components of these analyses are presented, along with loading plots with factors observed to contribute most to each principal component.

An additional PCA was conducted based on the subset of microparticles identified via spectroscopy as plastic, or strongly suspected of being plastic (i.e., microplastics). The relative contributions of the first two principal components for polymer types present in stormwater and wastewater microplastics were compared. This included specific polymers such as polyethylene, polypropylene, and polystyrene; the broader category of anthropogenic synthetic microplastics, which are conclusively plastic but could not be classified to the specific polymer via spectroscopy; and black, rubbery fragments that could not be identified conclusively as plastic, but are suspected of being rubber. For purposes of this work, all anthropogenic polymers, including rubber, are defined as plastics.

The same microparticle and microplastic characteristics were evaluated in surface water vs. sediment samples collected in San Francisco Bay (excluding surface water samples from the marine sanctuaries and sediment samples from the reference site, Tomales Bay). The analyses were conducted using surface water datasets refined in two different ways: 1) including only sites where all particle morphologies, including fibers, were enumerated; and 2) including all sites, and excluding all fibers in water and sediment samples.

These comparisons were evaluated based on proportion rather than particle counts in each sample to account for the difference in sample volume and total number of particles within and between sample types.

Microplastics sources and pathways in the San Francisco Bay Area

A conceptual model describing major sources of microplastics, as well as the pathways by which they are transported to the Bay, is an essential tool for understanding these contaminants and informing pollution prevention activities. Some microplastics were designed and manufactured to be microplastic-sized (less than 5 mm), and are often referred to as primary microplastics. In contrast, most microplastics end up in the environment through fragmentation of larger plastic items, which release microplastics during or after use; these are known as secondary microplastics.

Microplastics from primary and secondary sources are carried to receiving waters through transport pathways. For the Bay, the major pathways relevant to microplastics include: 1) urban stormwater runoff; 2) treated wastewater effluent; 3) atmospheric deposition (directly to the Bay, as well as to other pathways, particularly urban stormwater); 4) discharges from the Sacramento and San Joaquin Rivers, which aggregate upstream inputs from wastewater, urban stormwater, as well as agricultural runoff; and 5) aquatic activities such as fishing, boating, marine transportation and shipping, and in-water or shoreline recreation (Figure 8.1). Because there is relatively little agricultural land use in the Bay Area, agricultural inputs are not presently considered a major pathway; this data gap is discussed below.

In this study, we focused on characterizing microparticles and microplastics in stormwater and wastewater discharged to the Bay, with the goal of understanding their relative contributions and identifying potential microplastic sources. Our data indicated stormwater may contribute hundreds of times more microparticles and microplastics to the Bay on an annual basis relative to wastewater. Because urban stormwater and wastewater collect microplastics from different arrays of sources (Tables 8.1 and 8.2), another key hypothesis of our study was that each of these pathways would have distinctive and specific microplastic signatures related to the influence of these different sources.

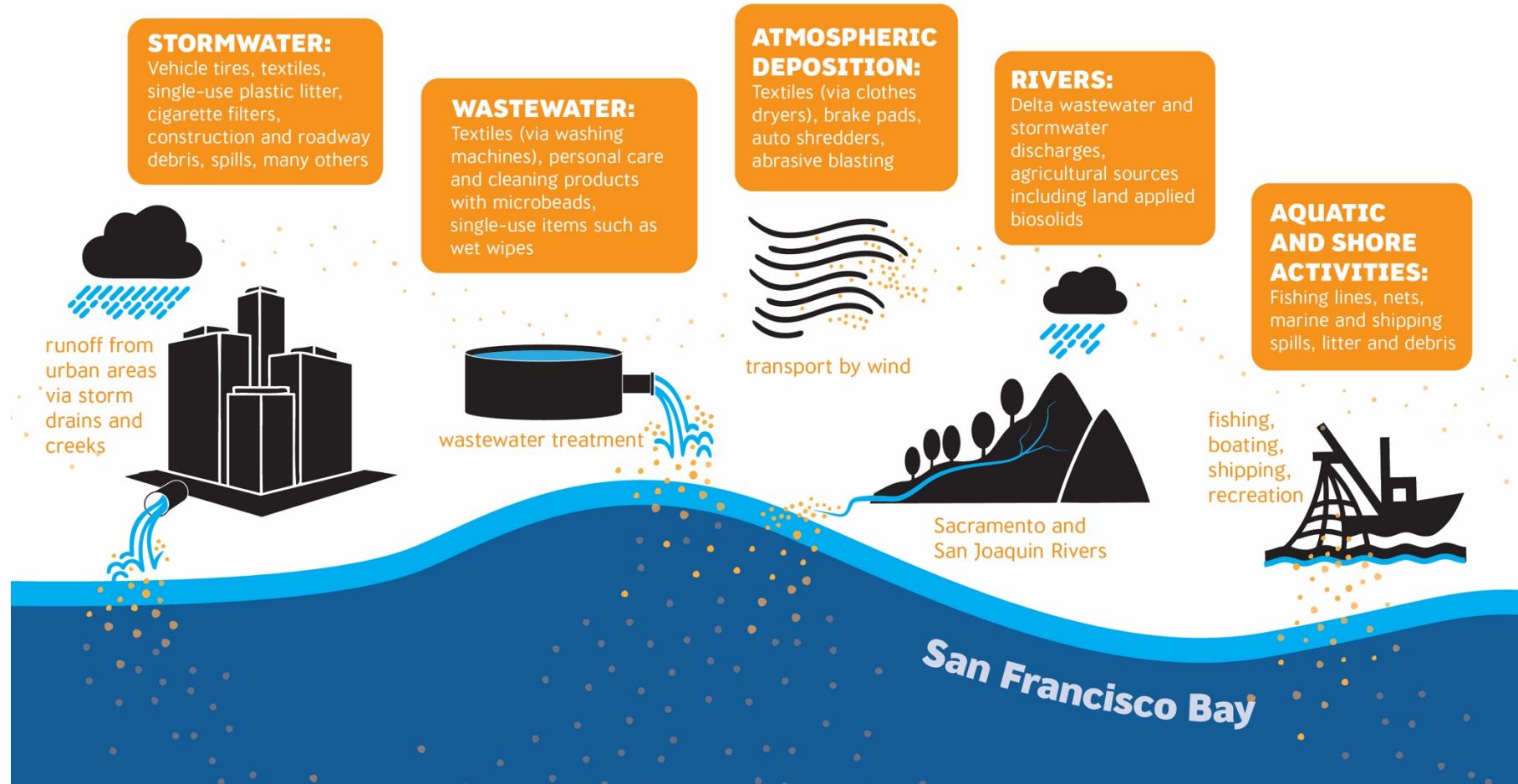


Figure 8.1. Conceptual model of major sources and pathways for microplastics in San Francisco Bay. Designed by Katie McKnight, SFEI.

Table 8.1. Sources of microplastics to wastewater.

Source	Common Polymers	Typical Particle Characteristics	Relevant References
Textiles (including synthetic clothing, carpets, home furnishings; discharged from washing machines)	Acrylic; Nylon; Polyester (PET) ¹	Fiber and Fiber bundles	Almroth et al., 2018; Browne et al., 2011; Dris et al., 2017; Hartline et al., 2016; Henry et al., 2019; Hernandez et al., 2017; Pirc et al., 2016
Microbeads used in personal care and cleaning products	Polyethylene; Polylactic acid; Polypropylene; Polyurethane	Sphere; 10 to 500 µm (~10%) Fragment; 10 to 500 µm (~90%)	Browne, 2015; Chang, 2015; Fendall and Sewell, 2009; Rochman et al., 2015a, 2015b; Scudo et al., 2017; Verschoor et al., 2016
Microbeads used in water softening and purification as well as other medical and industrial processes	Polystyrene	Sphere; 200 to 1,000 µm ²	Ballent et al., 2016; Mani et al., 2019
Single-use items disposed of down the drain (including wet wipes, packaging, menstrual products, diapers, cotton swabs)	Broad	Broad	Galanty, 2012; Kawecki and Nowack, 2019; Mourkgogiannis et al., 2018; Pantoja Munoz et al., 2018
Items contribution to indoor dust (including plastic particles derived from abrasion of objects and environment, primarily indoors)	Broad	Broad	Boucher and Friot, 2017; Dris et al., 2017; Liu et al., 2019
Treatment-related flocculants	Polyacrylamide, Polyvinylamine	Fragment, foam ³	Scudo et al., 2017

¹ Synthetic textiles do not include those made exclusively of natural fibers such as cotton, wool, silk, rayon, modal, Lyocell, etc.² Observed in surface water and sediment, but not wastewater in this study.³ Not observed in this study.

Table 8.2. Sources of microplastics to urban stormwater runoff.

Source	Common Polymers	Typical Particle Characteristics	Relevant References
Vehicle tires	Synthetic (styrene-butadiene) rubber; Natural (isoprene) rubber ¹	Fragment	Edil, 2008; Kole et al., 2017; Sommer et al., 2018; Verschoor et al., 2016
Synthetic turf and other recycled tire materials	Synthetic (styrene-butadiene) rubber; Natural (isoprene) rubber; Polyurethane	Fragment	Hann et al., 2018; Lassen et al., 2015
Brake pads	Phenolic resins; Kevlar	Fragment, fiber; most under 20 µm, too small to be observed using present study methods	Grigoratos and Martini, 2015; Sommer et al., 2018
Road markings (thermoplastic) ²	Styrene copolymer; Ethylene/vinyl acetate copolymer (EVA); Nylon	Fragment	Vogelsang et al., 2019
Building paint	Polyethylene; Polypropylene; Polylactic acid; Polystyrene/ acrylic copolymer	Fragment, sphere, fiber	Lassen et al., 2015; Scudo et al., 2017; Verschoor et al., 2016
Other construction materials	Broad	Broad	Correia Diogo, 2015; Gordon, 2006; Kawecki and Nowack, 2019; Scudo et al., 2017
Single-use bags and wraps	Polyethylene; Polypropylene	Film	McKeen, 2013
Single-use foam polystyrene	Polystyrene	Foam	Allen et al., 2017; Gordon, 2006
Other single-use packaging and foodware	Broad	Broad	Allen et al., 2017; Gordon, 2006

Source	Common Polymers	Typical Particle Characteristics	Relevant References
Cigarette filters	Cellulose acetate	Fiber	Allen et al., 2017; Gordon, 2006; Slaughter et al., 2011
Textiles (including synthetic clothing, carpets, home furnishings, outdoor items; lint from clothing dryers)	Acrylic; Nylon; Polyester (PET)	Fiber	Almroth et al., 2018; Browne et al., 2011; Dris et al., 2017; Hartline et al., 2016; Henry et al., 2019; Hernandez et al., 2017; Pirc et al., 2016
Outdoor dust (including microplastics derived from abrasion of objects and other infrastructure)	Broad	Broad	Boucher and Friot, 2017
Pre-production pellets (nurdles)	Broad	Sphere, generally 1-5 mm	Berg, 2019; Karlsson et al., 2018
Pre-production powders (fluff)	Broad	Sphere; Fragment (more irregularly shaped as decrease in size)	Duis and Coors, 2016; Karlsson et al., 2018
Industrial waste	Broad	Broad	Boucher and Friot, 2017; Kawecki and Nowack, 2019
Spills (leakage from trash collection efforts)	Broad	Broad	Kawecki and Nowack, 2019; Löhr et al., 2017
Landfill leachate	Broad	Broad	He et al., 2019

¹ While vehicle tires include synthetic (styrene-butadiene) and/or natural (isoprene) rubber, they also include carbon black as a major non-plastic ingredient, at up to 40% of the mass (Kole et al., 2017). Spectra matching carbon black and similar materials were frequently observed for black, rubbery particles in stormwater and sediment samples; however, these spectra cannot be considered a sufficiently selective indicator of vehicle tires. Such particles were classified as unknown potentially rubber.

² Glass spheres have been observed in Bay Area stormwater and are thought to be components of reflective road paint (Gilbreath et al., 2019).

Comparison of urban stormwater and wastewater data

Principal component analysis indicated Bay Area urban stormwater and wastewater samples had noticeably different distributions of microparticles and microplastics (Figures 8.2–8.4). A PCA-based examination of the morphologies of all microparticles showed urban stormwater and wastewater readily separated into distinct clusters (Figure 8.2). This clustering appears to be driven by greater contributions of foams (largely non-plastic stearates) and fibers in wastewater, and more prevalent fragments in urban stormwater (including but not limited to the black, rubbery fragments classified as unknown potentially rubber). When color was included alongside morphology, the analysis indicated that for these urban stormwater and wastewater samples, foams are positively correlated with the color white, fragments are positively correlated with the color black (and negatively correlated with clear), and fibers are correlated with blue (Figure 8.3). An analysis of urban stormwater and wastewater microparticles based on color alone did not result in separate clusters by pathway (not shown).

Additional analysis focusing solely on the subset of sample particles spectroscopically confirmed or strongly suspected to be microplastics—including all identified plastic polymers as well as the black, rubbery particles classified as unknown potentially rubber—also revealed distinct clustering by pathway (Figure 8.4). These clusters appeared to be driven by the presence of greater amounts of unknown potentially rubber particles in urban stormwater, as well as greater amounts of polyethylene, anthropogenic synthetic (specific plastic polymer could not be identified), and acrylic particles in wastewater.

In general, these analyses did not reveal a major difference between wastewater treated with more advanced dual media filtration and wastewater that did not receive this additional treatment (Figures 8.2–8.4). While the overall levels of microparticles were lower with this additional filtration (Chapter 3 Wastewater), there was no evidence that particular plastic types were selectively removed.

Overall, we can begin to see pathway-relevant profiles of microparticles and microplastics based on PCA and an examination of sources. Despite the diversity of potential sources to urban stormwater (Table 8.2), the large contribution of black, rubbery fragments in Bay Area samples overwhelmed the influence of most other microparticle and microplastic types. In contrast, wastewater appeared to be more evenly influenced by multiple sources (Table 8.1), including plastics commonly used in textiles (acrylic, polyester) and microbeads and single-use items (polyethylene), among others.

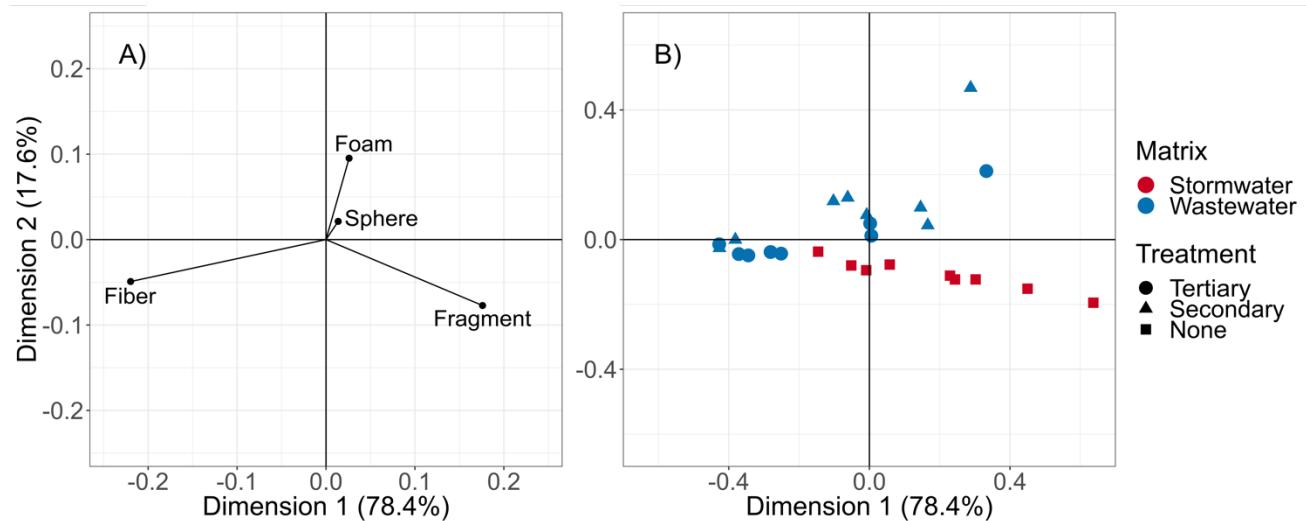


Figure 8.2. Principal component analysis of microparticle morphology in urban stormwater and wastewater: A) loading plot with influential characteristics; B) PCA data plot. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 96%.

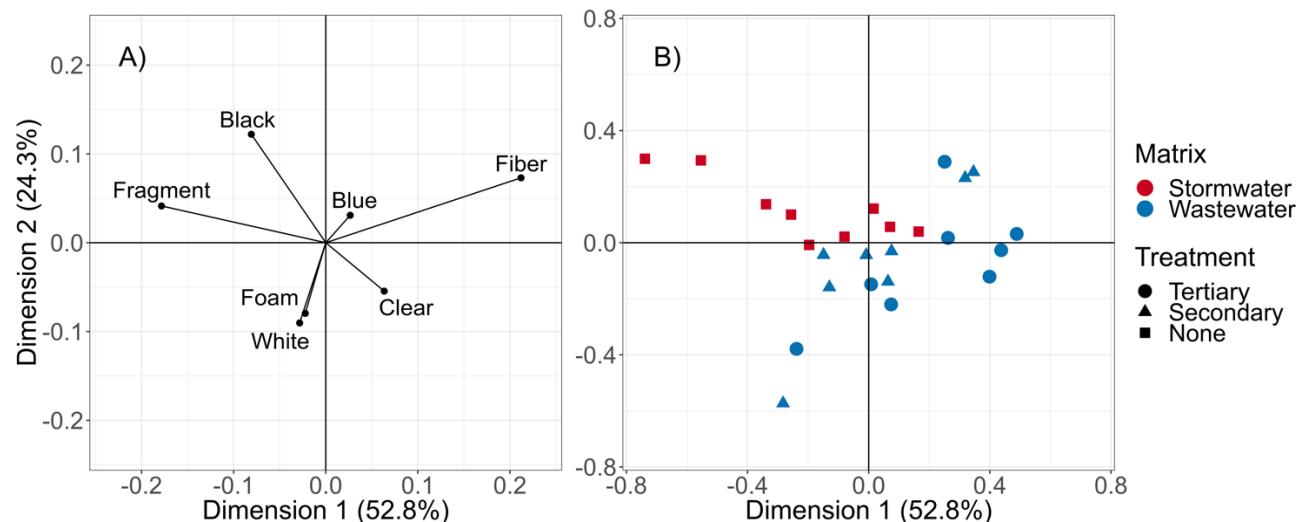


Figure 8.3. Principal component analysis of microparticle morphology and color in urban stormwater and wastewater: A) loading plot with influential characteristics; B) PCA data plot. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 77%.

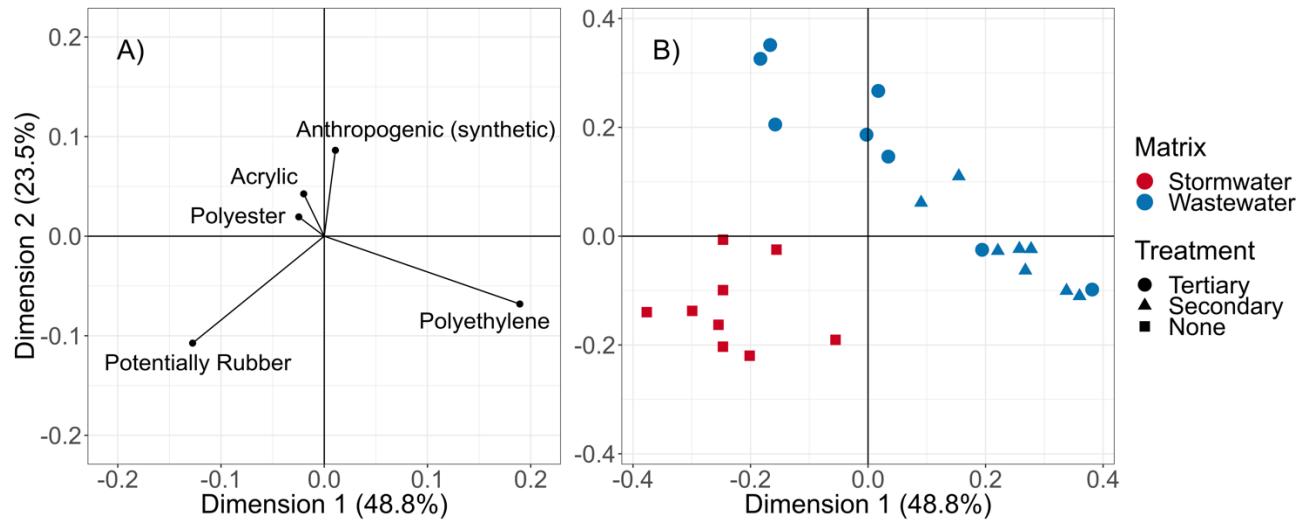


Figure 8.4. Principal component analysis of microplastic polymers in urban stormwater and wastewater: A) loading plot with influential characteristics; B) PCA data plot. Here, microplastics include the subset of microparticles confirmed as plastic, as well as those identified as unknown potentially rubber. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 72%.

Microplastics transport and fate in San Francisco Bay and adjacent ocean environment

Once discharged to San Francisco Bay from any of the five major pathways identified in the conceptual model (Figure 8.1), microplastics are subject to physical, chemical, and biological processes that affect their movement or transport within both the Bay and the coastal ocean environment. Macroplastic trash (greater than 5 mm) can also enter the Bay via stormwater or aquatic and shoreline activities, then fragment to release smaller microplastics through weathering and abrasion, adding to the overall burden of microplastics in the environment. Ultimately, the complex interactions between the physical and chemical characteristics of individual microplastics and broader estuarine and marine processes dictate their fate in the ecosystem.

Transport and fate within San Francisco Bay

A generalized conceptual model for macro- and microplastic distribution mechanisms in estuarine environments identified the following relevant factors (Vermeiren et al., 2016):

- ◆ Particle-specific processes affecting the position of plastics in the water column (polymer or particle density, morphological characteristics, and biofouling);
- ◆ Larger-scale processes related to the mixing of fresh and saltwater, with different patterns of movement depending on the level of stratification in the estuary; and
- ◆ Local processes resulting from the influences of wind, topography, and organism-plastic interactions.



According to this conceptual model, denser microplastics would be expected to deposit within the Bay sediment, while more buoyant microplastics could be transported for longer distances in surface waters or deeper in the water column (Vermeiren et al., 2016).

The transport model developed as part of the San Francisco Bay Microplastics Project generated predictions consistent with this conceptual model (Chapter 7 Transport Model). Particle rising and settling rates were driven primarily by density and/or morphology and, according to the transport model, microplastics with

even minimal settling rates would be trapped within the Bay. Meanwhile, a portion of buoyant particles could be transported through the Golden Gate and into the adjacent National Marine Sanctuaries. The strong influence of density and morphology on the behavior of microparticles and microplastics in the transport model suggests that Bay surface water and sediment samples may have noticeably divergent distributions of these properties.

Comparison of Bay surface water and sediment data

Principal component analysis indicated that Bay surface water and sediment samples did, in fact, display different overall microplastic signatures (Figures 8.5–8.12). Our first analyses focused on the portion of each sample spectroscopically evaluated and considered to be likely or confirmed microplastics. Sediment data were compared to two subsets of surface water data: 1) data from only those surface water sites where all particle morphologies, including fibers, were enumerated; and 2) data from all sites, with all fibers excluded from both water and sediment samples.

Examination of the distribution of polymers in sediment and surface water revealed a clear distinction between sample types (Figures 8.5 and 8.6), although excluding all fiber-related information revealed slightly less differentiation between the two sample types (Figure 8.6). The sediment cluster in both PCA plots was strongly influenced by contributions from unknown potentially rubber, polystyrene, and cellulose acetate (fibers), while the surface water cluster was strongly influenced by polyethylene, and for fibers, polyester and polypropylene microplastics. The drivers of the sediment cluster were all polymers with estimated densities greater than water ($1.03\text{--}1.30\text{ g/cm}^3$; Table 7.1), and would therefore be expected to sink.

Two of the three polymers driving the water cluster have densities lighter than water (polyethylene 0.94 g/cm^3 , polypropylene 0.90 g/cm^3 ; Table 7.1), and would be expected to float in the absence of biofouling. In contrast, polyester, a term used specifically to describe fibers made of polyethylene terephthalate (PET), has a density greater than water (1.38 g/cm^3). As noted in Chapter 7, fibers have significant drag, such that they tend towards passive transport (negligible rising/settling velocities), and are therefore not expected to sink as quickly as their density alone might suggest.

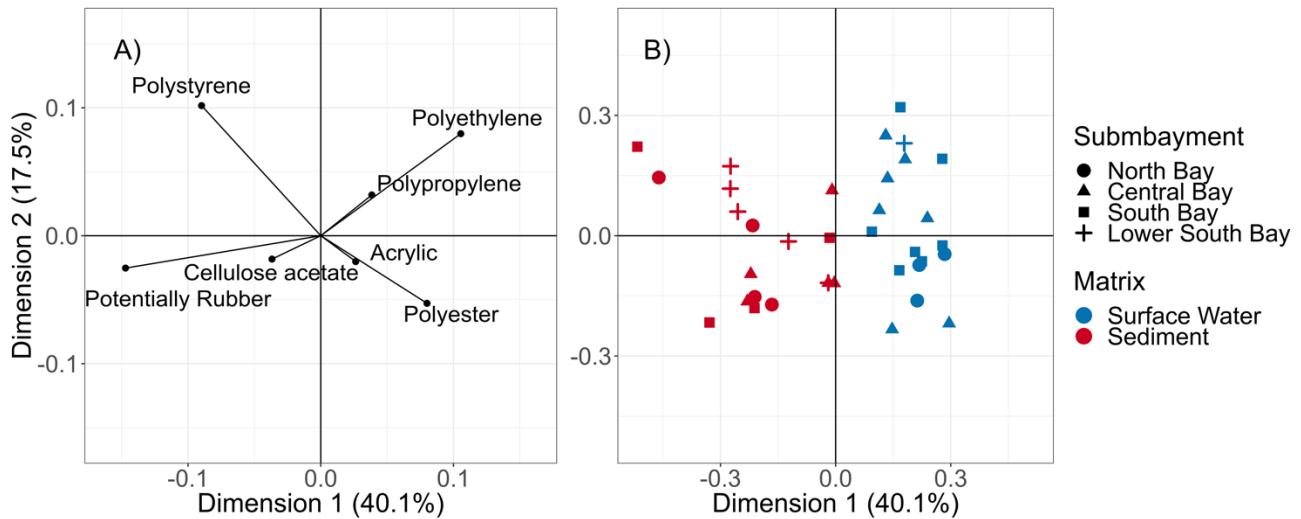


Figure 8.5. Principal component analysis of microplastic polymers in surface water and sediment: A) loading plot with influential characteristics; B) PCA data plot. Here, microplastics include the subset of microparticles confirmed as plastic, as well as those identified as unknown potentially rubber. Surface water sites are limited to those for which all morphologies, including fibers, were enumerated. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 58%.

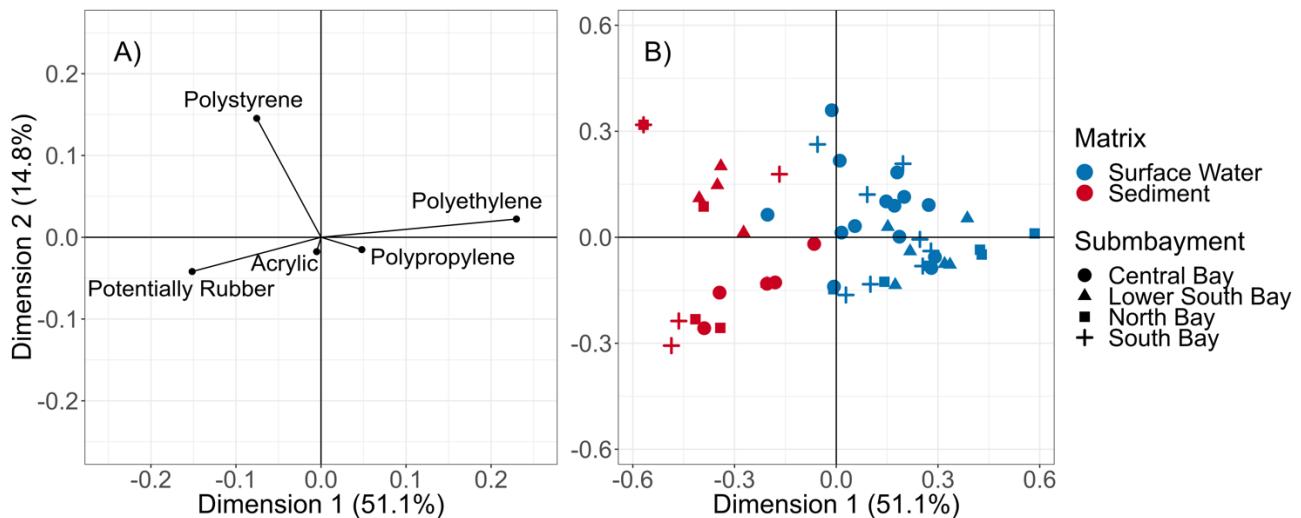


Figure 8.6. Principal component analysis of microplastic polymers in surface water and sediment, excluding fibers: A) loading plot with influential characteristics; B) PCA data plot. Here, microplastics include the subset of microparticles confirmed as plastic, as well as those identified as unknown potentially rubber. All Bay surface water sites are included, and all fiber data in both surface water and sediment is excluded. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 66%.

In contrast to these polymer-specific analyses, PCA of microparticle morphologies, including all particles observed rather than the subset of those identified as plastic or potentially plastic via spectroscopy, indicated more limited separation of water and sediment. Examination of the microparticle morphologies for the subset of surface water data including fibers revealed relatively little differentiation by matrix (Figure 8.7). Both surface water and sediment appeared to contain broad and relatively consistent distributions of fragments and fibers. Inclusion of a

greater number of surface water sites and exclusion of fibers improved the level of differentiation slightly, due to more foam contributions to surface water samples and more consistent fragment contributions to sediment samples (Figure 8.8). Of note, the foams included in this PCA may be plastic, unlike the foam typically observed in wastewater (Chapter 3 Wastewater). These surface water foams are assumed to be highly buoyant with minimal density (estimated 0.10 g/cm³; Table 7.1) due to the bubbles of gas in the expanded plastic.

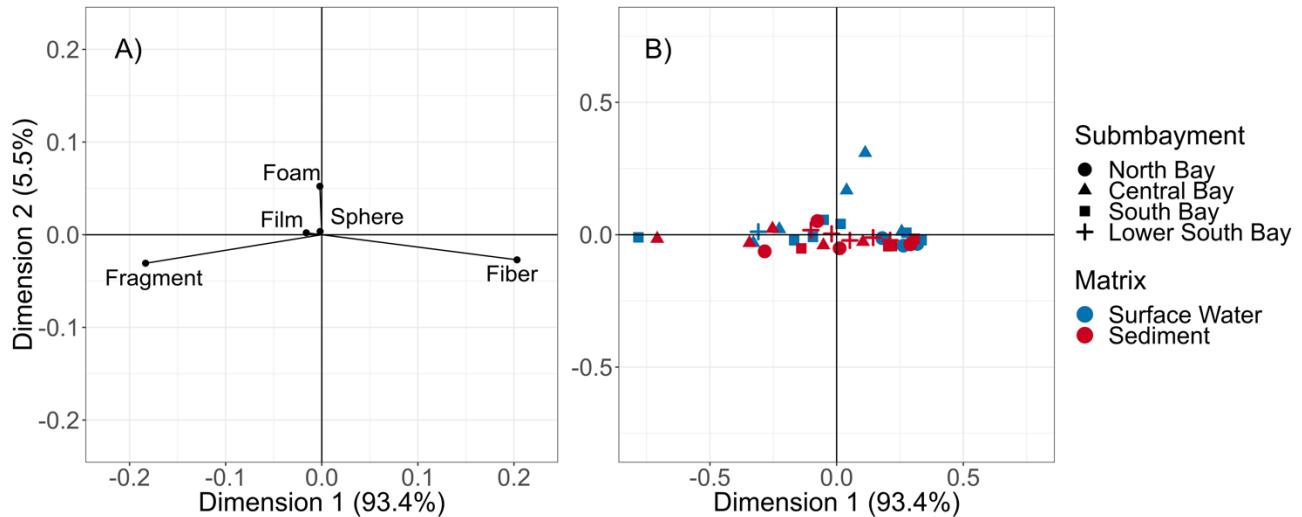


Figure 8.7. Principal component analysis of microparticle morphology in surface water and sediment: A) loading plot with influential characteristics; B) PCA data plot. Surface water sites are limited to those for which all morphologies, including fibers, were enumerated. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 99%.

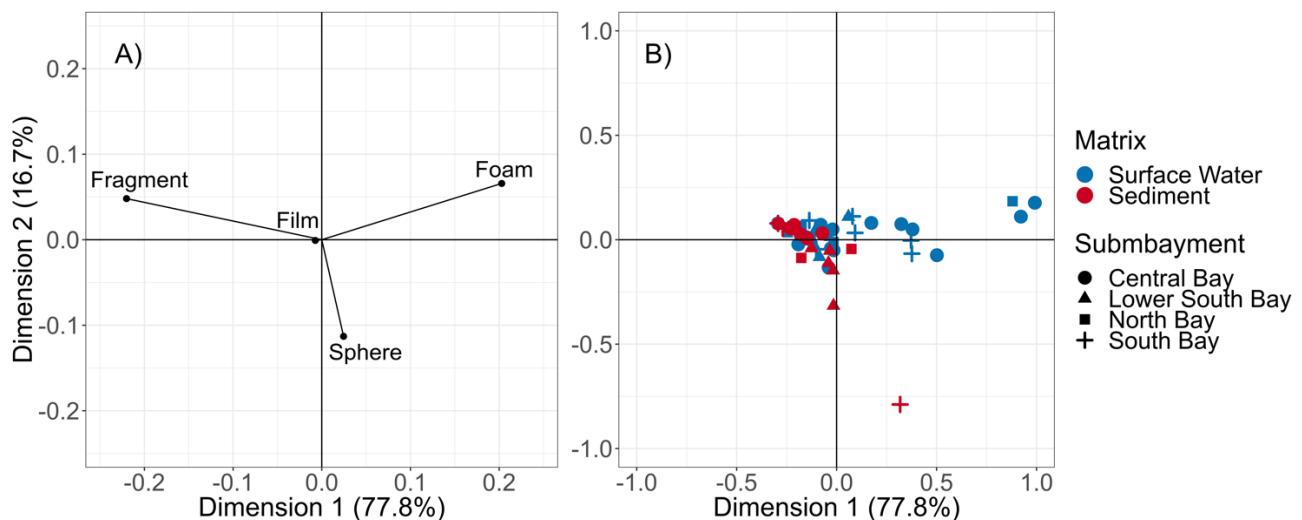


Figure 8.8. Principal component analysis of microparticle morphology in surface water and sediment, excluding fibers: A) loading plot with influential characteristics; B) PCA data plot. All Bay surface water sites are included, and all fiber data in both surface water and sediment are excluded. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 95%.

Inclusion of color alongside morphology resulted in better separation of the datasets with fibers and limited water sites (Figure 8.9) and all Bay sites without fibers (Figure 8.10). When fibers were included, drivers of the separation included black and dark blue microparticles in surface water and clear microparticles in sediment (Figure 8.9). When a greater number of surface water sites were included, and fibers excluded, surface water clusters were evident due to higher proportions of foam particles, and white and clear microparticles, while sediment clusters were related to black microparticles (Figure 8.10). Analyses of surface water and sediment microparticles based on color alone resulted in intermediate levels of clustering by sample type (not shown).

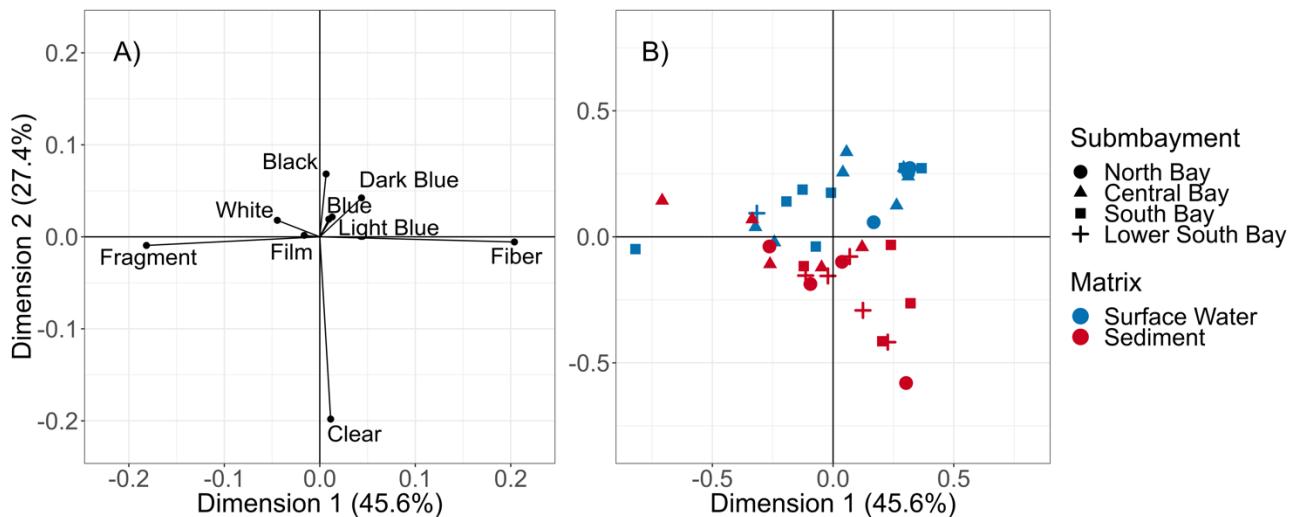


Figure 8.9. Principal component analysis of microparticle morphology and color in surface water and sediment: A) loading plot with influential characteristics; B) PCA data plot. Surface water sites are limited to those for which all morphologies, including fibers, were enumerated. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 73%.

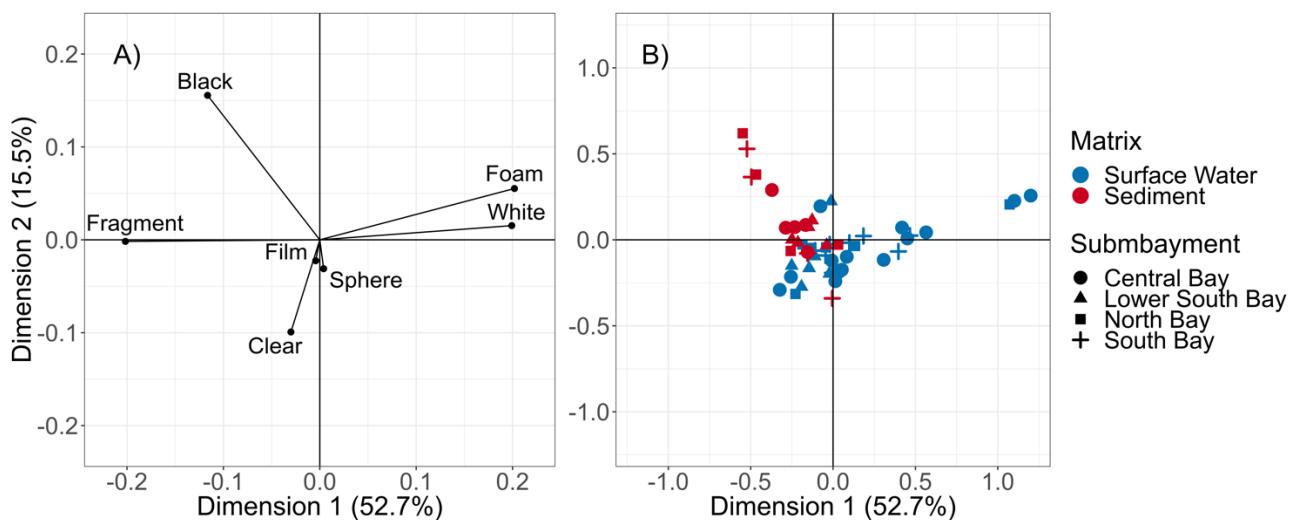


Figure 8.10. Principal component analysis of microparticle morphology and color in surface water and sediment, excluding fibers: A) loading plot with influential characteristics; B) PCA data plot. All Bay surface water sites are included, and all fiber data in surface water and sediment are excluded. Scales of each plot were optimized for display. Proportion of variance explained by the first two principal components is 68%.

Chapter 8—Synthesis

This PCA-based exploration of surface water and sediment data indicated that polymer type was generally the most influential variable determining whether individual microplastics would be found in surface water or sediment. Particle morphology, particularly foam, can also influence the location of microplastics in the Bay. This is consistent with the findings of the transport model (Chapter 7 Transport Model) and a more general conceptual model of microplastic distribution in estuaries (Vermeiren et al., 2016).

The conceptual model further suggests microplastics may accumulate in zones where fresh and saltwater converge near the Bay bottom, and at sites of higher wind exposure for buoyant microplastics (Vermeiren et al., 2016). The transport model likewise predicted accumulation zones at a site of estuarine convergence (North Bay), as well as a more wind-driven location (South Bay shoals; Chapter 7 Transport Model).

The predictions of the transport model within the Bay were largely consistent with limited available sediment monitoring data (Chapter 5 Sediment), which indicated the greatest abundance of microparticles was in Lower South Bay, a region with limited oceanic or freshwater flushing that is strongly influenced by both wastewater and urban stormwater discharges. Agreement was not as strong for surface water measurements within the Bay (Chapter 4 Surface Water), which is likely a product of the high degree of variability in this matrix, as suggested by comparison of field duplicates as well as by transport model predictions of persistent spatial gradients.

The ultimate fate of microplastics that remain within San Francisco Bay is inextricably tied to the movement and fate of sediment, an area of active investigation in the region. The transport model indicated that a large portion of the buoyant microplastics, and essentially all of the denser microplastics, are likely to remain trapped within the Bay, at least under the timescales examined (Chapter 7 Transport Model). Dense particles are expected to congregate at the Bay bottom. Even the more buoyant microplastics floating at marine or estuarine surfaces may eventually settle to the sediment due to biofouling, although defouling and resurfacing of particles is also possible (Vermeiren et al., 2016).

Much of the channel of the Bay is considered an erosional sediment environment (Barnard et al., 2013), and microplastics deposited onto the sediment in these regions could be particularly susceptible to remobilization. Meanwhile, sites in the nearshore margins of the Bay are often depositional, such that entrained microplastics could be subject to net accumulation through burial. In either setting, remobilization and potential transport through ingestion by mobile aquatic organisms could modify the fate of a subset of particles. Broader study of sediment in the Bay, largely designed to inform adaptation to sea level rise, may inform an improved understanding of the fate of sediment-associated microplastics.

Transport to the coastal ocean

According to the general conceptual model for microplastics in estuarine environments (Vermeiren et al., 2016), as well as hypotheses specifically developed for the San Francisco Bay Microplastics Project (Sedlak et al., 2017), San Francisco Bay was anticipated to be a source of microplastic contamination to the marine environment. While the Pacific Ocean is not pristine and contaminant-free, levels of microplastics were expected to be lower in coastal Pacific surface waters relative to the Bay, with net movement of particles from the Bay to the marine sanctuaries. Surface water monitoring results supported this hypothesis, with observations of microparticle abundances considerably lower in surface water in the marine sanctuaries than in the Bay (Chapter 4 Surface Water).

Likewise, the transport model predictions (Chapter 7 Transport Model) indicated that some microparticles released into San Francisco Bay could reach the nearby marine sanctuaries over a modeled time period of 30 days. Essentially all settling particles were predicted to be trapped within the Bay, whereas 40–100% of buoyant particles were estimated to exit the Bay over a time scale of days to weeks. These buoyant particles were efficiently transported in the freshwater plume leaving the Bay. The model predicts that the freshwater plume could transport many particles northward along the coast, while regional winds and coastal currents could result in dispersion south and west.

The fate of microplastics in terms of retention or export from an estuarine setting like San Francisco Bay appears to be largely dependent on their buoyancy, determined by polymer type in most cases, and secondarily by morphology. This dependence makes polymer identification an essential aspect in predicting the transport and fate of microparticles entering estuarine waters.

One uncertainty that remains unaddressed by the present monitoring and modeling effort is the potential for import of oceanic microplastics into the Bay through the movement of heavier bedload material via incoming tidal flow. Partially mixed estuaries like San Francisco Bay can experience this type of sediment transport (Vermeiren et al., 2016), and marine sediments often hold higher levels of microplastics relative to surface waters (Hidalgo-Ruz et al., 2012). Recent study of the flux of suspended sediment at the Golden Gate suggested import of bed sediment from the marine environment is possible (Downing-Kunz et al., 2017). As noted previously, the behavior of sediment in the Bay is under-characterized and an area of active research, with future findings likely to inform an improved understanding of the transport and fate of microplastics in both the Bay and the coastal ocean environments.

Key data gaps remain for microplastics in San Francisco Bay

Mitigating microplastic pollution in the environment requires understanding the sources of microplastics, the pathways microplastics can follow to reach the Bay, and their relative contributions. Effective risk evaluation requires an understanding of both exposure and potential adverse impacts. Finally, new and improved methodologies will be required to collect data and inform our understanding of sources, pathways, transport, and fate of microplastics.

Sources and pathways

The conceptual model developed for San Francisco Bay identifies a myriad of possible sources of microplastics to urban stormwater and wastewater in particular (Figure 8.1, Tables 8.1 and 8.2). A few of these sources have been evaluated in the literature with respect to their potential to contribute to plastic pollution to marine ecosystems (Boucher and Friot, 2017), but many others have not been thoroughly characterized. Improved information regarding the relative contributions of sources of microplastics is particularly useful to regional stakeholders attempting to identify effective solutions to address microplastic contamination.

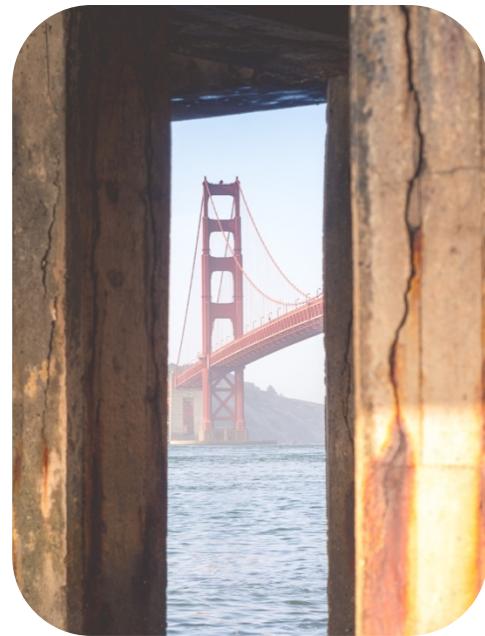
In particular, identifying the relative contribution of secondary microplastics formed from the breakdown of larger trash has not been thoroughly studied. The properties of individual particles, including polymer type, other chemical components, color, morphology, and size can be used to help elucidate microplastic sources (Ballent et al., 2016). However, this approach has so far provided only a limited level of source-related information, and requires more evidence of direct links between particle characteristics and sources (Fahrenfeld et al., 2019).

Five primary pathways channel plastic pollution to the Bay (Figure 8.1): urban stormwater discharges; wastewater effluent from treatment plants; wind or airborne particles; riverine inputs, which aggregate urban stormwater, wastewater, and agricultural runoff inputs from the greater Central Valley watershed; and aquatic and shoreline activities. This study characterized microplastic inputs from wastewater and urban stormwater, but the relative contributions of other pathways, especially air deposition, remain unknown. Our findings regarding blank contamination, as well as recent studies of another urban setting, Paris, France (Dris et al., 2018, 2015), suggest that airborne deposition may be a significant pathway in urban areas, especially for fibers; further evaluation is warranted.

Likewise, characterization of inputs from agricultural runoff, particularly from fields that use wastewater-derived biosolids as fertilizer, is also a major data gap. Agricultural runoff enters

San Francisco Bay from relatively small amounts of local agricultural lands, as well as from much more extensive agricultural lands in the Central Valley via the Sacramento-San Joaquin River Delta. While Bay Area agricultural runoff is not presently considered a major pathway, further research may alter this conceptual model of microplastics in the Bay.

This study indicated that fibers are especially ubiquitous microplastics, yet the fibers entering the Bay via treated wastewater (e.g., from synthetic clothing shedding microfibers during washing) do not appear to be the only input. Investigation of additional fiber sources and pathways is warranted. The significant difference in estimated microplastic loads contributed from wastewater compared to stormwater also indicates that key urban sources, such as tire wear, may be more important than previously hypothesized. The large number of black, rubbery fragments identified in urban stormwater and sediment suggests that inputs from tire and road wear and use of recycled tires (e.g., artificial turf) may also merit further investigation. Recent studies have indicated that tire wear during use and tire recycling applications release significant amounts of rubber particles (Kole et al., 2017; Lassen et al., 2015; Sommer et al., 2018).



A comprehensive review of Bay Area stormwater microparticle and microplastic data in the context of the scientific literature may suggest land-use classifications or landscape attributes (e.g., impervious surface area or proximity to roadways) as key factors explaining higher levels of discharge and should be examined in future monitoring studies. Evaluating possible factors influencing microparticle and microplastic loads is important to identify potential sources, to better understand areas of uncertainty, and to identify key attributes that influence the generation of microplastics in urban stormwater.

Exposure and effects

This study assessed concentrations of microparticles and microplastics in prey fish gastrointestinal tracts. Prey fish represent an important link between abiotic environmental compartments and the food web, and may be an indicator of exposure to higher trophic level organisms, including sport fish and larger predators such as birds, seals, and humans.

Fibers were ubiquitous in fish gut samples in San Francisco Bay (Chapter 6 Prey Fish). Of the fibers detected in fish that were further analyzed via Raman spectroscopy, 21% were confirmed to be plastic, while 60% were classified as anthropogenic unknown because dyes

within the microfibers interfered with the spectra. These study findings indicate that microplastics, particularly plastic microfibers, are entering the Bay food web. Microplastics and their chemical components are likely to transfer up trophic levels (Nelms et al., 2018); assessing exposure of Bay predators such as sport fish is a priority for further study.

The likely effects of plastic microfibers on Bay wildlife are unknown. While there are studies identifying impacts of microplastic exposure to organisms, most have used only virgin plastic spheres and used exposures above ecologically relevant concentrations. Although fibers have been frequently detected in fish around the world, the only study, to our knowledge, to directly investigate microfiber effects on fish showed that ingested ethylene vinyl acetate fibers caused higher frequencies of progressive and inflammatory changes in the livers and intestines of goldfish compared to fragments and spheres (Jabeen et al., 2018). However, this study used high concentrations of virgin microplastics and thus is not predictive of environmental microfiber exposures. There is a clear need for ecotoxicological studies that evaluate the effects of plastic microfibers, especially those that have been weathered, at environmentally relevant concentrations.

The potential impacts of plastic microfibers may include those triggered by exposure to the dyes and other chemical additives they contain. Globally, over 10,000 different synthetic dyes and pigments are produced and used annually, many of which are environmentally recalcitrant and toxic to aquatic life (Drumond Chequer et al., 2013; Hassaan and Nemr, 2017). Azo dyes account for over half of all synthetic dye production and use, and can be carcinogenic and mutagenic, especially when metabolized by animal liver enzymes (Gita et al., 2017). However, the majority of ecotoxicological testing of dyes has focused on textile industry effluent, and impacts of chronic low-dose exposure to these dyes are still largely unknown. Similar to dyes, exposure to and effects of other chemical additives in microplastics (e.g., flame retardants, plasticizers) require additional study.

There is an urgent need for ecotoxicological studies that evaluate the effects of microplastics at environmentally relevant concentrations in organisms at multiple life stages (de Sá et al., 2018). However, even with more ecotoxicological data, establishing risk thresholds may not be possible given the diversity of microplastic sizes, morphologies, and chemistry. Threshold values for a single contaminant in a given environmental media are normally set to protect the most sensitive (tested) species, but in the case of microplastics, the adverse impacts are potentially more contaminant- and species-specific than has been thus far assumed in the field. Furthermore, initial evidence suggests that accumulation and toxicity of microplastics and other chemicals can be significantly different when organisms are exposed to mixtures rather than individual contaminants (Barboza et al., 2018; Chen et al., 2017; Pannetier et al., 2019a, 2019b; Rainieri et al., 2018; Wen et al., 2018; Zhang et al., 2019). Thus, while further

ecotoxicological study is essential, a truly robust and comprehensive evaluation of risk to aquatic wildlife may require more sophisticated approaches than currently available.

Methodology

The selection of sample collection and analysis methods for microplastics is a critical component of study design and will dramatically influence the results (Chapter 9 Lessons Learned; Brander et al., in review; Elert et al., 2017; Hidalgo-Ruz et al., 2012). Methods for characterizing microplastic contamination are rapidly evolving and critical gaps remain.

Current sample collection methods may not provide accurate characterization of environmental loads of fibers due to their ubiquitous presence as background contamination, as well as the uncertainty associated with the efficacy of sampling methods for catching microfibers (Barrows et al., 2017; Covernton et al., 2019; Kang et al., 2015). In addition, underestimates of microplastic occurrence are likely because many methods are not able to capture smaller size classes (Conkle et al., 2018). Development of reliable, standardized methods to capture fibers and smaller microplastics is necessary. One option suggested for collection of surface water samples is to pair the widely-used manta trawl collection method with another method better designed to capture fibers and smaller particles, such as bulk water grab samples (Chapter 4 Surface Water; Chapter 9 Lessons Learned).

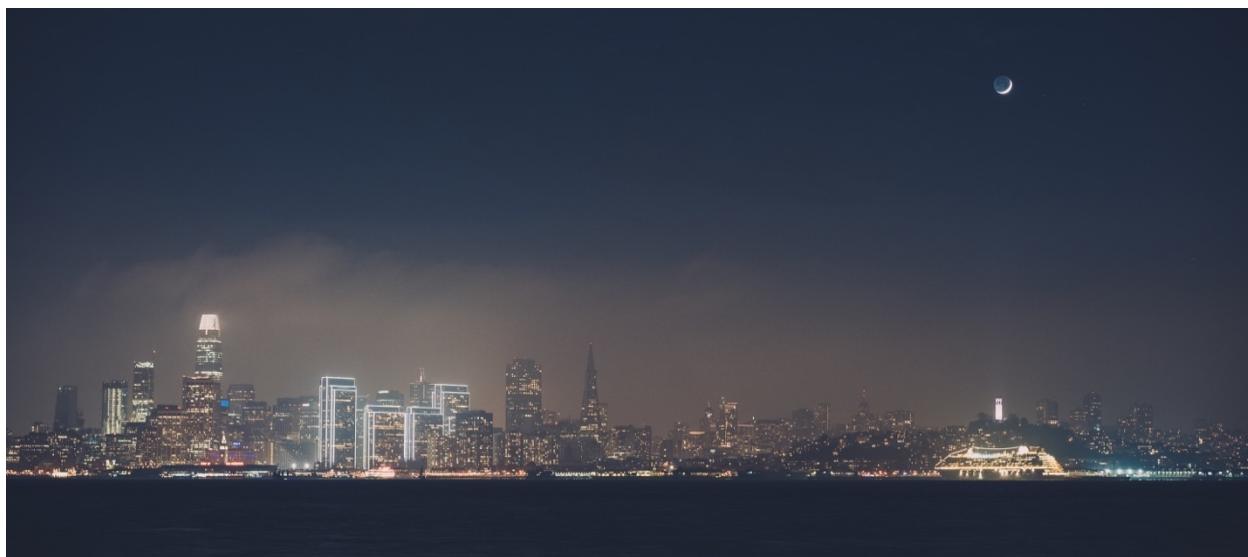
At present, analytical methods for identifying microplastics are extremely resource intensive, with automation an increasingly urgent priority for method development. Typically, researchers must visually select and extract microparticles by hand, then painstakingly apply confirmatory techniques such as Raman or FTIR spectroscopy to verify whether individual microparticles are microplastics. Many of the microparticles extracted from samples collected as part of this project could not be confirmed as plastic because it was time- and resource-prohibitive to conduct spectroscopy on every microparticle. Automated techniques that reduce the analytical burden to individual researchers and improve and standardize detection and quantification would be an invaluable contribution to the field.

Another major constraint in microplastic analysis is the technical difficulty in identifying the underlying plastic or material of many microparticles. For example, fibers from textiles often have dyes that create large peaks in the Raman spectrum, which can prevent determination of the type of material present—whether plastic or naturally-derived—via Raman spectroscopy. The large portion of particles, especially fibers, that could not be conclusively identified using spectroscopy in this study indicates a need for further development of material identification methods. A new identification method specific to microfibers, which combines several lines of evidence to support identification of the polymer or material present, was developed in parallel with the present study (Zhu et al., 2019). Widespread adoption of this approach and additional

analytical techniques are expected to increase information about the occurrence and characteristics of microplastics in environmental matrices.

Another challenge to polymer identification is the ability of existing spectral libraries to provide adequate matches for microplastics recovered from the environment, which reflect plastic compositions based on diverse sources, may have entrained environmental contaminants, and have undergone varying degrees of weathering. The San Francisco Bay Microplastics Project supported the development of an open access Raman spectroscopy polymer library that will be available to the public soon (Munno et al., *in review*). This library has a great diversity of environmentally observed microplastic morphologies, types, and colors, and will facilitate more accurate identification of particles, as well as consistency among research groups.

Finally, an urgent need in the field of microplastics is standard methods for both collecting and reporting field and laboratory blanks, and accounting for evidence of background contamination in field samples. In this report, we collected both field and laboratory blanks, which is not a consistent practice in the field. We presented blank sample data (microparticle counts) alongside field sample data, and provided guidance in interpreting the results by indicating which measurements were below thresholds for data qualification specific to each matrix and each particle morphology, in order to highlight which samples may be strongly influenced by background contamination from sample collection and analysis. A variety of other methods to acknowledge and account for background contamination are observed in the literature (Brander et al., *in review*), and this lack of standardization inhibits accurate calculation of occurrence, as well as cross-study comparisons. Improved understanding of the sources of background contamination in the field and laboratory may inform the selection of a reporting method, and may result in additional measures to reduce background contamination.



Additional monitoring with a focus on solutions

Additional monitoring in the region is needed, with specific focus on two questions relevant to management actions: 1) Have current and future pollution prevent activities led to reduced levels of microplastics in the Bay? and 2) How can green stormwater infrastructure be used to reduce levels of microplastics entering the Bay via urban stormwater?

This study represents a preliminary baseline characterization of microplastics in San Francisco Bay; monitoring to assess trends in microplastic levels will also be necessary, particularly to assess the efficacy of management actions. Follow-up or long-term monitoring must be designed carefully, as the influence of matrix, number and location of sites, and sample collection and analysis methods can all impact the ability to identify trends in a dataset (see Chapter 9 Lessons Learned).

Insights based on the present study provide useful guidance for future study design to assess trends. For example, both monitoring and modeling of surface water microplastic levels (Chapter 4 Surface Water, Chapter 7 Transport Model) indicated the distribution of particles in this matrix is highly variable, and suggested a high number of sites would be needed to provide a fully representative quantification and characterization of microplastics in the Bay. This would likely be cost-prohibitive without identifying ways to decrease the amount of time and effort involved in sample collection and analysis, while still obtaining robust data on microplastics. It may instead be advisable to do follow-up monitoring of urban stormwater and wastewater pathways.

Additional study of microplastic retention by green stormwater infrastructure is also needed to explore the effects that implementation of larger and more varied landscaping solutions would have on concentrations of microplastics in urban stormwater runoff. A study of a Bay Area rain garden designed to remove regulated contaminants and metals showed that it was also effective at removing microplastics from stormwater (Gilbreath et al., 2019). Implementation of green stormwater infrastructure is underway in the Bay Area, often designed and built to reduce contamination from regulated pollutants. Evaluation of the co-benefits provided by these nature-based stormwater treatments as a trap for microplastics would fill a critical data gap to inform potential solutions.

Conclusions

With this chapter, we have presented a refined conceptual model of pathways of microplastic pollution specific to San Francisco Bay (Figure 8.1), which when combined with a comprehensive review of likely sources (Tables 8.1 and 8.2), can shed light on the contributions of urban stormwater runoff and treated wastewater discharges to the Bay and inform the prioritization of source reduction efforts. Cross-matrix synthesis and PCA indicated that despite the diversity of potential sources to urban stormwater (Table 8.2), the large contribution of black, rubbery fragments in Bay Area samples was the dominant feature. In contrast, wastewater appeared to be influenced by a broader array of sources, including plastics used in textiles (acrylic, polyester), as well as those used in microbeads and single-use items (polyethylene). Meanwhile, the ubiquitous detection of fibers in field and laboratory blanks is one line of evidence supporting future work to characterize a third potential pathway in the region: atmospheric deposition.

Integration of Bay and marine sanctuaries monitoring data with transport model predictions suggested the transport and fate of microplastics in the region is primarily driven by particle buoyancy, as dictated by polymer type and modified by morphology. Analysis of surface water and sediment data using a principal components approach indicated that polymer type was generally the most influential variable in determining whether relative contributions of different types of microplastics were preferentially concentrated in one matrix or the other. Particle morphology, particularly buoyant foams, also influenced the location of microplastics in the Bay. The predicted transport and fate of microplastics discharged to the Bay was heavily dependent on polymer type, with denser particles destined to be trapped within the Bay, while many more buoyant particles may be carried to the marine sanctuaries.

This study synthesis demonstrates that identification of plastic polymer type is essential to identifying potential sources of microplastics, as well as predicting their movement in estuarine waters. A major priority for future work in the greater scientific community is improving methods for polymer identification, including automation, improved spectral libraries, and approaches involving multiple lines of evidence where spectroscopy alone is inconclusive.

The study of microplastics is still in its infancy, and many challenges to developing a comprehensive understanding of this contaminant class must be overcome. Nevertheless, the San Francisco Bay Microplastics Project provides key insights and essential baseline data for the region. The science documented in this report can be used to inform solutions specific to the Bay Area; future monitoring to track the efficacy of informed management actions is recommended.

References

- Allen, K., Cohen, D., Culver, A., Cummins, A., Curtis, S., Eriksen, M., Gordon, M., Howe, A., Lapis, N., Prindiville, M., Thorpe, B., Wilson, S., 2017. Better Alternatives Now B.A.N. List 2.0. 5 Gyres, Algalita, Californians Against Waste, Clean Production Action, Plastic Pollution Coalition, Responsible Purchasing Network Story of Stuff, Surfrider Foundation, UPSTREAM.
- Almroth, B.M.C., Åström, L., Roslund, S., Petersson, H., Johansson, M., Persson, N.-K., 2018. Quantifying shedding of synthetic fibers from textiles: A source of microplastics released into the environment. Environ. Sci. Pollut. Res. 25, 1191–1199. <https://doi.org/10.1007/s11356-017-0528-7>
- Ballent, A., Corcoran, P.L., Madden, O., Helm, P.A., Longstaffe, F.J., 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. Mar. Pollut. Bull. 110, 383–395. <https://doi.org/10.1016/j.marpolbul.2016.06.037>
- Barboza, L.G.A., Vieira, L.R., Branco, V., Figueiredo, N., Carvalho, F., Carvalho, C., Guilhermino, L., 2018. Microplastics cause neurotoxicity, oxidative damage and energy-related changes and interact with the bioaccumulation of mercury in the European seabass, *Dicentrarchus labrax* (Linnaeus, 1758). Aquat. Toxicol. 195, 49–57. <https://doi.org/10.1016/j.aquatox.2017.12.008>
- Barnard, P.L., Schoellhamer, D.H., Jaffe, B.E., McKee, L.J., 2013. Sediment transport in the San Francisco Bay Coastal System: An overview. Mar. Geol., A multi-discipline approach for understanding sediment transport and geomorphic evolution in an estuarine-coastal system: San Francisco Bay 345, 3–17. <https://doi.org/10.1016/j.margeo.2013.04.005>
- Barrows, A.P.W., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab vs. neuston tow net: A microplastic sampling performance comparison and possible advances in the field. Analytical Methods 9, 1446–1453. <https://doi.org/10.1039/C6AY02387H>
- Berg, M., 2019. Global plastic pollution: the impact of ‘nurdles’ [WWW Document]. R. Soc. Chem. Environ. Chem. Group. URL <https://www.envchemgroup.com/impact-nurdles.html> (accessed 7.19.19).
- Boucher, J., Friot, D., 2017. Primary Microplastics in the Oceans: A Global Evaluation of Sources. Gland, Switzerland: International Union for Conservation of Nature (IUCN), 43pp.
- Brander, S., Renick, V., Foley, M., Steele, C., Woo, M., Lusher, A., Carr, S.A., Helm, P.A., Box, C., Cherniak, S., Andrews, R., Rochman, C.M., in review. Sampling and QA/QC, or how many blanks do I need? A guide for scientists investigating the occurrence of microplastics across matrices.
- Browne, M.A., Crump, P., Niven, S.J., Teuten, E., Tonkin, A., Galloway, T., Thompson, R., 2011. Accumulation of microplastic on shorelines worldwide: Sources and sinks. Environ. Sci.

Chapter 8—Synthesis

- Technol. 45, 9175–9179. <https://doi.org/10.1021/es201811s>
- Chang, M., 2015. Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. Marine Pollution Bulletin 101, 330–333. <https://doi.org/10.1016/j.marpolbul.2015.10.074>
- Chen, Q., Yin, D., Jia, Y., Schiwy, S., Legradi, J., Yang, S., Hollert, H., 2017. Enhanced uptake of BPA in the presence of nanoplastics can lead to neurotoxic effects in adult zebrafish. Sci. Total Environ. 609, 1312–1321. <https://doi.org/10.1016/j.scitotenv.2017.07.144>
- Conkle, J.L., Báez Del Valle, C.D., Turner, J.W., 2018. Are we underestimating microplastic contamination in aquatic environments? Environmental Management 61, 1–8. <https://doi.org/10.1007/s00267-017-0947-8>
- Correia Diogo, A., 2015. Polymers in Building and Construction, in: Gonçalves, M.C., Margarido, F. (Eds.), Materials for Construction and Civil Engineering: Science, Processing, and Design. Springer International Publishing, Cham, pp. 447–499. https://doi.org/10.1007/978-3-319-08236-3_10
- Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. Science of the Total Environment 667, 124–132. <https://doi.org/10.1016/j.scitotenv.2019.02.346>
- de Sá, L.C., Oliveira, M., Ribeiro, F., Rocha, T.L., Futter, M.N., 2018. Studies of the effects of microplastics on aquatic organisms: What do we know and where should we focus our efforts in the future? Sci. Total Environ. 645, 1029–1039. <https://doi.org/10.1016/j.scitotenv.2018.07.207>
- Downing-Kunz, M., Schoellhamer, D., Work, P., Work, P., 2017. Water and Suspended-Sediment Flux Measurements at the Golden Gate, 2016-2017. SFEI Contribution No. 856. San Francisco Estuary Institute, Richmond, CA.
- Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: A case study in Greater Paris. Environmental Chemistry 12, 592. <https://doi.org/10.1071/EN14167>
- Dris, R., Gasperi, J., Mirande, C., Mandin, C., Guerrouache, M., Langlois, V., Tassin, B., 2017. A first overview of textile fibers, including microplastics, in indoor and outdoor environments. Environ. Pollut. 221, 453–458. <https://doi.org/10.1016/j.envpol.2016.12.013>

Chapter 8—Synthesis

Dris, R., Gasperi, J., Tassin, B., 2018. Sources and Fate of Microplastics in Urban Areas: A Focus on Paris Megacity, in: Wagner, M., Lambert, S. (Eds.), Freshwater Microplastics : Emerging Environmental Contaminants? Springer International Publishing, Cham, pp. 69–83.
https://doi.org/10.1007/978-3-319-61615-5_4

Drumond Chequer, F.M., de Oliveira, G.A.R., Anastacio Ferraz, E.R., Carvalho, J., Boldrin Zanoni, M.V., de Oliveira, D.P., 2013. Textile Dyes: Dyeing Process and Environmental Impact, in: Gunay, M. (Ed.), Eco-Friendly Textile Dyeing and Finishing. InTech.
<https://doi.org/10.5772/53659>

Duis, K., Coors, A., 2016. Microplastics in the aquatic and terrestrial environment: Sources (with a specific focus on personal care products), fate and effects. Environ. Sci. Eur. 28.
<https://doi.org/10.1186/s12302-015-0069-y>

Edil, T.B., 2008. A review of environmental impacts and environmental applications of shredded scrap tires, in: Hazarika, H., Yasuhara, K. (Eds.), Scrap Tire Derived Geomaterials - Opportunities and Challenges. Taylor & Francis Group, London, pp. 3–18.

Elert, A.M., Becker, R., Duemichen, E., Eisentraut, P., Falkenhagen, J., Sturm, H., Braun, U., 2017. Comparison of different methods for MP detection: What can we learn from them, and why asking the right question before measurements matters? Environ. Pollut. 231, 1256–1264.
<https://doi.org/10.1016/j.envpol.2017.08.074>

Fahrenfeld, N.L., Arbuckle-Keil, G., Naderi Beni, N., Bartelt-Hunt, S.L., 2019. Source tracking microplastics in the freshwater environment. TrAC Trends in Analytical Chemistry 112, 248–254. <https://doi.org/10.1016/j.trac.2018.11.030>

Fendall, L.S., Sewell, M.A., 2009. Contributing to marine pollution by washing your face: Microplastics in facial cleansers. Marine Pollution Bulletin 58, 1225–1228.
<https://doi.org/10.1016/j.marpolbul.2009.04.025>

Galanty, W., 2012. Grinders, Shredders & Comminutors. Water Wastes Dig.

Gilbreath, A., McKee, L., Shimabuku, I., Lin, D., Werbowski, L.M., Zhu, X., Grbic, J., Rochman, C., 2019. Multiyear Water Quality Performance and Mass Accumulation of PCBs, Mercury, Methylmercury, Copper, and Microplastics in a Bioretention Rain Garden. J. Sustain. Water Built Environ. 5, 04019004. <https://doi.org/10.1061/JJSWBAY.0000883>

Gita, S., Hussan, A., Choudhury, T.G., 2017. Impact of textile dyes waste on aquatic environments and its treatment. Environment & Ecology 35, 2349—2353.

Gordon, M., 2006. Eliminating Land-based Discharges of Marine Debris in California: A Plan of Action from The Plastic Debris Project. State Water Resources Control Board.

Chapter 8—Synthesis

- Grigoratos, T., Martini, G., 2015. Brake wear particle emissions: A review. Environ. Sci. Pollut. Res. Int. 22, 2491–2504. <https://doi.org/10.1007/s11356-014-3696-8>
- Hann, S., Sherrington, C., Jamieson, O., Hickman, M., Kershaw, P., Bapasola, A., Cole, G., 2018. Investigating options for reducing releases in the aquatic environment of microplastics emitted by (but not intentionally added in) products Final Report. DG Environment of the European Commission.
- Hartline, N.L., Bruce, N.J., Karba, S.N., Ruff, E.O., Sonar, S.U., Holden, P.A., 2016. Microfiber masses recovered from conventional machine washing of new or aged garments. Environ. Sci. Technol. 50, 11532–11538. <https://doi.org/10.1021/acs.est.6b03045>
- Hassaan, M.A., Nemr, A.E., 2017. Health and environmental impacts of dyes: Mini review. American Journal of Environmental Science and Engineering 1, 64–67. <https://doi.org/10.11648/j.ajese.20170103.11>
- He, P., Chen, L., Shao, L., Zhang, H., Lü, F., 2019. Municipal solid waste (MSW) landfill: A source of microplastics? -Evidence of microplastics in landfill leachate. Water Res. 159, 38–45. <https://doi.org/10.1016/j.watres.2019.04.060>
- Henry, B., Laitala, K., Klepp, I.G., 2019. Microfibres from apparel and home textiles: Prospects for including microplastics in environmental sustainability assessment. Sci. Total Environ. 652, 483–494. <https://doi.org/10.1016/j.scitotenv.2018.10.166>
- Hernandez, E., Nowack, B., Mitrano, D.M., 2017. Polyester textiles as a source of microplastics from households: A mechanistic study to understand microfiber release during washing. Environ. Sci. Technol. 51, 7036–7046. <https://doi.org/10.1021/acs.est.7b01750>
- Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: A review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075. <https://doi.org/10.1021/es2031505>
- Jabeen, K., Li, B., Chen, Q., Su, L., Wu, C., Hollert, H., Shi, H., 2018. Effects of virgin microplastics on goldfish (*Carassius auratus*). Chemosphere 213, 323–332. <https://doi.org/10.1016/j.chemosphere.2018.09.031>
- Kang, J.-H., Kwon, O.Y., Lee, K.-W., Song, Y.K., Shim, W.J., 2015. Marine neustonic microplastics around the southeastern coast of Korea. Marine Pollution Bulletin 96, 304–312. <https://doi.org/10.1016/j.marpolbul.2015.04.054>
- Karlsson, T.M., Arneborg, L., Broström, G., Almroth, B.C., Gipperth, L., Hassellöv, M., 2018. The unaccountability case of plastic pellet pollution. Mar. Pollut. Bull. 129, 52–60. <https://doi.org/10.1016/j.marpolbul.2018.01.041>

Chapter 8—Synthesis

- Kawecki, D., Nowack, B., 2019. Polymer-Specific Modeling of the environmental emissions of seven commodity plastics as macro- and microplastics. *Environ. Sci. Technol.* 53, 9664–9676. <https://doi.org/10.1021/acs.est.9b02900>
- Kole, P.J., Löhr, A.J., Van Belleghem, F., Ragas, A., 2017. Wear and tear of tyres: A stealthy source of microplastics in the environment. *Int. J. Environ. Res. Public. Health* 14, 1265. <https://doi.org/10.3390/ijerph14101265>
- Lassen, C., Hansen, S.F., Magnusson, K., Norén, F., Hartmann, N.B., Jensen, P.R., Nielsen, T.G., Brinch, A., 2015. Microplastics: Occurrence, effects and sources of releases to the environment in Denmark (No. 1793, 2015). The Danish Environmental Protection Agency Ministry of Environment and Food.
- Lê, S., Josse, J., Husson, F., 2008. FactoMineR: An R Package for Multivariate Analysis. *J. Stat. Softw.* 25, 1–18. <https://doi.org/10.18637/jss.v025.i01>
- Liu, C., Li, J., Zhang, Y., Wang, L., Deng, J., Gao, Y., Yu, L., Zhang, J., Sun, H., 2019. Widespread distribution of PET and PC microplastics in dust in urban China and their estimated human exposure. *Environ. Int.* 128, 116–124. <https://doi.org/10.1016/j.envint.2019.04.024>
- Löhr, A., Savelli, H., Beunen, R., Kalz, M., Ragas, A., Van Belleghem, F., 2017. Solutions for global marine litter pollution. *Curr. Opin. Environ. Sustain., Sustainability governance* 28, 90–99. <https://doi.org/10.1016/j.cosust.2017.08.009>
- Mani, T., Blarer, P., Storck, F.R., Pittroff, M., Wernicke, T., Burkhardt-Holm, P., 2019. Repeated detection of polystyrene microbeads in the Lower Rhine River. *Environ. Pollut. Barking Essex* 1987 245, 634–641. <https://doi.org/10.1016/j.envpol.2018.11.036>
- McKeen, L.W., 2013. 1 - Introduction to Use of Plastics in Food Packaging, in: Ebnesajjad, S. (Ed.), *Plastic Films in Food Packaging, Plastics Design Library*. William Andrew Publishing, Oxford, pp. 1–15. <https://doi.org/10.1016/B978-1-4557-3112-1.00001-6>
- Mourkgogiannis, N., Kalavrouziotis, I.K., Karapanagioti, H.K., 2018. Questionnaire-based survey to managers of 101 wastewater treatment plants in Greece confirms their potential as plastic marine litter sources. *Mar. Pollut. Bull.* 133, 822–827. <https://doi.org/10.1016/j.marpolbul.2018.06.044>
- Munno, K., De Frond, H., O'Donnell, B., Rochman, C., in review. Increasing the accessibility for characterizing microplastics: Introducing new application-based and spectral libraries of plastic particles (SLoPP & SLoPP-E).
- Nelms, S.E., Galloway, T.S., Godley, B.J., Jarvis, D.S., Lindeque, P.K., 2018. Investigating microplastic trophic transfer in marine top predators. *Environmental Pollution* 238, 999–1007. <https://doi.org/10.1016/j.envpol.2018.02.016>

Chapter 8—Synthesis

- Pannetier, P., Cachot, J., Clérandeau, C., Faure, F., Van Arkel, K., de Alencastro, L.F., Levasseur, C., Sciacca, F., Bourgeois, J.-P., Morin, B., 2019a. Toxicity assessment of pollutants sorbed on environmental sample microplastics collected on beaches: Part I-adverse effects on fish cell line. Environ. Pollut. 248, 1088–1097. <https://doi.org/10.1016/j.envpol.2018.12.091>
- Pannetier, P., Morin, B., Clérandeau, C., Laurent, J., Chapelle, C., Cachot, J., 2019b. Toxicity assessment of pollutants sorbed on environmental microplastics collected on beaches: Part II-adverse effects on Japanese medaka early life stages. Environ. Pollut. 248, 1098–1107. <https://doi.org/10.1016/j.envpol.2018.10.129>
- Pantoja Munoz, L., Gonzalez Baez, A., McKinney, D., Garelick, H., 2018. Characterisation of “flushable” and “non-flushable” commercial wet wipes using microRaman, FTIR spectroscopy and fluorescence microscopy: To flush or not to flush. Environ. Sci. Pollut. Res. Int. 25, 20268–20279. <https://doi.org/10.1007/s11356-018-2400-9>
- Pirc, U., Vidmar, M., Mozer, A., Kržan, A., 2016. Emissions of microplastic fibers from microfiber fleece during domestic washing. Environ. Sci. Pollut. Res. 23, 22206–22211. <https://doi.org/10.1007/s11356-016-7703-0>
- Rainieri, S., Conlledo, N., Larsen, B.K., Granby, K., Barranco, A., 2018. Combined effects of microplastics and chemical contaminants on the organ toxicity of zebrafish (*Danio rerio*). Environ. Res. 162, 135–143. <https://doi.org/10.1016/j.envres.2017.12.019>
- Rochman, C., Kross, S., Bogan, M., Darling, E., Green, S., Veríssimo, D., Smyth, A., Armstrong, J., 2015. Scientific Evidence Supports a Ban on Microbeads. Society for Conservation Biology.
- Rochman, C.M., Kross, S.M., Armstrong, J.B., Bogan, M.T., Darling, E.S., Green, S.J., Smyth, A.R., Veríssimo, D., 2015. Correction to Scientific Evidence Supports a Ban on Microbeads. Environmental Science & Technology 49, 14740–14740. <https://doi.org/10.1021/acs.est.5b05043>
- Scudo, A., Liebmann, B., Corden, C., Tyrer, D., Kreissig, J., Warwick, O., 2017. Intentionally added microplastics in products (No. 39168 Final Report 17271i3). Amec Foster Wheeler Environment & Infrastructure UK Limited.
- Sedlak, M., Sutton, R., Box, C., Sun, J., Lin, D., 2017. FINAL Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries. SFEI Contribution No. 819. San Francisco Estuary Institute and 5 Gyres, Richmond, CA.
- Slaughter, E., Gersberg, R.M., Watanabe, K., Rudolph, J., Stransky, C., Novotny, T.E., 2011. Toxicity of cigarette butts, and their chemical components, to marine and freshwater fish. Tob. Control 20, i25–i29. <https://doi.org/10.1136/tc.2010.040170>
- Sommer, F., Dietze, V., Baum, A., Sauer, J., Gilge, S., Maschowski, C., Gieré, R., 2018. Tire

abrasion as a major Source of microplastics in the environment. *Aerosol Air Qual. Res.* 18, 2014–2028. <https://doi.org/10.4209/aaqr.2018.03.0099>

Sutton, R., Sedlak, M., 2017. Microplastic Monitoring and Science Strategy for San Francisco Bay. SFEI Contribution No. 798. San Francisco Estuary Institute, Richmond, CA.

Vermeiren, P., Muñoz, C.C., Ikejima, K., 2016. Sources and sinks of plastic debris in estuaries: A conceptual model integrating biological, physical and chemical distribution mechanisms. *Mar. Pollut. Bull.* 113, 7–16. <https://doi.org/10.1016/j.marpolbul.2016.10.002>

Verschoor, A., de Poorter, L., Dröge, R., Kuenen, J., de Valk, E., 2016. Emission of microplastics and potential mitigation measures: Abrasive cleaning agents, paints and tyre wear (RIVM Report 2016-0026). Dutch National Institute for Public Health and the Environment (RIVM).

Vogelsang, C., Lusher, A.L., Dadkhah, M.E., Sundvor, I., Umar, M., Ranneklev, S.B., Eidsvoll, D., Meland, S., 2019. Microplastics in road dust – Characteristics, pathways and measures (No. 7361–2019). Norwegian Institute for Water Research.

Wen, B., Jin, S.-R., Chen, Z.-Z., Gao, J.-Z., Liu, Y.-N., Liu, J.-H., Feng, X.-S., 2018. Single and combined effects of microplastics and cadmium on the cadmium accumulation, antioxidant defence and innate immunity of the discus fish (*Sympodus aequifasciatus*). *Environ. Pollut.* 243, 462–471. <https://doi.org/10.1016/j.envpol.2018.09.029>

Zhang, S., Ding, J., Razanajatovo, R.M., Jiang, H., Zou, H., Zhu, W., 2019. Interactive effects of polystyrene microplastics and roxithromycin on bioaccumulation and biochemical status in the freshwater fish red tilapia (*Oreochromis niloticus*). *Sci. Total Environ.* 648, 1431–1439. <https://doi.org/10.1016/j.scitotenv.2018.08.266>

Zhu, X., Nguyen, B., You, J.B., Karakolis, E., Sinton, D., Rochman, C., 2019. Identification of microfibers in the environment using multiple lines of evidence. *Environ. Sci. Technol.* <https://doi.org/10.1021/acs.est.9b05262>

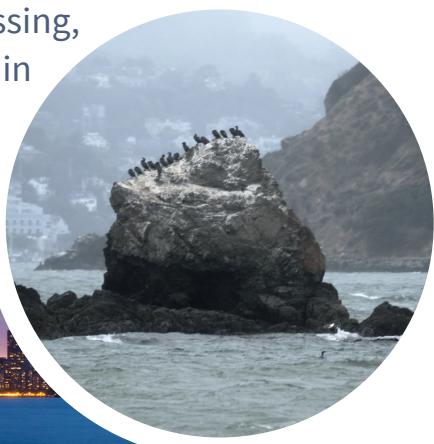
CHAPTER

9

by Liz Miller &
Meg Sedlak

LESSONS LEARNED:

Recommended Best Practices for Collecting, Processing,
Analyzing, and Reporting Microplastics in
Environmental Media



Introduction

The Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) is a world leader in water quality monitoring, demonstrating how regional collaboration can provide the science needed to protect and improve water quality in a treasured ecosystem. San Francisco Bay (the Bay) is critical habitat for a multitude of estuarine species and drains approximately 40% of the land area of California. It is surrounded by a dense urban population with a myriad of commercial and industrial activities. Because of its hydrodynamics, the Bay can act as a long-term trap for persistent contaminants, with recovery taking decades or longer when contamination is extensive and persistent (e.g., mercury and polychlorinated biphenyls). The Bay is therefore a prime example of an ecosystem that merits investigation of the scope of contamination and potential impacts of anthropogenic contaminants.

The study of microplastics is a burgeoning field with a recent exponential increase in technical reports and scientific articles and growing interest from the public and policy makers. However, relatively few studies have incorporated rigorous study designs, particularly quality assurance and quality control (QA/QC) measures that are emblematic of more established fields of trace chemical contaminant monitoring (European Chemicals Agency, 2019a, 2019b; Lares et al., 2019; Science Advice for Policy by European Academies, 2019).

This chapter presents a discussion of best practices based on a recently completed comprehensive assessment of microplastics in San Francisco Bay water, sediment, fish, stormwater, and wastewater effluent. We also evaluated microplastics in surface water in three National Marine Sanctuaries hydrologically linked to the Bay.

This case study is instructive, as there is an urgent and critical need to develop standard methods and QA/QC practices in the field of microplastics. Standard methods allow comparison among studies to identify areas of concern and trends that can inform policy and management actions. It is imperative to use appropriate QA/QC methods to ensure microplastic measurements reported for environmental samples are accurate and not significantly influenced by background contamination in the field or laboratory. Implementation of QA/QC measures also allows researchers to assess variation in microplastic analyses to determine whether the differences observed in the field are statistically significant or merely a reflection of the variation in collection and analysis. Development and implementation of standard methods and QA/QC procedures will allow management actions to be evaluated to ensure they are being implemented effectively and in areas where they will make the most difference.

Study objectives

The overall objective for the San Francisco Bay Microplastics Project was to improve understanding of microplastics in the Bay, with a focus on source identification, estimation of relative loads, and occurrence in the Bay and sanctuaries, to inform policy and management actions. An additional goal was to further work in the field by implementing a rigorous study design and developing and refining field and analytical methods that can be applied broadly.

Specific objectives and corresponding scientific needs are listed in Table 9.1. We used these objectives to develop the study design. Below we outline the study design elements that we considered and how we used these objectives to drive our decisions.

Table 9.1. Specific study objectives that drove site selection and method considerations.

Study Objective	Scientific Needs to Adequately Address Objective		
	Site Selection	Field Sampling	Laboratory Analysis
Conduct a comprehensive baseline evaluation of microplastics in Bay and adjacent National Marine Sanctuaries surface water, as well as Bay sediment and Bay fish	Represent the overall condition of the Bay and sanctuaries (e.g., sufficient number of sites) Represent spatial variation (e.g., mid-Bay and near-shore or “margins” sample sites); include a less urban reference site	Represent current conditions (e.g., surface sediments instead of sediment cores) Collect samples during wet and dry seasons to evaluate seasonal influence Use standard sieve mesh sizes to collect particles within comparable operational size categories for comparison among matrices and future trend assessment Employ standardized field methods and collect QA/QC samples	Microparticle analysis must include polymer identification (source attribution and tracking trends) Use standardized laboratory methods with good QA/QC Minimize chemical digestion methods when possible Bulk water samples may require larger volumes to exceed blank contamination thresholds Bulk water samples are more appropriate for sampling fibers than manta trawls
Characterize microplastics in stormwater runoff and treated wastewater effluent discharges to the Bay to develop a baseline, assess relative loads, and identify unique sources	Select higher-flow sites (e.g., larger streams or wastewater treatment facilities) For wastewater, select facilities with different treatment types For stormwater, select sites with varying land-use patterns to allow extrapolation of loads	Focus on larger particles (i.e., not nanoparticles) that are more easily tied back to sources Use standard sieve mesh sizes for comparison among different pathways; 125 µm mesh is needed for capture of microbeads from wastewater For wastewater, avoid the weekend effect by sampling only Tuesday–Friday For stormwater, develop storm criteria to ensure consistency Employ standardized field methods and collect QA/QC samples	Microparticle analysis must include polymer identification (source attribution, trends) Consider using pyrolysis GC-MS to identify polymer if monitoring urban areas that will have tire wear particles Use consistent sieves across matrices to facilitate evaluation of transport Use standardized laboratory methods with good QA/QC

Study Objective	Scientific Needs to Adequately Address Objective		
	Site Selection	Field Sampling	Laboratory Analysis
Assess uptake of microplastics into the food web and identify areas of high concern for biota	<p>Select spatially diverse sampling sites, including a less urban reference site</p> <p>Select species with high site fidelity and known feeding habits and predation</p> <p>Sample multiple matrices at the same site (sediment, water, and biota)</p> <p>Determine appropriate backup sites should biota not be found at the original site of interest</p>	<p>Target sufficient numbers of individuals to determine a representative level of contamination in biota</p> <p>Collect biota using standardized field methods, including collection of individuals of similar size and weight</p>	<p>Microparticle analysis should include polymer identification</p> <p>Use standardized laboratory methods with good QA/QC</p> <p>Focus on the full gastrointestinal tract to assess ingestion; digest whole gut (rather than just rinse gut lumen) to capture entrenched particles</p> <p>Assess smaller size fractions (less than 150 µm); these particles have the potential to translocate out of the gut and bioaccumulate</p>
Develop an estuarine-marine microplastic transport model linking the transport of microplastics from the Bay out the Golden Gate to adjacent National Marine Sanctuaries	<p>Represent the overall condition of the Bay and Sanctuaries</p> <p>Coordinate with model needs (e.g., locate some sample sites near model boundary conditions)</p> <p>Sample multiple matrices at the same site (e.g., sediment and water)</p>	<p>Represent current conditions (e.g., surface sediments instead of sediment cores)</p> <p>Consider tidal fronts and currents; water dynamics can concentrate buoyant and semi-buoyant material, causing significant small-scale spatial variation</p> <p>Convergence zones tend to aggregate biological nutrients and microplastics; consider whether to incorporate them based on objectives</p> <p>Focus on larger particles (i.e., not nanoparticles) that are more easily tied to sources</p>	<p>Microparticle analysis must include polymer identification (to estimate buoyancy)</p> <p>Use standardized laboratory methods with good QA/QC</p> <p>Minimize chemical digestion methods when possible</p> <p>Bulk water samples may require larger volumes to exceed blank contamination thresholds</p> <p>Bulk water samples are more appropriate for sampling fibers than manta trawls</p>

Study design: Site selection

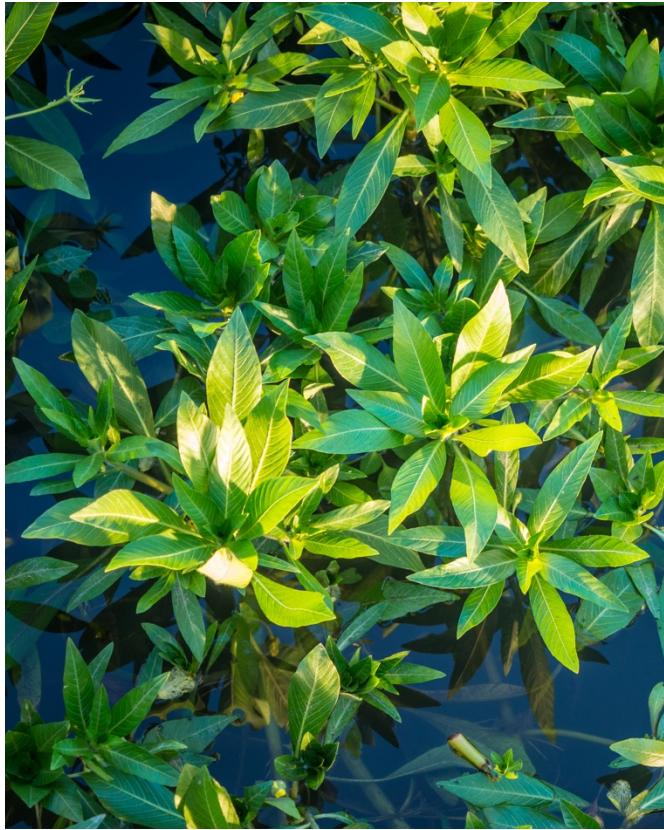
General considerations for site selection

The design and site selection of any study should be based on the questions to be answered and the statistical tests required to demonstrate differences. For example, answering the questions "what is the average concentration in all of San Francisco Bay?" and "what is the average concentration in major segments of the Bay?" would necessitate representative and sufficient sampling of the whole Bay and of each segment of the Bay, respectively. In contrast, addressing the question "are there significant differences between segments of the Bay?" would necessitate representative sampling and enough replication to provide the statistical power for doing an analysis of variance (ANOVA). Sampling design, statistical frameworks, and power analyses are important to consider, but are not our main focus here.

If the focus is on trend monitoring, a design with carefully located sites, and controlling for season, tidal stage, current, and other spatial and temporal factors expected to influence concentrations may be best. For this study, one of the primary objectives was to establish a baseline data set to provide a solid basis for future monitoring. We wished to be able to detect differences among sites and over time, and the resulting data set provides a starting point for power analyses to design sampling schemes that could potentially provide sufficient statistical power to detect differences of specified magnitudes among sites and over time.

The geography, hydrology, and bathymetry of the water body, as well as potential sources (e.g., outfall or discharge points) should be considered as part of the spatial design. In order to adequately capture spatial variation in contaminants caused by these factors, larger study areas should be subdivided into regions defined by their differences. For example, via the technical leadership of the RMP, the Bay has been divided into five subembayments for monitoring based on geography, hydrodynamics, best professional judgement, and management needs (Lowe et al., 2005).

In general, sites should be distributed throughout each region, and the number of sites should be sufficient to provide statistical power to achieve study objectives. In this study, the National Marine Sanctuary sites were selected to evaluate conditions in surface waters above the continental shelf that extends several kilometers from shore, as well as above the continental slope that drops steeply into the deep ocean. In the Bay, sites were chosen to range from the deeper open Bay to the shallow margins. This information was important to characterize conditions across the sanctuaries and Bay, as well as to provide input for model development. In addition, variation in physical attributes can also influence the uptake of microplastics into the food web. For example, shallow margin sites in the Bay are prime nursery and breeding habitat; they are also quiescent areas where particles are likely to deposit.



The selection of sites in the open ocean and shallow margins meant it was necessary to use multiple research vessels: one that could handle open ocean swells and currents and accommodate crew overnight for multi-day cruises, as well as a smaller vessel with a low draft to access shallow sites. Shallow sites require careful coordination with tidal cycles to avoid vessel stranding.

Desirable sites may be in areas that require additional planning. For example, surface water sites may be in or near restricted zones around airports or in logistically challenging areas, such as active shipping lanes or ferry transit corridors. Stormwater sites may also require special permits or approval to sample, or present access challenges due to the site location (e.g., busy highway bridges). Wastewater sample collection requires approval and support from the wastewater treatment facility leadership and staff. Scientific

collection permits, access permits, and approval to sample parks, protected areas, or other locations can take several months to obtain.

Where possible and appropriate for addressing the study objectives, inclusion of a reference site can provide valuable context for understanding the degree of contamination in the primary study area. Figure 9.1 shows the study area, sites, and matrices sampled in San Francisco Bay and Tomales Bay, which was the reference site for this study.



Figure 9.1. Study sites for the San Francisco Bay Microplastics Project (additional surface water sites were located in the adjacent National Marine Sanctuaries; see Chapter 4 Surface Water for ocean site locations). Trawl samples include surface water trawls performed during wet and dry seasons. Only watersheds sampled for stormwater are pictured.

Matrix-specific considerations for site selection

SURFACE WATER

Hydrology is an important consideration for surface water site selection. For example, this study required more sites in the Central Bay region because this is a convergence area from the north and south embayments and receives tidal inflow from the open ocean. Thus, additional data were needed for spatial characterization and model development, particularly since the Central Bay was a boundary condition for the interface of the Bay model to the open ocean model.

For studies focused on source identification, site selection should also consider the influence of possible pathways such as wastewater or stormwater. For example, we preferentially chose surface water sites along the margins of the Bay due to their proximity to pathways such as stormwater outfalls and wastewater discharges.

One particular consideration is that tides, currents, and wind can concentrate buoyant and semi-buoyant material in oceanic and estuarine fluid dynamic features such as tidal fronts, windrows, and eddies (Welden and Lusher, 2017). In the present study, at least one instance of sampling occurred in which the manta trawl passed through a prominent tidal front halfway through sample collection. Vegetation, woody debris, trash, and plastic fragments were observed floating at the surface and were captured in the sample. This sample had the highest microparticle abundance recorded within the study, and could be indicative of the microparticle levels found in these biologically important fronts within the Bay and the nearshore marine sanctuaries. If a feature like this is unexpectedly encountered (and noticed), and if resources allow, it would be ideal to have samples within the feature, as well as and nearby but outside of the feature, to assess its potential impact on study observations.

SEDIMENT

It is important to consider whether the sediment sites are located in areas that are depositional or erosional, and if there are unique attributes that could impact microplastic levels locally, such as known trash hot spots, or areas of active dredging or deposition of dredged material. If there are study components examining cross-matrix interactions and dependencies, sites should be located and distributed to complement the sampling plans for other matrices to the extent possible.

In this study, sediment sampling sites were chosen to characterize ambient conditions and the influence of potential pathways for microplastic transport. Therefore, most of the Bay sediment sites were located in the nearshore Bay margins. These areas are nearest to and likely most affected by stormwater runoff from urban creeks and shallow wastewater discharges (the

latter primarily located in Lower South Bay) and are frequently depositional rather than erosional. Sediment sites were also co-located with fish sampling sites to facilitate the study of bioavailability. None of the sites were impacted by dredging.

FISH

Fish sampling locations are predicated on the targeted species and their ecological niches. In contrast to surface water and sediment sampling, it is important to have flexibility in the study design in the event that no fish are present at the original target site.

Species should be selected based on the study objectives. Studies that are evaluating potential exposure to humans should target sport fish, while studies characterizing food web transfer or wildlife exposure can target lower trophic level prey fish. In this study, we chose to monitor two species of prey fish, anchovy (*Engraulis mordax*) and topsmelt (*Atherinops affinis*), to assess microplastics entering the Bay food web as a result of differing foraging strategies. Prey fish can serve as indicators of the bioavailability of microplastics, as they are an important food source for piscivorous fish, birds, and marine mammals.

The feeding habits of the chosen species may also influence the design of accompanying water and sediment sampling. Benthic feeders may more readily ingest microplastics in sediment as they forage, while pelagic feeding species are less likely to encounter this exposure route. Concentrations of contaminants in sediment can correlate with concentrations in benthic feeding fish, as has been demonstrated for polychlorinated biphenyls in Bay prey fish (Greenfield and Allen, 2013). We chose two prey fish species with different foraging habits to explore whether this would also be true for microplastics.

The home range of the species is also important; organisms with high site fidelity are better for comparing to other environmental matrices, as co-located sampling will more accurately represent exposure. For this study, another reason we chose prey fish is that they exhibit high site fidelity relative to other fish species, allowing for potential identification of areas of higher contamination and concern.

STORMWATER

To assess potential sources, stormwater site selection should account for the land-use characteristics in the watersheds of interest. In this study, including a wide range of land-use types such as agricultural, commercial, industrial, and transportation helped to assess whether certain land uses (that could potentially be used to tie microplastics to their sources) were associated with the generation of more microplastics.

To develop models, it is important to understand relative loads, and therefore the study design should include a sufficient number of watersheds and span the range of expected variation to

characterize the pathway as a whole. In aggregate, the watersheds sampled in this study comprised 11% of the total small tributary watershed area draining into the Bay, with total urban area within the watersheds ranging from 9%–98%.

WASTEWATER

To provide robust estimates of loads, it is important to capture the major wastewater discharges. In this study, eight wastewater treatment facilities were selected, including many of the largest dischargers in the Bay Area. These eight facilities represent approximately 70% of the total flow of effluent to San Francisco Bay, or approximately 887 million cubic meters of treated wastewater per year (San Francisco Bay Regional Water Quality Control Board - Region 2, 2019).

Different treatment methods are used in wastewater treatment facilities to process influent; the choice of method is a function of a number of factors including the composition of the influent, flow, cost, and discharge permit requirements. Understanding the potential influence of different treatment types requires sampling at facilities that employ different processes. In this study, four of the selected facilities employed secondary treatment (biological treatment), and four employed additional tertiary treatment that included dual media filtration as a finishing step. In addition, one of the secondary facilities treated a combined flow of stormwater and wastewater influent, allowing for insights on the influence of stormwater.

REFERENCE SITES

A good reference site for urban microplastic monitoring is a location that is similar to the study site (i.e., similar climate, geography, geology, and hydrology) but is rural and undeveloped. The surrounding watershed should be largely in pristine condition with a limited area of impervious surfaces and little transportation infrastructure. Sites that have wastewater and stormwater outfalls should be avoided. We selected Tomales Bay as our reference site, which is located 45 km north of San Francisco Bay and is bordered on one side by a national park and on the other side by a low population density, rural area. Past examination of other urban contaminants there have shown lower concentrations in biota relative to the more urbanized Bay (e.g., Sedlak and Greig, 2012).

A reference site is particularly important for biota so differences in contamination due to relevant site conditions (e.g., urbanization) can be identified. Our two targeted species of fish reside in both San Francisco Bay and Tomales Bay, the reference site.

Study design: Field sampling

General considerations for selecting field methods

The selection of sample collection methods is critical and will dramatically influence the study results. With this in mind, it is imperative to think about the questions the study is being designed to answer and how the data will be used to answer these questions.

There are four key questions regarding collection methods that must be considered.

What is the lowest size fraction of interest and why?

In general, the smaller the particle size of interest, the more particles will be collected and will need to be analyzed for a given volume of sample (Covernton et al., 2019). For example, microparticle concentrations reported in wastewater in the literature range four orders of magnitude from the largest size fractions to the smallest (this report Figure 3.9; Leslie et al., 2017; Magnusson and Norén, 2014; Simon et al., 2018; Wolff et al., 2019). Smaller particles can be harder to trace to sources, so if the goal of the study is to investigate sources, it may not be necessary to use less than a 125 µm screen sieve.

How will the data be used relative to other datasets?

If the goal is to compare among matrices (e.g., to assess the relative importance of loads or food web accumulation), it is critical to standardize the mesh size of sieves across matrices to facilitate comparisons via consistent operational size categories. If the goal is to compare to past measurements or those reported in the literature, methods consistent with prior studies or literature studies of interest should be used.

What type of particle morphology is of interest and why?

Fibers are the most challenging morphology to collect and analyze due to their elongated shape and thin diameter. Depending on the orientation of the fiber, it may or may not be caught by a net or sieve, which can result in significant undercounts (Barrows et al., 2017; Covernton et al., 2019). This can make comparisons among results from different sampling procedures challenging. If the goal of the study design is to accurately capture and quantitatively assess fibers, it is more appropriate to use bulk grab samples rather than net sampling methods. Filters with pore sizes small enough to prevent fiber passage should also be used.

How much volume to sample?

Large sample volumes are often desirable because they are less affected by the heterogeneity of the matrix, and therefore are more likely to be representative. In addition, a larger volume may be necessary to overcome background levels of contamination. However, given the labor intensity of sample extraction and analysis, the smallest sample size should be collected; otherwise subsampling may be necessary.

Matrix-specific considerations for field sampling

SURFACE WATER

Water samples from oceans and lakes are typically collected within one meter of the water surface, rather than at deeper depths or depth-integrated. For example, the manta trawl net is designed to skim the surface to a maximum depth of 16 cm. Relatively few studies have explored the vertical profile of microparticles with depth or sampled significantly beneath the surface water (Choy et al., 2019; Reisser et al., 2013).



Surface water samples are most frequently collected using manta trawls or Neuston nets, although pump systems or grab samples have also been employed (Barrows et al., 2017; Eriksen et al., 2013; Miller et al., 2017; Prata et al., 2019). Towed for 30 minutes at slow speeds, manta trawls and Neuston nets provide an integrated sample representing a large quantity of surface water (Eriksen et al., 2013; Mason et al., 2016).

Pump and grab samples require far less effort and smaller sample volumes; however, due to their smaller volume, they are more easily influenced by heterogeneity and so may be less representative of the surrounding surface conditions. Comparison between Neuston nets and grab samples showed that grab sampling collected over three orders of magnitude more microplastics per volume of water, as well as a smaller size range than sampling with a Neuston net (Barrows et al., 2017). Grab samples are able to characterize all morphologies, can capture smaller particles, and can be used in environments where nets are impractical, but the small volume of water sampled may result in high variability among samples (Dubaish and Liebezeit, 2013). We explored using a one-liter grab; however, based on

our results, a larger grab sample volume was needed to overcome background contamination. It is recommended that a larger sample that is at least two to four liters be used (Brander et al., in review).

For baseline characterization, replicate samples (same time/place) are valuable to assess general background variability. Potential seasonal differences in surface water concentrations should be considered. In our study, we collected samples during dry and wet seasons, and observed statistically significantly higher concentrations following wet weather events, which were defined as 1.3 cm rainfall within 24 hours (see Chapter 4 Surface Water).

SEDIMENT

Sediment samples are collected using a variety of devices: a sediment grab device (such as a Van Veen or Ponar grab), a bed sediment trap, or a coring device (Ballent et al., 2016; Prata et al., 2019). The depth of the sediment to be sampled is an important consideration, with shallow samples more likely to represent more recent conditions. Sediment cores can give an indication of trends over time, provided the area has remained depositional over time and the sediment has not been significantly disturbed.

The European Union in the Marine Strategy Framework Directive recommends that surface sediment samples be collected from the top 5 cm of sediment (European Commission Joint Research Center, 2013); this is also consistent with the RMP’s protocol for monitoring Bay surface sediment (Yee et al., 2018). In this study, a stainless steel scoop was used to sample the top 5 cm of sediment in the grab, taking care to avoid the sides of the grab, and was deposited directly into a clean glass sampling jar. In the Bay, the surface (top 5 cm) subtidal sediment layer primarily includes sediment from the past few decades, with extensive but heterogeneous mixing in many areas due to bioturbation and abiotic processes (Fuller et al., 1999; Yee et al., 2011). Estimating the age of surface sediment is difficult due to mixing and transport of sediment from adjoining margin areas and Bay segments.

FISH

A variety of techniques can be used to sample fish; the selection of the method is a function of the targeted fish species and the habitat in which they reside. Fish of standardized weight and length should be targeted to facilitate comparisons among fish and locations and to reduce variation due to external factors such as age.

Due to the wide variation in biological samples, it is important to include sufficient fish for each site to enable statistical analyses. A sample size of at least 50 individuals per research unit (species, food web, ecoregion, feeding type, etc.) has been recommended, but this number is somewhat arbitrary (Hermsen et al., 2018). If ingestion incidence appears to be low, larger

sample sizes will be needed to give reliable results, whereas smaller sample sizes may be sufficient for populations with a high incidence of microplastics ingestion. In this study, we collected ten of each species at each site and analyzed them individually. Analyzing samples of individuals from each site provided us with more granular data by which we could assess statistical power to detect differences among sites. Composite sampling would have resulted in fewer analyses and provided a more integrative measure; however, it would have also decreased our ability to detect differences between sites and species, and would have required many more fish samples to obtain a sufficient number of composites to assess statistical differences.

STORMWATER

Studies of microplastics in streams and stormwater have been scarce; samples have been collected using manta nets, handnets, pump systems, or discrete bulk water grabs (Gilbreath et al., 2019; Moore et al., 2011). Using a net or pump system measures the average over time, whereas a grab sample gives a snapshot of a moment during the storm.

If a goal for the study design is determining loads, we recommend collecting depth-integrated samples. It is possible that depth-integrated sampling could underestimate the lighter weight microplastics floating on the surface, which can comprise a disproportionate amount of the overall composition of microplastics in some settings (Lattin et al., 2004). However, the turbulent nature of storm-driven stream flows may cause considerable mixing of these more buoyant particles within the water column, suggesting depth-integrated sampling during storm events may be more representative.

At a single site in our study, we sampled using an alternative method more likely to capture microplastics at the surface of the stream (using an 11 L stainless steel pail). The results suggested that this method may underestimate microparticles, supporting the use of depth-integrated samples for collecting microplastic measurements to estimate loads.

It is also important to consider the influence of seasonality and storm-related transport. Research suggests significant increases in microplastic loading during wet weather events (Moore et al., 2011). With a Mediterranean climate, Bay Area rainfall largely occurs November through April. Because 95% of the flow in Bay Area small tributaries is the direct result of rainfall (McKee et al., 2003), and dry weather sample collection may be less important for load calculations, we focused our tributary sampling during rainfall events that occur during the winter. Most work in streams to date has been dry weather sampling (Kataoka et al., 2019; Xiong et al., 2019), but the results from this study indicate stormwater delivers relatively large loads of microplastics during wet weather.

We recommend establishing a storm threshold level for sampling prior to starting a study. We chose to collect samples during storms that were predicted to have more than 1.3 cm of rain within six hours (Bay Area Stormwater Management Agencies Associations (BASMAA), 2016; Sedlak et al., 2017). Based on prior studies of legacy contaminants in Bay Area watersheds, these conditions are sufficiently intense to mobilize small particles from the watershed (Gilbreath and McKee, 2015); storms forecasted for shorter duration and smaller magnitude often result in storms that lead to little runoff. This threshold will likely be site-specific; for the Los Angeles Basin a criteria of 0.6 cm over 24 hours was used to define a wet weather event (Moore et al., 2011). For areas that receive regular rainfall, the time between wet weather events (e.g., number of days without rain) may also need to be specified. In the Bay Area, rainfall is so sporadic that this consideration was unnecessary.

We recommend being on-site at the beginning of the storm to sample the start of storm-related flow, as pollutants can be mobilized off the landscape by the first flush (Lee et al., 2007), particularly in places like California where there are long periods of dry weather. If possible, we also recommend choosing tributaries that have streamflow gauges so flow-weighted samples can be collected and loads more accurately estimated.

WASTEWATER

Wastewater effluent samples can be collected as bulk water grabs (Simon et al., 2018), or wastewater can be filtered on-site to provide a sample that is more representative of a longer period of time and a larger volume of water (Dyachenko et al., 2017; Mason et al., 2016). Because one of the goals of our study was to calculate loads and develop a baseline, we filtered the wastewater effluent on-site for 24 hours. These samples were inclusive of variation in flow and composition over a day (Dyachenko et al., 2017), rather than over two hours as was conducted in our prior Bay study (Sutton et al., 2016). We also sampled Tuesday through Friday to facilitate comparison among sites and to provide a more consistent sample, as variation in effluent contaminant concentrations on weekends relative to weekdays is well documented for many contaminants (McKinney, 2004).

To capture microbeads, a 125 µm or smaller sieve should be used, as this sieve size has been found to be particularly successful for trapping microparticles used as abrasives in personal care products (Carr et al., 2016; Napper et al., 2015). In a survey of facial cleansers available in the Bay Area, the average diameter of microbeads was 264 µm (Chang, 2015).

If calculation of loads is important, collection of flow-weighted and depth-integrated samples should be considered. This study's wastewater samples were not flow-weighted, and as a result they may be biased low due to the constant flow from the sampling port (i.e., not modulated).

In addition, the samples were not depth-integrated across the sampling tank. It is not known whether this creates a bias.

As with stormwater, it is important to consider the influence of seasonality. Wet weather may increase the volume of water coming into the facilities due to infiltration of water into pipes, and thus may decrease concentrations of microparticles observed. Conversely, if the sewer system is a combined system, there can be an increase in microparticles observed as a result of stormwater runoff, particularly particles related to trash. For example, an increase in foam particles was observed in a combined sewer system during a wet weather event (Mason et al., 2016).

Quality assurance and quality control in the field

It is imperative that field QA/QC samples are collected and analyzed to evaluate the efficacy of methods and to ensure accurate quantification of microplastics (Hermsen et al., 2018; Hidalgo-Ruz et al., 2012; Koelmans et al., 2019; Prata et al., 2019; Simon et al., 2018; Van Cauwenberghe et al., 2015). Field QA/QC measures include field blanks, which provide a measurement of procedural and background contamination during sampling, and field duplicates, which provide a measure of variability in sample collection and analyses. Both field blanks and duplicates should be collected in the same manner as the samples. For trace environmental organic and metal contaminants monitored within the Bay RMP, it is standard practice to collect one field blank and one duplicate every 20 samples (Yee et al., 2018); however, in this study, due to our concern about external contamination, we collected these QA/QC samples at a greater frequency. Blanks were highly variable and had high levels of microparticles relative to field samples, especially for microfibers. Until sources of external microparticle contamination are better understood, we recommend that at least one field blank and duplicate be collected for every ten samples, and possibly more if the study design is focused on microfibers.

AVOIDING FIELD CONTAMINATION

Procedural contamination during sample collection is of particular concern given the range of plastic materials that may be used in the field that could become sources of secondary particles in samples. Fibers especially have been shown to be a significant component of background contamination (Hermsen et al., 2018; Torre et al., 2016; Wesch et al., 2017). Potential sources of field contamination include: clothing, wet weather gear, and personal flotation devices (PFDs); ropes, mats, and other vessel materials; plastic tubing and other

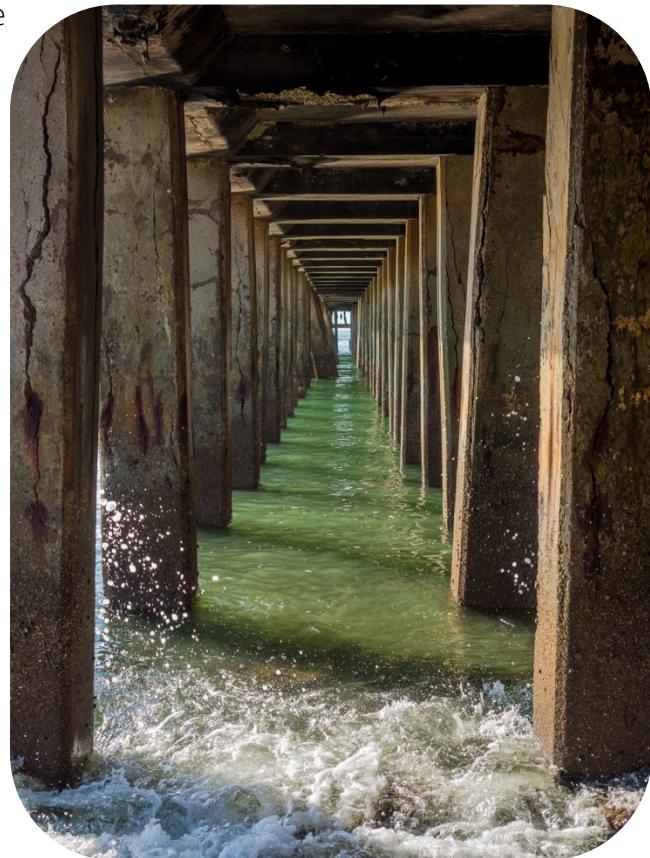
components of the sampling apparatus; more generalized air deposition; and cross-contamination caused by incomplete cleaning of sampling equipment.

It is important to reduce the use of or remove synthetic mats, ropes, and other materials as much as possible while sampling. It may be worth noting the colors of ropes used and clothing worn to assess whether these are significant sources of contamination in the samples. In this study, some surface water samples were collected using a sailing vessel, which had the potential to shed fibers from ropes, sails, and sail covers. Field staff were instructed to avoid wearing synthetic clothing, particularly those which might shed fibers with movement. Nonetheless, we did identify orange fibers in some of our surface water field blanks that we believe came from orange PFDs. In addition, during sampling, we identified a synthetic mat on the rear deck of the vessel that was shedding black curly fibers (2–9% of the fibers found in samples). The mat was removed, but this example illustrates the importance of identifying and minimizing possible microplastic sources before sampling.

Sample glass containers should be thoroughly pre-cleaned (e.g., certified as organic clean, washed with detergent, and rinsed with RO or MilliQ water) and randomly checked to ensure they are not a source of contamination. Plastic containers, sampling tools, and equipment should be avoided where possible. Blank water should be pre-analyzed to ensure it is free of microplastics. Given the potential for background contamination, we recommend a thorough evaluation of any materials used in conjunction with field sampling. That is, analysis of the rinse water, field reagents, bottles, nets, and other equipment should be conducted prior to field sampling.

FIELD BLANKS

Field blanks should be true field blanks, in which every step of the collection procedure is duplicated as closely as possible. Simplified field blanks in which only portions of the process are evaluated do not provide sufficient information to assess the magnitude of blank contamination. For example, for surface water samples, it is important that the blank water is



flushed through the manta net into the cod end and onto the sieves and then processed onboard in the same manner as the field samples; although it will, by necessity, represent a smaller flush volume. Only pouring the blank water through sieves does not provide a representative indication of the potential for field contamination. The highest quality water possible (e.g., ultra-pure water) should be used for blanks so as not to introduce contamination. In addition, a bottle blank should be collected to assess the cleanliness of the sample containers.

FIELD DUPLICATES

Field duplicates allow evaluation of variability in field measurements. Any given sample represents a discrete time and place, but because sampled environments are not static and homogeneous, field duplicates can help characterize the degree of variability that is likely to exist for all the other samples not collected in duplicate. Depending on the matrix and sampling method, the collection difficulty and degree of similarity between field duplicates may vary (e.g., for sequential versus simultaneous collections), and thus the acceptable variability between duplicates may also differ.

For example, the field duplicates for wastewater effluent in the Bay were collected by dividing the flow from a sample port using a Y-splitter pipe connection, enabling the simultaneous collection of two sample sieve sets in parallel. Stormwater field duplicates were conducted sequentially across the hydrograph with the primary sample receiving the first sip followed by the duplicate. For both of these methods, we expected and observed excellent agreement: less than 30% relative percent difference (RPD) for total microparticles. However, duplicates were more variable when counts were divided by morphology; the majority of variation was due to fibers, which was likely a consequence of their orientation and long, narrow shape affecting whether they were captured.

In contrast, collection of a field duplicate for surface water manta trawls was far more challenging because two trawls could not be deployed simultaneously without running the risk of entangling the towlines around the boat propeller. Furthermore, it can be difficult to maintain similar conditions for duplicating sequential trawls (e.g., avoiding the presence of foam lines or convergence zones). In this study, the primary sample was collected first (a 30-minute trawl), and then the vessel returned approximately 45 minutes later to the same latitude and longitude of the prior starting point and commenced the second trawl with a similar heading and speed. The RPD for these surface water duplicates varied between 2 to 105%, with open ocean waters having lower RPDs than Bay waters. Although these trawl duplicates were separated in space and time at much smaller scales than individual sites within the overall sampling effort, the variability in the field replicates suggest that large sample counts or scales of integration may be necessary to find statistically significant differences.

Laboratory analysis

Methods for characterizing microplastic contamination are rapidly evolving. Microplastics are an unusual and complicated analyte; unlike typical chemical contaminants, microplastics vary in size, shape, and composition, and thus require a different style of sample extraction, analysis, and reporting. Analysis of microplastics in environmental samples also requires an experienced laboratory due to the prevalence of background contamination sources and the need for polymer identification that often requires the use of multiple techniques and instruments.

Few interlaboratory comparisons have been performed to assess method and interlaboratory variability. Using the same laboratory for analysis of multiple matrices therefore likely yields more consistent and comparable results. In this study, fish samples were analyzed in a different laboratory, and therefore had a different blank contamination profile than the other matrices, which was not ideal for comparing across samples. If multiple laboratories will be used for analysis, cross-analysis of blank samples between laboratories involved in different components of the study may be helpful in distinguishing whether the contamination is primarily associated with the blank matrix or with the analytical process and laboratory environment.

The Southern California Coastal Water Research Project (SCCWRP) is currently designing a method evaluation study to establish a scientific foundation for selecting and standardizing laboratory analytical methods for microplastics (Southern California Coastal Water Research Project, 2019).

Extraction of particles

To be enumerated and characterized, microplastics must be separated from the surrounding environmental matrix. Numerous methods have been developed for separating microplastics from environmental matrices, and they vary significantly with matrix, laboratory capabilities, and the size of the microplastics of interest (Hidalgo-Ruz et al., 2012). In general, samples are reduced in volume by sieving; separated from other materials via filtration and/or density separation using a salt solution; and organic matrix matter digested using oxidizing, acidic, alkaline, or enzymatic methods (Prata et al., 2019). However, there is not yet a clear consensus on the most effective methods for any of these steps.

A challenge with extracting microplastics from environmental matrices is developing techniques that do not change the attributes of the microplastics (e.g., alter color or cause further fragmentation). Chemical digestion to remove organic matter, in particular, can be too

aggressive, resulting in alteration of recoveries and decreased ability to identify polymer types (Munno et al., 2018), highlighting a need for further method development and validation.

For fish and other biota, a digestion step must be included to dissolve organic matter in the sample; simply rinsing the digestive tract does not extract particles entrenched in the tissue. The general consensus supports using a 10% potassium hydroxide solution and enzymatic digestion methods for small organisms (Hermsen et al., 2018). Heating or drying samples at high temperatures should be avoided (Hermsen et al., 2018). Methods for larger organisms are still in development, as it is often ideal to sample plastic ingestion without sacrificing the organism (Provencher et al., 2019, 2017).

For non-chemical separation, standard sieve sizes and fractionation should be used for grouping particles in different operational size categories to facilitate comparisons. For this study, we initially used different size fractions for some matrices, but as we proceeded, we realized how important size fractionation was and sought to standardize size fractions across samples. We also established microparticle size limits for our current analytical methods and instrumentation. This learning process resulted in fractionation that was not always consistently conducted, making comparisons between and among matrices challenging. For example, sediment samples were initially passed through 45 µm and 500 µm sieves. After initial processing, samples were passed through 125 µm and 355 µm sieves in order to make the sample extraction and analysis process more efficient. In addition, after a few initial samples where microparticles in the 45 µm sieve fraction were quantified, this sieve size fraction was not quantified for the remaining samples because of challenges and uncertainties with material identification of this particle size fraction without automated Raman software. This made it challenging to compare samples even within the sediment dataset because different size fractions were quantified.

Visual identification of potential microplastics, the first screening step, is generally conducted manually and is very labor intensive. As a result, this step can be subjective and influenced by analyst fatigue and experience, which can result in overestimating or underestimating the microplastics present (Prata et al., 2019). Even with well-trained staff, it is likely that there will be errors. It is very labor intensive to pick microparticles from contaminated samples, and it can be cost prohibitive and logically infeasible to analyze all samples or all particles within each sample. For these reasons, development of new automated methods is needed.

In water bodies near densely populated urban areas, it may be necessary to subsample, particularly for fibers. Due to the high number of fibers present in the Bay samples, it was logically impossible to enumerate all samples for fibers (e.g., two samples each contained over 1,400 fibers). In addition, the long, narrow shape of fibers and the orientation of each

fiber can affect whether they are captured during sampling and retained throughout sample processing steps. For example, it is not clear that manta trawl sampling captures a quantitatively reliable proportion of the fibers in a sampled volume (Covernton et al., 2019; Barrows et al., 2017); as a result, we only analyzed fibers in approximately half of the surface water samples.

For studies evaluating microplastics in fish, it is important to decide what part of the fish will be targeted for microplastic analyses: whole fish, specific tissues (e.g., liver or muscle), or the digestive tract. The study questions should guide the approach (i.e., questions surrounding human health exposure should focus on the tissues that are eaten; in contrast, questions regarding wildlife exposure should focus on whole fish). To ensure all ingested microplastics are accounted for, the full gastrointestinal tract (esophagus to vent) of fish should be examined (Hermsen et al., 2018). We analyzed the digestive tract in this study because we were interested in exposure via feeding pathways. The presence of microplastics in the gut of small prey fish can also indicate possible entry into higher trophic levels because prey fish are eaten whole by their predators.

Microplastic confirmation via spectroscopic analysis

To accurately characterize microplastics in the environment, it is critical to confirm the composition of collected microparticles with methods beyond visual inspection. Without polymer identification, microparticles of other origins can be erroneously characterized as microplastics, an issue that becomes more likely with decreasing particle size and increasing sample heterogeneity (Hidalgo-Ruz et al., 2012; Primpke et al., 2018). For example, dyed natural microfibers may be lumped in with plastic microfibers, artificially inflating the apparent microplastics in a sample (Dyachenko et al., 2017).

Classification of environmental microplastics based on spectroscopy is challenging. Different techniques are more suited to different sizes and morphologies, requiring laboratories to invest in multiple pieces of specialized equipment. In general, we found that Fourier-transform infrared (FTIR) spectroscopy methods are best used on larger particles (greater than 250 µm), while Raman spectroscopy is useful for smaller particles. However, Raman does not work well for microplastics that are dyed with dark colors because the dye signal masks the polymeric signal. In this case, pyrolysis gas chromatography-mass spectrometry may be used to confirm the composition of these particles.

For fibers, the most challenging particles to analyze, another option for identification of composition is to use a process flow chart based in part on knowledge of the dyes that are used on different textiles (Zhu et al., in review). For dyes that can be used for multiple materials

(plastic and natural-based), additional staining techniques and density preparations can be used to further distinguish the materials (Zhu et al., 2019).

There is also currently a dearth of appropriate spectral reference libraries. Weathered plastics have different spectra than corresponding virgin plastic. An important outgrowth of this study is the development of an open access spectral library of microplastics (Munno et al., in review).

SUBSAMPLING

Field samples may contain many, many microparticles. In this first baseline study, in which particle concentrations were unknown, far more microparticles were collected than could reasonably be analyzed. As a result, it was necessary to subsample for spectral confirmation to complete the project within a reasonable timeframe and budget. The European Union in the Marine Strategy Framework Directive (MSFD) recommends that all microparticles smaller than 100 µm undergo spectroscopy, as at this size range it is difficult to accurately visually identify plastic (European Commission Joint Research Center, 2013). For microparticles 100 µm to 5 mm, the MSFD recommends that at least 5–10% of the microparticles of each size bin (e.g., 20–100 µm, 101–200 µm, 201–300µm, etc.) be analyzed using spectroscopy, up to 50 microparticles/bin. In the case of biological samples, Hermsen et al. (2018) recommend samples with 100 particles or less be analyzed in full; in the instances where there are greater than 100 particles, at least 50% should be analyzed with a minimum of 100 particles (Hermsen et al., 2018). If subsampling is necessary, performing spectroscopy on all particles within at least three to five sample filters and using the patterns observed to decide on a subsampling scheme may provide a more accurate subsampling method than random subsampling (Thaysen et al., in review).

For this study, we chose to focus on a subset of particles based on color and morphology. For water and sediment samples, we analyzed all particles via spectroscopy when there were less than ten of a particular particle type and color (e.g., clear spheres) in a sample. If there were more than 100 particles, 10% were analyzed by Raman or FTIR spectroscopy to determine the chemical composition of the particle. For the fish samples, the first three identified microparticles of each color of each morphology were analyzed by Raman or FTIR spectroscopy to determine the chemical composition of the particle, resulting in approximately 66% of the microparticles undergoing spectroscopic analysis. If there were more than ten microparticles in a color/morphology category within one fish, spectroscopy was conducted on 30% of that color/morphology category. For particles for which relatively few were identified (e.g., films), this resulted in a greater proportion of these samples undergoing spectroscopy.

In hindsight, given the information in the literature, it would make more sense to apportion the spectroscopy based on what is generally observed (that is, a higher percentage of expected

dominant morphologies should undergo spectroscopy). If we had used that criterion, we would have characterized more fibers and fragments via spectroscopy, and fewer films, spheres, and foams.

Quality assurance and quality control in the laboratory

Similar to field sampling, it is imperative that laboratory QA/QC measures are implemented to evaluate the efficacy of the methods and to ensure accurate quantification of microplastics. These measures include preparing and analyzing matrix spikes, laboratory blanks, analyzing certified or standard reference standards (when available), as well as maintaining a meticulously clean laboratory to reduce the introduction of external sources of microplastics.

EVALUATION OF LABORATORY METHODS

The extraction efficiency of laboratory methods should be assessed using matrix spikes, in which a known quantity of the target analytes are added to a clean matrix similar to the environmental matrix and then extracted in the same manner as the samples. In our study, we chose a range of particle sizes, morphologies, and compositions that we believed to be characteristic of what we would likely identify in Bay samples to use as the matrix spike (some purchased directly and others made in the laboratory by breaking apart larger plastic items). The laboratory also developed synthetic clean matrices to spike. The matrix spike samples were processed, extracted, and enumerated using the study methods to assess overall efficiencies. In general, the recovery for evaluated sample types and particle types should fall within established criteria for organic chemical matrix spikes (expected value \pm 35%; Yee et al., 2018).

It is a common practice in the field of trace environmental chemistry to use standard reference materials to assess the efficacy of methods in extracting and correctly identifying analytes. A standard reference material has a known quantity and composition of analyte and can be used by a laboratory to validate instruments, compare method performance and results among laboratories, and evaluate systems. Frequently, these reference materials are prepared by the National Institute of Standards and Technology (NIST). To date, certified standard reference materials for microplastics do not exist; however, there are standard reference materials for plastics that may be adapted for use in microplastics studies (Silva et al., 2018). For example, NIST has standard reference materials for 25 polymers that have been used to identify plastics in sea turtles (Jung et al., 2018). Standard reference materials for microplastics would be exceedingly helpful in standardizing the field and improving the quality of the analyses.

LABORATORY BLANKS AND SOURCES OF CONTAMINATION

Laboratory blanks are necessary to quantify and understand the sources of background contaminants in the laboratory. The RMP generally recommends one laboratory blank for every 20 samples (Yee et al., 2018). Due to concerns about background contamination, in this study, the frequency was increased to approximately one laboratory blank per ten samples. We recommend this frequency, especially in studies quantifying microfibers. Laboratory blanks from all matrices frequently contained high counts of fibers. Fragments were also observed; however, they were typically present at levels an order of magnitude lower than fibers. Very low to no counts were observed of films, foams, and spheres in laboratory blanks.

As with field sampling, laboratory blanks should duplicate every step of sample processing and analysis as closely as possible. Simplified blanks in which only portions of the process are evaluated do not provide sufficient information to assess the true magnitude of background contamination. Laboratory blanks should be included each time samples are processed and analyzed.

Procedural contamination, particularly by fibers, is a serious concern in microplastic studies (Hermsen et al., 2018; Koelmans et al., 2019; Lusher et al., 2017; Wesch et al., 2017; Woodall et al., 2015). Contamination problems are largely unavoidable without investing in clean-air devices; studies comparing samples in various settings indicated that microfiber pollution levels were usually higher when sample processing occurred outside a clean-air device such as a laminar flow hood, clean bench with particle filtration, or glove box (Torre et al., 2016; Wesch et al., 2017).

Additional recommendations for reducing airborne contamination include using bright, unusually colored lab cotton coats, as white cotton lab coats provide no way of distinguishing between laboratory contamination and white fibers that are commonly observed in the field. For example, recent reports of microplastics in fish from an urban river indicate that a majority of fibers ingested by fish are white/clear (Zheng et al., 2019), so by excluding white and clear fibers, we may be undercounting microplastics. Additionally, laboratory surfaces should be wiped down daily. Based on our experience with the detection of clear and white fibers in the laboratory blanks, we recommend using brightly colored sponges (Barrows et al., 2017) rather than Kimwipes. All glassware and metal tools should be rinsed with verified microplastics-free water three times between each sample.

It may be a good idea to evaluate blank contamination before starting a study, especially if using a new laboratory for analysis. In this study, fish samples were analyzed in a laboratory that was found to have higher blank contamination than the laboratory used for the other matrices; in hindsight, it would have been better to not use that laboratory.

Reporting results

Evaluation of field and laboratory blanks

Laboratory and field blanks may often show significant contamination, so blank results should be reported alongside field sample results. Field blank particle counts can be compared to those of the field samples in different ways; as yet, there are no standard methods for qualifying or adjusting field results based on blank results (Brander et al., in review).

If the blank contamination is relatively uniform and the source of the contamination is understood, it may be appropriate to conduct blank subtraction (Brander et al., in review; Covernton et al., 2019; Lusher et al., 2017; Provencher et al., 2019; Vandermeersch et al., 2015). Because microplastics are a diverse class of contaminant, there are multiple ways to perform blank subtraction. In some cases, only the microplastics detected in the blanks with a known source are subtracted; for example, white fibers may be removed from the analysis because they were likely shed from lab coats (e.g., this study's fish analysis; Davison and Asch, 2011; Frias et al., 2016; Güven et al., 2017; Ory et al., 2018, 2017; Rochman et al., 2015). In our study, curly black fibers that were attributed to a curly black fiber mat were identified in some of the field samples, so we subtracted this type of fiber from the particle counts for samples collected on the days when the mat was present. In the most rigorous version of this method, Kroon et al. (2018) constructed a customized spectral library made up of materials that may have contributed to contamination of their samples by collecting and analyzing materials from their field collection (e.g., net, rope used for winching, paint and rust chips, various deck and internal hoses used for cleaning), the surrounding marine environment (e.g., coral skeleton, lobster hair), the researchers (e.g., skin, hair, clothes), and laboratory processing (e.g., rubber bands, laboratory coat fibers, nitrile gloves, paper). Any microparticles in their samples with a spectral match to their customized library were omitted from sample tallies (Kroon et al., 2018). However, even the most rigorous versions of this method do not account for background microplastics from unknown sources.

Others have subtracted the mean blank total particle counts (e.g., Baalkhuyur et al., 2018) from all samples. Although the vast majority of published studies only used laboratory and not field blanks, if blank subtraction is conducted, we would recommend considering both field and laboratory blanks, for a more representative estimate of contamination. It has been recommended to group or categorize particles found in blanks in the same manner as those from environmental samples, and then subtract blank counts from the counts of each applicable categorization or grouping in the corresponding environmental sample. Prior studies have conducted subtraction by morphology (e.g., Lusher et al., 2017), or by size-morphology-color combined categories (e.g., Covernton et al., 2019; Vandermeersch et al.,

2015). Using maximum blank values provides a more conservative estimate of blank contamination than using means.

If the procedural contamination observed in the field blanks is variable and intermittent, subtracting the average of the laboratory and field blanks may not be a robust means for correcting for background levels. In our study, field samples were not blank corrected (i.e., blank counts were not subtracted from the field sample counts) due to the non-uniform nature of the observed background field and laboratory contamination. Instead, the field and laboratory blanks were used to develop data qualification threshold values to provide an index of the uncertainty of the data. We used the average of the field and laboratory blanks plus two times the blank standard deviation to provide a threshold for data qualification; for values below thresholds, we provided a caveat for the results indicating the potential for significant influence from background contamination. This is similar to the commonly used limits of detection/quantification (LODs/LOQs) to quantify uncertainty in the study of chemical environmental contaminants, and proposed for use in microplastic research by Bråte et al. (2018). LODs give the lowest concentration where detection is possible, while LOQs are values that exhibit a greater probability of being a true quantitative value and not a random fluctuation of the blank.



Standardized reporting categories and vocabulary

Many studies in the literature use the term “microplastics” even when the particles were not confirmed to be made of plastic. We propose using “microparticles” to describe small particles (less than 5 mm) that are visually identified as potentially plastics, and “microplastics” only to describe the subset of microparticles that have been confirmed to be plastic through spectroscopy or other techniques.

It is critical to develop consistent microparticle groupings and vocabulary to describe observations accurately and communicate study results effectively. A consistent vocabulary of shapes or morphologies, colors, and plastic and material types should be developed prior to commencing work, and applied consistently throughout the study. Size bins are operational in nature, but should also be clearly defined and consistently used.

Previous studies have defined many different shape and morphology categories, and we recommend the following, which appear to be most consistently used and will therefore be most conducive to comparison with the wider literature:

- ◆ Fragment — irregularly-shaped particle;
- ◆ Sphere/Pellet — hard, rounded, or spherical particle;
- ◆ Film — thin plane of flimsy material;
- ◆ Foam — lightweight, sponge-like particle; and
- ◆ Fiber or fiber bundle — thin or fibrous particle (fiber bundles consist of a number of fibers that cannot be disentangled; individual fibers within a bundle may be of similar or differing chemical composition and color).

Another challenge is the vocabulary used to describe plastics and other materials identified based on spectroscopic analyses. At present, there is not an agreed upon vocabulary, especially for how to group plastics into larger categories. We developed the categories and definitions outlined below, drawing particular guidance from previous work by Primpke et al. (2018). We have requested these categories be included in the California Environmental Data Exchange Network (CEDEN), the state repository for environmental monitoring data.

Plastic polymers primarily observed in this study:

- ◆ Anthropogenic synthetic — interpretation of Raman or FTIR spectrum indicates the material is plastic, but does not indicate which polymer is present
- ◆ Acrylic — a broad class including Polyacrylonitrile, Polyacrylamide, Polymethacrylate
- ◆ Cellulose acetate
- ◆ Nylon (also known as polyamide)

- ◆ Polycarbonate
- ◆ Polyethylene — including high density and low density polyethylenes as well as polyethylene wax; separately listed copolymers include:
 - ◆ Polyethylene co-acrylic acid
 - ◆ Ethylene/vinyl acetate copolymer
 - ◆ Polyethylene/polypropylene copolymer
- ◆ Polyester (fibers) and Polyethylene terephthalate (PET; non-fiber microplastics); separately listed copolymer:
 - ◆ Polyethylene terephthalate/polyurethane
- ◆ Polypropylene
- ◆ Polystyrene; separately listed copolymers include:
 - ◆ Polystyrene/acrylic copolymer
 - ◆ Acrylonitrile butadiene styrene
 - ◆ Styrene copolymer (multiple)
- ◆ Polyurethane
- ◆ Polyvinyl acetate
- ◆ Polyvinyl alcohol
- ◆ Polyvinyl butyral
- ◆ Polyvinyl chloride
- ◆ Polyvinyl ether; separately listed copolymer:
 - ◆ Methyl vinyl ether copolymers
- ◆ Rubber — a combination of natural (isoprene) and synthetic (styrene-butadiene) polymers

Far smaller numbers of microplastics were identified as:

- ◆ Fluorine-containing polymers including Fluoroelastomer and Polytetrafluoroethylene
- ◆ Phenolic resin
- ◆ Poly(aryletherketone)
- ◆ Polyacrolein
- ◆ Polycaprolactone
- ◆ Polyether block amide
- ◆ Polyethylenimine

Particles that may or may not be plastic include:

- ◆ Anthropogenic unknown — evidence indicates the material is anthropogenic in origin, frequently due to the presence of a color (i.e., not clear or white) or the Raman or FTIR spectrum of dyes or other synthetic compounds; the underlying material may or may not be plastic
- ◆ Unknown — spectroscopy is inconclusive
- ◆ Unknown potentially rubber — black, rubbery fragment with Raman or FTIR spectrum of carbon black or similar; while carbon black is used as a filler in vehicle tire rubber, this spectrum cannot be considered an exclusive marker for rubber

Non-plastic particles detected in this study:

- ◆ Anthropogenic cellulosic — evidence indicates the material is anthropogenic in origin, frequently due to the presence of a color (i.e., not clear or white) or the Raman or FTIR spectrum of dyes or other synthetic compounds; the underlying material is cellulosic
- ◆ Anthropogenic protein – can include silk and wool
- ◆ Asphalt
- ◆ Cellulosic — specific identification is not possible for these particles, but they may be made of cotton, rayon, modal, or Lyocell
- ◆ Cotton
- ◆ Glass
- ◆ Inorganic natural material
- ◆ Organic natural material
- ◆ Paint
- ◆ Silicone
- ◆ Stearates, lubricants, waxes
- ◆ Wool

Comparison between studies

Due to the diversity of microplastics, as well as variations in study design, sampling methods, analysis methods, and reporting, comparisons between studies can be challenging. Care should be taken when comparing results from studies using different sampling and extraction techniques. Similarly, comparison among studies with and without additional polymer identification methods should be avoided due to the potential for misidentification of microplastics using solely visual techniques, particularly for particles less than 500 µm (Lusher et al., 2017). Sieve sizes for all studies should be noted, as more microparticles will be collected from smaller sieves.

For all studies, but especially those on organisms (e.g., fish), extra care should be taken to minimize the introduction of additional variables, as each variable is an additional possible source of variation. The study objective will drive whether comparisons between species are valid. For example, it makes sense to compare different trophic levels or feeding habits only if the goal is to assess these variables, as species with vastly different food sources will have different probabilities of accumulating microplastics through their diet. Therefore, in this study, we compared our fish results only to literature reports of marine, low trophic level fish instead of all fish. For comparisons between populations of the same species, other differences between the populations should be minimized; for example, comparing fish of different sizes may not be valid unless an objective is to understand differences between fish of different ages/sizes, as fish size can correlate with accumulation of microplastics (McNeish et al., 2018).

Comparisons between fish studies should also only include studies in which the same tissues were collected and similar methods were used for microparticle extraction. Comparing whole fish concentrations and gut concentrations is not valid because microplastics can translocate out of the digestive tract and into other tissues (Abbasi et al., 2018; Collard et al., 2017; Ding et al., 2018; Messinetti et al., 2019). Likewise, studies that simply rinse the lumen of the gastrointestinal tract rather than chemically extract the tissue may underestimate accumulation if particles are entrenched in the tissue. Finally, comparisons of measured microplastic abundance in fish and effects studies must be qualified, as the vast majority of toxicological effects studies have been acute (only short-term exposure) and have used high concentrations of virgin microplastic spheres of a single polymer type (as opposed to the chronic exposures and weathered microplastics of multiple morphologies and chemical characteristics seen in wild fish).

Conclusions

This comprehensive and pioneering assessment of microplastics in San Francisco Bay and adjacent National Marine Sanctuaries provided an ideal learning opportunity for designing and implementing microplastic research. This project was a prime example of the importance of using the study questions to drive the design and selection of sampling sites, sampling and laboratory methods, and QA/QC. One of the most formidable challenges in this study was the lack of established methods for field sampling and laboratory analyses. The lessons learned in this study provide a foundation for others to build upon as they embark on studies of this important global pollutant.

References

- Abbasi, S., Soltani, N., Keshavarzi, B., Moore, F., Turner, A., Hassanaghaei, M., 2018. Microplastics in different tissues of fish and prawn from the Musa Estuary, Persian Gulf. *Chemosphere* 205, 80–87. <https://doi.org/10.1016/j.chemosphere.2018.04.076>
- Baalkhuyur, F.M., Bin Dohaish, E.-J.A., Elhalwagy, M.E.A., Alikunhi, N.M., AlSuwailem, A.M., Røstad, A., Coker, D.J., Berumen, M.L., Duarte, C.M., 2018. Microplastic in the gastrointestinal tract of fishes along the Saudi Arabian Red Sea coast. *Marine Pollution Bulletin* 131, 407–415. <https://doi.org/10.1016/j.marpolbul.2018.04.040>
- Ballent, A., Corcoran, P.L., Madden, O., Helm, P.A., Longstaffe, F.J., 2016. Sources and sinks of microplastics in Canadian Lake Ontario nearshore, tributary and beach sediments. *Marine Pollution Bulletin* 110, 383–395. <https://doi.org/10.1016/j.marpolbul.2016.06.037>
- Barrows, A.P.W., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab vs. neuston tow net: A microplastic sampling performance comparison and possible advances in the field. *Analytical Methods* 9, 1446–1453. <https://doi.org/10.1039/C6AY02387H>
- Bay Area Stormwater Management Agencies Associations (BASMAA), 2016. Tracking California's Trash (TCT) Testing Trash "Flux" Monitoring Methods in Flowing Water Bodies.
- Brander, S., Renick, V., Foley, M., Steele, C., Woo, M., Lusher, A., Carr, S.A., Helm, P.A., Box, C., Cherniak, S., Andrews, R., Rochman, C.M., in review. Sampling and QA/QC, or how many blanks do I need? A guide for scientists investigating the occurrence of microplastics across matrices.
- Bråte, I.L.N., Hurley, R., Iversen, K., Beyer, J., Thomas, K.V., Steindal, C.C., Green, N.W., Olsen, M., Lusher, A., 2018. *Mytilus* spp. as sentinels for monitoring microplastic pollution in Norwegian coastal waters: A qualitative and quantitative study. *Environmental Pollution* 243, 383–393. <https://doi.org/10.1016/j.envpol.2018.08.077>
- Carr, S.A., Liu, J., Tesoro, A.G., 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Research* 91, 174–182. <https://doi.org/10.1016/j.watres.2016.01.002>
- Chang, M., 2015. Reducing microplastics from facial exfoliating cleansers in wastewater through treatment versus consumer product decisions. *Marine Pollution Bulletin* 101, 330–333. <https://doi.org/10.1016/j.marpolbul.2015.10.074>
- Choy, C.A., Robison, B.H., Gagne, T.O., Erwin, B., Firl, E., Halden, R.U., Hamilton, J.A., Katija, K., Lisin, S.E., Rolsky, C., S. Van Houtan, K., 2019. The vertical distribution and biological transport of marine microplastics across the epipelagic and mesopelagic water column. *Scientific Reports* 9, 7843. <https://doi.org/10.1038/s41598-019-44117-2>

Chapter 9—Lessons Learned

- Collard, F., Gilbert, B., Compère, P., Eppe, G., Das, K., Jauniaux, T., Parmentier, E., 2017. Microplastics in livers of European anchovies (*Engraulis encrasicolus*, L.). Environmental Pollution 229, 1000–1005. <https://doi.org/10.1016/j.envpol.2017.07.089>
- Covernton, G.A., Pearce, C.M., Gurney-Smith, H.J., Chastain, S.G., Ross, P.S., Dower, J.F., Dudas, S.E., 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. Science of the Total Environment 667, 124–132. <https://doi.org/10.1016/j.scitotenv.2019.02.346>
- Davison, P., Asch, R., 2011. Plastic ingestion by mesopelagic fishes in the North Pacific Subtropical Gyre. Marine Ecology Progress Series 432, 173–180. <https://doi.org/10.3354/meps09142>
- Ding, J., Zhang, S., Razanajatovo, R.M., Zou, H., Zhu, W., 2018. Accumulation, tissue distribution, and biochemical effects of polystyrene microplastics in the freshwater fish red tilapia (*Oreochromis niloticus*). Environmental Pollution 238, 1–9. <https://doi.org/10.1016/j.envpol.2018.03.001>
- Dubaish, F., Liebezeit, G., 2013. Suspended microplastics and black carbon particles in the Jade System, Southern North Sea. Water, Air, & Soil Pollution 224, 1352. <https://doi.org/10.1007/s11270-012-1352-9>
- Dyachenko, A., Mitchell, J., Arsem, N., 2017. Extraction and identification of microplastic particles from secondary wastewater treatment plant (WWTP) effluent. Analytical Methods 9, 1412–1418. <https://doi.org/10.1039/C6AY02397E>
- Eriksen, M., Mason, S., Wilson, S., Box, C., Zellers, A., Edwards, W., Farley, H., Amato, S., 2013. Microplastic pollution in the surface waters of the Laurentian Great Lakes. Marine Pollution Bulletin 77, 177–182. <https://doi.org/10.1016/j.marpolbul.2013.10.007>
- European Chemicals Agency, 2019a. Annex XV Restriction Report – Microplastics (Version 1.1). European Chemicals Agency (ECHA).
- European Chemicals Agency, 2019b. Annex to the Annex XV Restriction Report – Microplastics. European Chemicals Agency (ECHA).
- European Commission Joint Research Center, 2013. Guidance on monitoring of marine litter in European seas. Publications Office, Luxembourg.
- Frias, J.P.G.L., Gago, J., Otero, V., Sobral, P., 2016. Microplastics in coastal sediments from Southern Portuguese shelf waters. Marine Environmental Research 114, 24–30. <https://doi.org/10.1016/j.marenvres.2015.12.006>
- Fuller, C.C., van Geen, A., Baskaran, M., Anima, R., 1999. Sediment chronology in San Francisco

Chapter 9—Lessons Learned

- Bay, California, defined by ^{210}Pb , ^{234}Th , ^{137}Cs , and $^{239,240}\text{Pu}$. *Marine Chemistry* 64, 7–27. [https://doi.org/10.1016/S0304-4203\(98\)00081-4](https://doi.org/10.1016/S0304-4203(98)00081-4)
- Gilbreath, A., McKee, L., Shimabuku, I., Lin, D., Werbowski, L.M., Zhu, X., Grbic, J., Rochman, C., 2019. Multiyear water quality performance and mass accumulation of PCBs, mercury, methylmercury, copper, and microplastics in a bioretention rain garden. *Journal of Sustainable Water in the Built Environment* 5, 04019004. <https://doi.org/10.1061/JSWBAY.0000883>
- Gilbreath, A.N., McKee, L.J., 2015. Concentrations and loads of PCBs, dioxins, PAHs, PBDEs, OC pesticides and pyrethroids during storm and low flow conditions in a small urban semi-arid watershed. *Science of the Total Environment* 526, 251–261. <https://doi.org/10.1016/j.scitotenv.2015.04.052>
- Greenfield, B.K., Allen, R.M., 2013. Polychlorinated biphenyl spatial patterns in San Francisco Bay forage fish. *Chemosphere* 90, 1693–1703. <https://doi.org/10.1016/j.chemosphere.2012.09.066>
- Güven, O., Gökdağ, K., Jovanović, B., Kideyş, A.E., 2017. Microplastic litter composition of the Turkish territorial waters of the Mediterranean Sea, and its occurrence in the gastrointestinal tract of fish. *Environmental Pollution* 223, 286–294. <https://doi.org/10.1016/j.envpol.2017.01.025>
- Hermsen, E., Mintenig, S.M., Besseling, E., Koelmans, A.A., 2018. Quality criteria for the analysis of microplastic in biota samples: A critical review. *Environmental Science & Technology*. 52, 10230–10240. <https://doi.org/10.1021/acs.est.8b01611>
- Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: A review of the methods used for identification and quantification. *Environmental Science & Technology*. 46, 3060–3075. <https://doi.org/10.1021/es2031505>
- Jung, M.R., Horgen, F.D., Orski, S.V., Rodriguez C., V., Beers, K.L., Balazs, G.H., Jones, T.T., Work, T.M., Brignac, K.C., Royer, S.-J., Hyrenbach, K.D., Jensen, B.A., Lynch, J.M., 2018. Validation of ATR FT-IR to identify polymers of plastic marine debris, including those ingested by marine organisms. *Marine Pollution Bulletin* 127, 704–716. <https://doi.org/10.1016/j.marpolbul.2017.12.061>
- Kataoka, T., Nihei, Y., Kudou, K., Hinata, H., 2019. Assessment of the sources and inflow processes of microplastics in the river environments of Japan. *Environmental Pollution* 244, 958–965. <https://doi.org/10.1016/j.envpol.2018.10.111>
- Koelmans, A.A., Mohamed Nor, N.H., Hermsen, E., Kooi, M., Mintenig, S.M., De France, J., 2019. Microplastics in freshwaters and drinking water: Critical review and assessment of data quality. *Water Research* 155, 410–422. <https://doi.org/10.1016/j.watres.2019.02.054>

Chapter 9—Lessons Learned

- Kroon, F., Motti, C., Talbot, S., Sobral, P., Puotinen, M., 2018. A workflow for improving estimates of microplastic contamination in marine waters: A case study from North-Western Australia. *Environmental Pollution* 238, 26–38. <https://doi.org/10.1016/j.envpol.2018.03.010>
- Lares, M., Ncibi, M.C., Sillanpää, Markus, Sillanpää, Mika, 2019. Intercomparison study on commonly used methods to determine microplastics in wastewater and sludge samples. *Environmental Science Pollution Research* 26, 12109–12122. <https://doi.org/10.1007/s11356-019-04584-6>
- Lattin, G.L., Moore, C.J., Zellers, A.F., Moore, S.L., Weisberg, S.B., 2004. A comparison of neustonic plastic and zooplankton at different depths near the southern California shore. *Marine Pollution Bulletin* 49, 291–294. <https://doi.org/10.1016/j.marpolbul.2004.01.020>
- Lee, H., Swamikannu, X., Radulescu, D., Kim, S., Stenstrom, M.K., 2007. Design of stormwater monitoring programs. *Water Research* 41, 4186–4196. <https://doi.org/10.1016/j.watres.2007.05.016>
- Leslie, H.A., Brandsma, S.H., van Velzen, M.J.M., Vethaak, A.D., 2017. Microplastics en route: Field measurements in the Dutch river delta and Amsterdam canals, wastewater treatment plants, North Sea sediments and biota. *Environment International* 101, 133–142. <https://doi.org/10.1016/j.envint.2017.01.018>
- Lowe, S., Thompson, B., Hoenicke, R., Leatherbarrow, J., Taberski, K., Smith, R., Stevens, D., 2005. Re-design Process of the San Francisco Estuary Regional Monitoring Program for Trace Substances (RMP) Status & Trends Monitoring Component for Water and Sediment (SFEI Contribution No. 109). San Francisco Estuary Institute, Richmond, CA.
- Lusher, A.L., Welden, N.A., Sobral, P., Cole, M., 2017. Sampling, isolating and identifying microplastics ingested by fish and invertebrates. *Analytical Methods* 9, 1346–1360. <https://doi.org/10.1039/C6AY02415G>
- Magnusson, K., Norén, F., 2014. Screening of microplastic particles in and down-stream a wastewater treatment plant (No. C 55). IVL Swedish Environmental Research Institute.
- Mason, S.A., Garneau, D., Sutton, R., Chu, Y., Ehmann, K., Barnes, J., Fink, P., Papazissimos, D., Rogers, D.L., 2016. Microplastic pollution is widely detected in US municipal wastewater treatment plant effluent. *Environmental Pollution* 218, 1045–1054. <https://doi.org/10.1016/j.envpol.2016.08.056>
- McKee, L.J., Leatherbarrow, J., Pearce, S., Davis, J.A., 2003. A review of urban runoff processes in the Bay Area: Existing knowledge, conceptual models, and monitoring recommendations. A report prepared for the Sources, Pathways and Loading Workgroup of the Regional

Chapter 9—Lessons Learned

Monitoring Program for Trace Substances. SFEI Contribution 66. San Francisco Estuary Institute, Oakland, CA.

McKinney, R.E., 2004. Environmental pollution control microbiology. M. Dekker, New York.

McNeish, R.E., Kim, L.H., Barrett, H.A., Mason, S.A., Kelly, J.J., Hoellein, T.J., 2018. Microplastic in riverine fish is connected to species traits. *Scientific Reports* 8. <https://doi.org/10.1038/s41598-018-29980-9>

Messinetti, S., Mercurio, S., Scarì, G., Pennati, A., Pennati, R., 2019. Ingested microscopic plastics translocate from the gut cavity of juveniles of the ascidian *Ciona intestinalis*. *European Zoological Journal* 86, 189–195. <https://doi.org/10.1080/24750263.2019.1616837>

Miller, R.Z., Watts, A.J.R., Winslow, B.O., Galloway, T.S., Barrows, A.P.W., 2017. Mountains to the sea: River study of plastic and non-plastic microfiber pollution in the northeast USA. *Marine Pollution Bulletin* 124, 245–251. <https://doi.org/10.1016/j.marpolbul.2017.07.028>

Moore, C.J., Lattin, G.L., Zellers, A.F., 2011. Quantity and type of plastic debris flowing from two urban rivers to coastal waters and beaches of Southern California. *Revista da Gestão Costeira Integrada* 11, 65–73. <https://doi.org/10.5894/rgci194>

Munno, K., De Frond, H., O'Donnell, B., Rochman, C., in review. Increasing the accessibility for characterizing microplastics: Introducing new application-based and spectral libraries of plastic particles (SLoPP & SLoPP-E).

Munno, K., Helm, P.A., Jackson, D.A., Rochman, C., Sims, A., 2018. Impacts of temperature and selected chemical digestion methods on microplastic particles: Impacts of temperature and digestion method on microplastics. *Environmental Toxicology and Chemistry* 37, 91–98. <https://doi.org/10.1002/etc.3935>

Napper, I.E., Bakir, A., Rowland, S.J., Thompson, R.C., 2015. Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics. *Marine Pollution Bulletin* 99, 178–185. <https://doi.org/10.1016/j.marpolbul.2015.07.029>

Ory, N., Chagnon, C., Felix, F., Fernández, C., Ferreira, J.L., Gallardo, C., Garcés Ordóñez, O., Henostroza, A., Laaz, E., Mizraji, R., Mojica, H., Murillo Haro, V., Ossa Medina, L., Preciado, M., Sobral, P., Urbina, M.A., Thiel, M., 2018. Low prevalence of microplastic contamination in planktivorous fish species from the southeast Pacific Ocean. *Marine Pollution Bulletin* 127, 211–216. <https://doi.org/10.1016/j.marpolbul.2017.12.016>

Ory, N.C., Sobral, P., Ferreira, J.L., Thiel, M., 2017. Amberstripe scad *Decapterus muroadsii* (*Carangidae*) fish ingest blue microplastics resembling their copepod prey along the coast of Rapa Nui (Easter Island) in the South Pacific subtropical gyre. *Science of the Total Environment* 586, 430–437. <https://doi.org/10.1016/j.scitotenv.2017.01.175>

Chapter 9—Lessons Learned

- Prata, J.C., da Costa, J.P., Duarte, A.C., Rocha-Santos, T., 2019. Methods for sampling and detection of microplastics in water and sediment: A critical review. *TrAC Trends in Analytical Chemistry* 110, 150–159. <https://doi.org/10.1016/j.trac.2018.10.029>
- Primpke, S., Wirth, M., Lorenz, C., Gerdts, G., 2018. Reference database design for the automated analysis of microplastic samples based on Fourier transform infrared (FTIR) spectroscopy. *Analytical and Bioanalytical Chemistry* 410, 5131–5141. <https://doi.org/10.1007/s00216-018-1156-x>
- Provencher, J.F., Bond, A.L., Avery-Gomm, S., Borrelle, S.B., Bravo Rebollo, E.L., Hammer, S., Kühn, S., Lavers, J.L., Mallory, M.L., Trevail, A., van Franeker, J.A., 2017. Quantifying ingested debris in marine megafauna: A review and recommendations for standardization. *Analytical Methods* 9, 1454–1469. <https://doi.org/10.1039/C6AY02419J>
- Provencher, J.F., Borrelle, S.B., Bond, A.L., Lavers, J.L., van Franeker, J.A., Kühn, S., Hammer, S., Avery-Gomm, S., Mallory, M.L., 2019. Recommended best practices for plastic and litter ingestion studies in marine birds: Collection, processing, and reporting. *FACETS* 4, 111–130. <https://doi.org/10.1139/facets-2018-0043>
- Reisser, J., Shaw, J., Wilcox, C., Hardesty, B.D., Proietti, M., Thums, M., Pattiaratchi, C., 2013. Marine plastic pollution in waters around Australia: Characteristics, concentrations, and pathways. *PLOS ONE* 8, e80466. <https://doi.org/10.1371/journal.pone.0080466>
- Rochman, C.M., Tahir, A., Williams, S.L., Baxa, D.V., Lam, R., Miller, J.T., Teh, F.-C., Werorilangi, S., Teh, S.J., 2015. Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. *Scientific Reports* 5, 14340. <https://doi.org/10.1038/srep14340>
- San Francisco Regional Water Quality Control Board - Region 2, 2019. San Francisco Basin Plan (downloaded Jan 23 2019 from website). www.waterboards.ca.gov
- Science Advice for Policy by European Academies, 2019. A Scientific Perspective on Microplastics in Nature and Society (Version 2019.1.1).
- Sedlak et al., 2017. Sampling and Analysis Plan for Microplastic Monitoring in San Francisco Bay and Adjacent National Marine Sanctuaries. SFEI Contribution No. 819. San Francisco Estuary Institute, Richmond, CA.
- Sedlak, M.D., Greig, D.J., 2012. Perfluoroalkyl compounds (PFCs) in wildlife from an urban estuary. *Journal of Environmental Monitoring* 14, 146–154. <https://doi.org/10.1039/C1EM10609K>
- Silva, A.B., Bastos, A.S., Justino, C.I.L., da Costa, J.P., Duarte, A.C., Rocha-Santos, T.A.P., 2018. Microplastics in the environment: Challenges in analytical chemistry - A review. *Analytica*

Chapter 9—Lessons Learned

- Chimica Acta 1017, 1–19. <https://doi.org/10.1016/j.aca.2018.02.043>
- Simon, M., van Alst, N., Vollertsen, J., 2018. Quantification of microplastic mass and removal rates at wastewater treatment plants applying Focal Plane Array (FPA)-based Fourier Transform Infrared (FT-IR) imaging. Water Research 142, 1–9.
<https://doi.org/10.1016/j.watres.2018.05.019>
- Southern California Coastal Water Research Project, 2019. Microplastics Methods Evaluation Study Plan Third draft for participant review.
- Sutton, R., Mason, S.A., Stanek, S.K., Willis-Norton, E., Wren, I.F., Box, C., 2016. Microplastic contamination in the San Francisco Bay, California, USA. Marine Pollution Bulletin 109, 230–235. <https://doi.org/10.1016/j.marpolbul.2016.05.077>
- Thaysen, C., Munno, K., Hermabessiere, L., Rochman, C., in review. Towards Raman automation for microplastics: Developing strategies for particle adhesion and filter subsampling.
- Torre, M., Digka, N., Anastasopoulou, A., Tsangaridis, C., Mytilineou, C., 2016. Anthropogenic microfibres pollution in marine biota. A new and simple methodology to minimize airborne contamination. Marine Pollution Bulletin 113, 55–61.
<https://doi.org/10.1016/j.marpolbul.2016.07.050>
- Van Cauwenberghe, L., Devriese, L., Galgani, F., Robbens, J., Janssen, C.R., 2015. Microplastics in sediments: A review of techniques, occurrence and effects. Marine Environmental Research 111, 5–17. <https://doi.org/10.1016/j.marenvres.2015.06.007>
- Vandermeersch, G., Van Cauwenberghe, L., Janssen, C.R., Marques, A., Granby, K., Fait, G., Kotterman, M.J.J., Diogène, J., Bekaert, K., Robbens, J., Devriese, L., 2015. A critical view on microplastic quantification in aquatic organisms. Environmental Research 143, 46–55.
<https://doi.org/10.1016/j.envres.2015.07.016>
- Welden, N.A., Lusher, A.L., 2017. Impacts of changing ocean circulation on the distribution of marine microplastic litter: Changing ocean circulation and marine microplastic litter. Integrated Environmental Assessment and Management 13, 483–487.
<https://doi.org/10.1002/ieam.1911>
- Wesch, C., Elert, A.M., Wörner, M., Braun, U., Klein, R., Paulus, M., 2017. Assuring quality in microplastic monitoring: About the value of clean-air devices as essentials for verified data. Scientific Reports 7. <https://doi.org/10.1038/s41598-017-05838-4>
- Wolff, S., Kerpen, J., Prediger, J., Barkmann, L., Müller, L., 2019. Determination of the microplastics emission in the effluent of a municipal waste water treatment plant using Raman microspectroscopy. Water Research X 2, 100014.
<https://doi.org/10.1016/j.wroa.2018.100014>

Chapter 9—Lessons Learned

- Woodall, L.C., Gwinnett, C., Packer, M., Thompson, R.C., Robinson, L.F., Paterson, G.L.J., 2015. Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments. *Marine Pollution Bulletin* 95, 40–46. <https://doi.org/10.1016/j.marpolbul.2015.04.044>
- Xiong, X., Wu, C., Elser, J.J., Mei, Z., Hao, Y., 2019. Occurrence and fate of microplastic debris in middle and lower reaches of the Yangtze River – From inland to the sea. *Science of the Total Environment* 659, 66–73. <https://doi.org/10.1016/j.scitotenv.2018.12.313>
- Yee, D., Bemis, B., Hammond, D., Heim, W., Jaffe, B., Rattonetti, A., van Bergen, S., 2011. Age Estimates and Pollutant Concentrations of Sediment Cores from San Francisco Bay and Wetlands. SFEI Contribution No. 652. San Francisco Estuary Institute, Richmond, CA.
- Yee, D., Franz, A., Trowbridge, P., Wong, A., 2018. Quality Assurance Program Plan for the Regional Monitoring Program for Water Quality in San Francisco Bay. SFEI Contribution No. 890. San Francisco Estuary Institute, Richmond, CA.
- Zheng, K., Fan, Y., Zhu, Z., Chen, G., Tang, C., Peng, X., 2019. Occurrence and species-specific distribution of plastic debris in wild freshwater fish from the Pearl River Catchment, China. *Environmental Toxicology and Chemistry* etc.4437. <https://doi.org/10.1002/etc.4437>
- Zhu, X., Nguyen, B., You, J.B., Karakolis, E., Sinton, D., Rochman, C. 2019. Identification of microfibers in the environment using multiple lines of evidence. *Environmental Science & Technology*. <https://doi.org/10.1021/acs.est.9b05262>
- Zhu, X., Nguyen, B., You, J.B., Sinton, D., Rochman, C., in review. A novel method of using Raman spectroscopy and fluorescent staining to identify microfibers to polymer type.

Appendices

Stormwater

Table A-2.1. Total particle counts for each sampled location and laboratory blanks.

<i>Sample Location / Lab Blank</i>	<i>Total Particle Count</i>
Line 12M	1744
Line 12M Duplicate	2344
Meeker Slough	1283
Line 12A	1504
Line 12J	768
Lower Coyote Ck	903
Line 12F	181
Colma Ck	1203
Line 12K	1551
Rodeo Ck	180
Guadalupe R	386
Refugio Ck	152
San Mateo Ck	153
Lab Blank 1	52
Lab Blank 2	53
Lab Blank 3	5
Field Blank	58

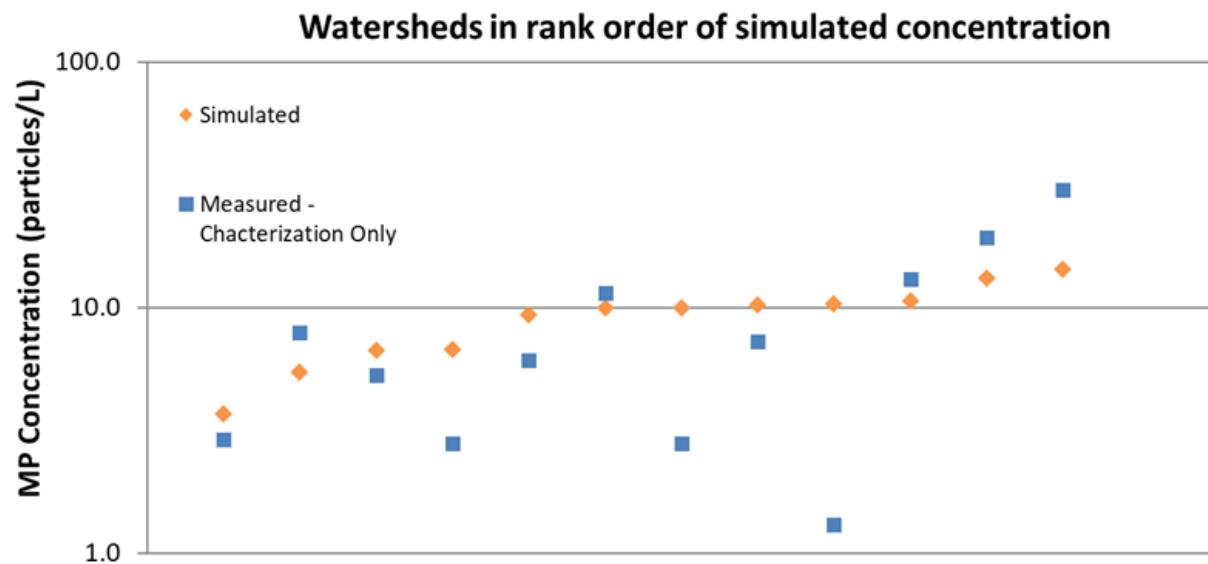


Figure A-2.1. Simulated vs. measured concentration results for the second model (Model 2) calibration. Watersheds in rank order of simulated concentrations.

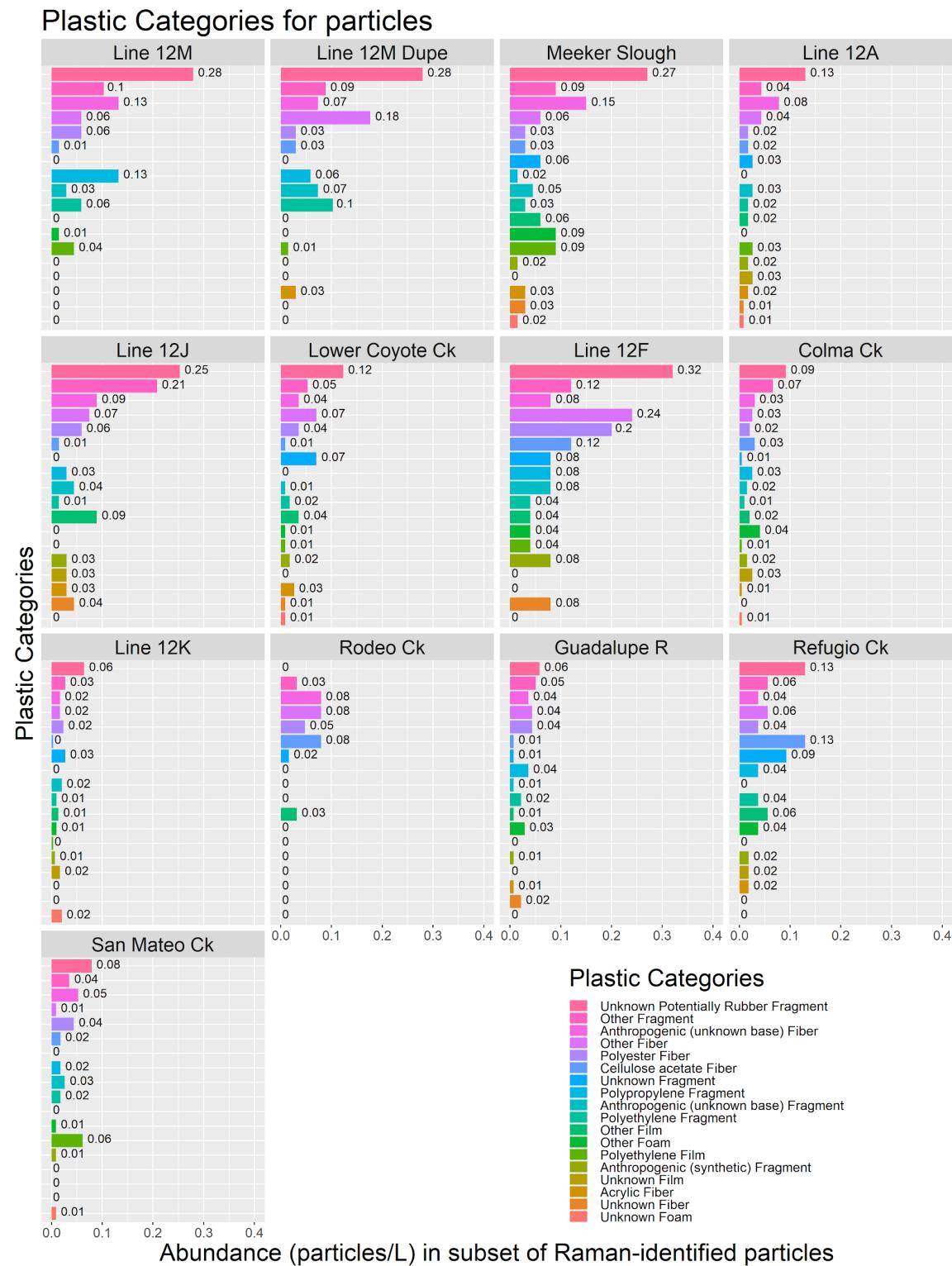


Figure A-2.2. Concentrations of spectroscopically examined microparticles at each site. These numbers reflect concentrations based on the number of particles examined at each site, but those numbers are not proportional to the total number of particles measured at each site. Polyethylene, polypropylene, cellulose acetate, polyester, and rubber are considered plastic. The most abundant categories are shown here, while the abundances of all other microparticles are lumped into the category labeled "Other."

Wastewater

Appendix Table A-3.1. Summary of microparticle effluent concentrations reported in the literature.

Location	Treatment	Concentration (Microparticles/L)	Sample analysis	Smallest sieve size (μm)	Reference
Bay Area study of 8 facilities	Secondary/ Tertiary	0.063 (ave)	Visual, Raman/FTIR (subset)	125	This study
Bay Area study of 8 facilities	Secondary/ Tertiary	0.086 (ave)	Visual	125	Sutton et al. 2016
EBMUD, California	Secondary	0.02	Visual/Raman/FTIR (subset)	125	Dyachenko et al. 2017
Los Angeles, California	Secondary	0.0008	Visual, FTIR (select particles)	45	Carr et al. 2016
National study of 17 facilities	Secondary /Tertiary	0.05 (ave)	Visual	125	Mason et al. 2016
Italy	Tertiary	0.4	FTIR		Magni et al. 2019
Helsinki, Finland	Tertiary	0.7 -3.5	FTIR	20	Talvitie et al. 2017
Sydney, Australia	Tertiary	1	Visual	NS	Browne et al. 2011
Paris, France	Secondary	35	Visual	100	Dris et al. 2015
Holland	Secondary /Tertiary	9-91	Visual, FTIR (subset)	10	Leslie et al. 2017
Germany	Secondary	3.0 (ave)	Raman (for select particles)	10	Wolff et al. 2019
Denmark (10 facilities)	Secondary (most)	54 (ave)	FTIR (subsampling)	10	Simon et al. 2018
German WWTPs (12 facilities)	Secondary/ Tertiary	2 -14	Visual/FTIR (subsampling)	10	Minnetig et al. 2017
Lysekil, Sweden	Secondary	0.008	Visual	300	Magnusson and Noren 2014
Glasgow, Scotland	Secondary	0.25	FTIR	65	Murphy et al. 2016

Surface Water

Table A-4.1. Locations and dates of surface water samples collected in the San Francisco Bay

Site ID	Location	Subregion	Start Latitude¹	Start Longitude¹	End Latitude¹	End Longitude¹	Rationale for site selection	Dry Date Sampled	Wet Date Sampled
LSB16	Lower South Bay	Near Guadalupe River	37.464	-122.027	37.463	-122.052	Receiving water for tributaries; wastewater	8/24/17 ^{2,3} (1L dup)	3/6/18 ^{2,3} 3/6/18 (pump dup)
LSB15	Lower South Bay	Near Palo Alto WWTP	37.461	-122.084	37.47	-122.06	Receiving water near wastewater	8/24/17 ³	3/6/18 ³ 3/6/18 (dup)
LSB14	Lower South Bay	Main stem of LSB	37.471	-122.064	37.483	-122.084	Ambient conditions in LSB embayment	8/24/17 ^{2,3}	3/6/18 ^{2,3}
SB11	South Bay	Main portion of Bay - Southeast	37.598	-122.25	37.633	-122.253	Ambient conditions in SB embayment	8/23/17 ³	3/19/18 ³
SB13	South Bay	Near San Mateo creek	37.57	-122.213	37.548	-122.181	Receiving water for tributaries	8/23/17 ^{2,3}	3/19/18 ^{2,3}
SB10	South Bay	Main portion of South Bay - Northeast	37.65	-122.243	37.671	-122.278	Ambient conditions in SB embayment	8/23/17 ³	3/19/18 ³
SB12	South Bay	Main portion of South Bay - Southwest	37.594	-122.283	37.578	-122.243	Ambient conditions in SB embayment	8/23/17 ^{2,3} (1L dup)	3/19/18 ^{2,3} 3/19/18 (dup)
CB9	Central Bay	Main portion of Bay - Near EBDA outfall	37.687	-122.291	37.699	-122.298	Receiving water for WWTP-EBDA	8/22/17 ^{2,3}	1/11/18 ^{2,3}
CB8	Central Bay	San Leandro Creek / Oakland Airport	37.751	-122.226	37.769	-122.231	Receiving waters for tributaries	8/25/17 ^{2,3} 8/25/17 (dup)	1/11/18 ^{2,3}
CB6	Central Bay	Emeryville	37.834	-122.32	37.828	-122.337	Receiving waters for tributaries	8/22/17 ³	11/16/17 ³

Appendices

<i>Site ID</i>	<i>Location</i>	<i>Subregion</i>	<i>Start Latitude¹</i>	<i>Start Longitude¹</i>	<i>End Latitude¹</i>	<i>End Longitude¹</i>	<i>Rationale for site selection</i>	<i>Dry Date Sampled</i>	<i>Wet Date Sampled</i>
CB7	Central Bay	South of Bay bridge	37.778	-122.355	37.804	-122.381	Ambient conditions	8/22/17 ³	11/16/17 ³ 11/16/17 (dup)
CB5	Central Bay	Main Channel in Central Bay, Southeast of Angel Island	37.843	-122.415	37.852	-122.454	Ambient conditions	8/22/17 ³ 11/5/17 11/5/17 (dup)	11/16/17 ³
CB4	Central Bay	South of Richmond / San Rafael bridge	37.916	-122.441	37.942	-122.42	Ambient conditions	8/21/17 ³	11/16/17 ³
SFBay	Central Bay	Southeast of Treasure Island / North of Bay Bridge	37.82	-122.357	37.833	-122.362	Ambient conditions	9/18/17	N/A
SPB3	North Bay	Point Pinole	38.024	-122.371	38.042	-122.322	Receiving water for tributaries	8/21/17 ^{2,3}	11/17/17 ^{2,3}
SUB1	North Bay	Suisun Bay main	38.107	-122.056	38.097	-122.065	Ambient conditions	8/21/17 ³	11/17/17 ³
SPB2	North Bay	San Pablo Bay main	38.051	-122.422	38.023	-122.428	Ambient conditions	8/21/17 ^{2,3}	11/16/17 ^{2,3}

¹ Latitude & longitude values recorded in this table represent the actual location where the first dry season manta trawl sample was collected at each site. Latitude and longitude for other water samples collected at each site are displayed in Figure 4-1 and 4-2 and will be available for download from CEDEN.

² Sampling events at which pump samples were also collected.

³ Sampling events at which 1-liter grab samples were also collected.

Appendices

Table A-4.2. Locations and dates of surface water samples collected in the National Marine Sanctuaries

Site ID	Location	Subregion	<i>Start Latitude¹</i>	<i>Start Longitude¹</i>	<i>End Latitude¹</i>	<i>End Longitude¹</i>	Rationale for site selection	Dry Date Sampled	Wet Date Sampled
CBNMS23	Cordell Banks	Central region	38.035	-123.313	38.043	-123.285	Ambient conditions	9/12/17 ^{2,3}	3/29/18 ²
CBNMS22	Cordell Banks	East side	38.107	-123.114	38.097	-123.098	Ambient conditions	9/12/17 ³	3/30/18
CBNMS24	Cordell Banks	West side	37.985	-123.497	37.98	-123.466	Ambient conditions	9/13/17 ³	3/29/18
GFNMS26	Greater Farallones	Farallon Islands	37.821	-123.007	37.819	-122.98	Ambient	9/12/17 ^{2,3} 9/12/17 (dup)	3/29/18 ^{2,3}
GFNMS28	Greater Farallones	At discharge of GG; SF Plume	37.806	-122.756	37.804	-122.729	Modeling; Load Calculations	9/13/17 ^{2,3}	3/30/18 ^{2,3} (pump dup)
GFNMS25	Greater Farallones	Off of Point Reyes	37.967	-122.927	37.957	-122.904	Convergence zone off of Pt Reyes	9/11/17 ³	3/30/18 ³
GFNMS27	Greater Farallones	West side	37.733	-123.263	37.725	-123.251	Remote part of Greater Farallones - reference comparison	9/13/17 ³	3/29/18
GFNMS29	Monterey Bay	At discharge of GG; SF Plume	37.805	-122.508	37.815	-122.471	Modeling; load calculations; outgoing tide	9/13/17 ^{2,3} 3/30/18 ² 3/31/18 ³ (1L grab only)	11/17/17 ^{2,3} 1/11/17 ³ 3/30/18 ²
MBNMS30	Monterey Bay	At discharge of GG; SF Plume	37.672	-122.611	37.662	-122.585	Modeling; load calculations	9/27/17 ^{2,3} 3/31/18 ^{2,3} (pump dup)	3/31/18 ^{2,3}

Appendices

Site ID	Location	Subregion	Start Latitude¹	Start Longitude¹	End Latitude¹	End Longitude¹	Rationale for site selection	Dry Date Sampled	Wet Date Sampled
MBNMS31	Monterey Bay	Off the coast of Ano Nuevo	37.507	-122.58	37.524	-122.58	Upwelling areas around Pt Ano Nuevo	9/27/17 ³	3/31/18 ³
MBNMS32	Monterey Bay	West side	37.45	-122.932	37.461	-122.905	Remote part of Monterey Bay - reference for comparison	9/27/17 ³	3/31/18 ^{2,3}

¹ Latitude & longitude values recorded in this table represent the actual location where the first dry season manta trawl sample was collected at each site. Latitude and longitude for other water samples collected at each site are displayed in the maps in Table 4-1 and 4-2 and will be available for download from CEDEN.

² Sampling events at which pump samples were also collected.

³ Sampling events at which 1-liter grab samples were also collected.

Appendices

Table A-4.3a. Manta trawl results (particle count) for Bay samples.

MANTA TRAWL RESULTS: PARTICLE COUNT (COUNT/SAMPLE) FOR BAY SAMPLES										
Location	Season	Sample ID	TOTAL particles (w/o fibers)	TOTAL particles (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
Bay Samples	Dry Weather	CB4-Manta-21Aug2017	5	89	5	0	0	0	84	0
		CB5-Manta-22Aug2017	62	-	46	8	0	8		
		CB5-Manta-5Nov2017	36	312	25	4	1	6	275	1 ¹
		CB5-Manta-DUP-5Nov2017	27	300	7	20	0	0	273	0
		CB6-Manta-22Aug2017	8	156	6	1	1	0	146	2 ¹
		CB7-Manta-22Aug2017	165	514	12	148	1	4	349	0
		CB8-Manta-25Aug2017	399	-	309	67	9	14		
		CB8-Manta-DUP-25Aug2017	333	333	160	153	5	15		
		CB9-Manta-22Aug2017	30	57 ¹	26	0	2	2	27 ¹	0
		LSB14-Manta-24Aug2017	165	303	122	3	16	24	137	1 ¹
		LSB15-Manta-24Aug2017	46	-	37	3	0	6		
		LSB16-Manta-24Aug2017	16	-	12	4	0	0		
		SB10-Manta-23Aug2017	18	52 ¹	16	2	0	0	32 ¹	2 ¹
		SB11-Manta-23Aug2017	34	100	25	9	0	0	56 ¹	10
		SB12-Manta-23Aug2017	73	-	56	2	5	10		
		SB13-Manta-23Aug2017	311	360	273	11	5	22	44 ¹	5 ¹
		Alcatraz-Manta-18Sept2017	42	-	30	2	7	3		
		SPB2-Manta-21Aug2017	7	94	7	0	0	0	86	1 ¹
		SPB3-Manta-21Aug2017	120	767	81	8	1	30	647	0
		SPB1-Manta-21Aug2017	12	-	2 ¹	10	0	0		

MANTA TRAWL RESULTS: PARTICLE COUNT (COUNT/SAMPLE) FOR BAY SAMPLES										
Location	Season	Sample ID	TOTAL particles (w/o fibers)	TOTAL particles (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
Bay Samples	Wet Weather	CB4-Manta-16Nov2017	630	-	476	72	69	13		
		CB5-Manta-16Nov2017	254	-	27	209	15	3		
		CB6-Manta-16Nov2017	60	-	27	24	2	7		
		CB7-Manta-16Nov2017	101	549	34	49	11	7	448	0
		CB7-Manta-DUP-16Nov2017	223	415	79	97	45	2	192	0
		CB8-Manta-11Jan18	1228	2675	1014	157	10	47	1,404	43
		CB9-Manta-11Jan18	4959	-	3932	627	241	159		
		LSB14-Manta-06Mar2018	26	-	20	0	0	6		
		LSB15-Manta-06Mar2018	23	-	22	0	0	1		
		LSB15-Manta-DUP-06Mar2018	78	-	72	0	0	6		
		LSB16-Manta-06Mar2018	48	-	36	0	1	11		
		SB10-Manta-19Mar2018	14	434	6	5	2	1	420	0
		SB11-Manta-19Mar2018	356	877	306	12	5	33	521	0
		SB12-Manta-19Mar2018	58	-	43	9	4	2		
		SB12-Manta-DUP-19Mar2018	110	1848	36	58	14	2	1,735	3 ¹
		SB13-Manta-19Mar2018	119	396	75	21	0	23	277	0
		SPB2-Manta-16Nov2017	13	323	10	0	0	3	310	0
		SPB3-Manta-17Nov17	212	-	198	0	5	9		
		SUB1-Manta-17Nov17	18	-	16	1	0	1		

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation.

Appendices

Table A-4.3b. Manta trawl results (abundance) for Bay samples.

MANTA TRAWL RESULTS: ABUNDANCE (PARTICLES/KM ²) FOR BAY SAMPLES											
Location	Season	Sample ID	Area (km ²)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Bay Samples	Dry Weather	CB4-Manta-21Aug2017	0.0021187	2,360	0	0	0	39,646	0	2,360	42,006
		CB5-Manta-22Aug2017	0.0014477	31,775	5,526	0	5,526			42,828	
		CB5-Manta-5Nov2017	0.0007099	35,214	5,634	1,409	8,451	387,354	1,409 ¹	50,708	439,471
		CB5-Manta-DUP-5Nov2017	0.0011566	6,052	17,292	0	0	236,038	0	23,344	259,383
		CB6-Manta-22Aug2017	0.0008632	6,951	1,158	1,158	0	169,132	2,317 ¹	9,268	180,716
		CB7-Manta-22Aug2017	0.0022228	5,398	66,581	450	1,799	157,006	0	74,229	231,235
		CB8-Manta-25Aug2017	0.0018467	167,330	36,282	4,874	7,581			216,067	
		CB8-Manta-DUP-25Aug2017	0.0018906	84,628	80,925	2,645	7,934			176,131	
		CB9-Manta-22Aug2017	0.0016082	16,167	0	1,244	1,244	16,789 ¹	0	18,655	35,444 ¹
		LSB14-Manta-24Aug2017	0.0017622	69,231	1,702	9,079	13,619	77,743	567 ¹	93,632	171,943
		LSB15-Manta-24Aug2017	0.0013061	28,328	2,297	0	4,594			35,218	
		LSB16-Manta-24Aug2017	0.001048	11,450	3,817	0	0			15,267	
		SB10-Manta-23Aug2017	0.0015401	10,389	1,299	0	0	20,778 ¹	1,299 ¹	11,687	33,764 ¹
		SB11-Manta-23Aug2017	0.0018783	13,310	4,792	0	0	29,814 ¹	5,324	18,102	53,240
		SB12-Manta-23Aug2017	0.0058678	9,544	341	852	1,704			12,441	
		SB13-Manta-23Aug2017	0.001123	243,109	9,796	4,453	19,591	39,182 ¹	4,453 ¹	276,948	320,583
		Alcatraz-Manta-18Sept2017	0.0010045	29,866	1,991	6,969	2,987			41,812	
		SPB2-Manta-21Aug2017	0.0011466	6,105	0	0	0	75,003	872 ¹	6,105	81,980
		SPB3-Manta-21Aug2017	0.0019546	41,440	4,093	512	15,348	331,006	0	61,392	392,398
		SPB1-Manta-21Aug2017	0.0015042	1,330 ¹	6,648	0	0			7,978	

Appendices

MANTA TRAWL RESULTS: ABUNDANCE (PARTICLES/KM ²) FOR BAY SAMPLES											
Location	Season	Sample ID	Area (km ²)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Bay Samples	Wet Weather	CB4-Manta-16Nov2017	0.0010817	440,037	66,560	63,787	12,018			582,402	
		CB5-Manta-16Nov2017	0.0014084	19,171	148,397	10,651	2,130			180,349	
		CB6-Manta-16Nov2017	0.0013267	20,351	18,090	1,507	5,276			45,224	
		CB7-Manta-16Nov2017	0.0014363	23,671	34,115	7,658	4,874	311,906	0	70,318	382,224
		CB7-Manta-DUP-16Nov2017	0.0013521	58,427	71,740	33,281	1,479	142,001	0	164,928	306,929
		CB8-Manta-11Jan18	0.0029719	341,201	52,829	3,365	15,815	472,432	14,469	413,210	900,112
		CB9-Manta-11Jan18	0.0007941	4,951,798	789,618	303,505	200,238			6,245,160	
		LSB14-Manta-06Mar2018	0.0009743	20,527	0	0	6,158			26,686	
		LSB15-Manta-06Mar2018	0.0009203	23,906	0	0	1,087			24,993	
		LSB15-Manta-DUP-06Mar2018	0.0009743	73,901	0	0	6,158			80,060	
		LSB16-Manta-06Mar2018	0.000948	37,974	0	1,055	11,603			50,632	
		SB10-Manta-19Mar2018	0.0005851	10,255	8,546	3,418	1,709	717,827	0	23,928	741,755
		SB11-Manta-19Mar2018	0.0014311	213,824	8,385	3,494	23,059	364,059	0	248,762	612,821
		SB12-Manta-19Mar2018	0.0012192	35,270	7,382	3,281	1,640			47,574	
		SB12-Manta-DUP-19Mar2018	0.0009992	36,027	58,044	14,011	2,002	1,736,319	3,002 ¹	110,084	1,849,405
		SB13-Manta-19Mar2018	0.0008933	83,954	23,507	0	25,746	310,071	0	133,207	443,278
		SPB2-Manta-16Nov2017	0.0013158	7,600	0	0	2,280	235,596	0	9,880	245,476
		SPB3-Manta-17Nov17	0.001572	125,956	0	3,181	5,725			134,862	
		SUB1-Manta-17Nov17	0.0013426	11,917	745	0	745			13,407	

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation.

Appendices

Table A-4.3c. Manta trawl results (particle count) for marine sanctuary samples.

MANTA TRAWL RESULTS: PARTICLE COUNT (COUNT/SAMPLE) FOR MARINE SANCTUARY SAMPLES										
Location	Season	Sample ID	TOTAL particles (w/o fibers)	TOTAL particles (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
Sanctuary Samples	Dry Weather	CBNMS22-Manta-12Sept2017	5	43 ¹	2 ¹	1	0	2	38 ¹	0
		CBNMS23-Manta-12Sept2017	15	36 ¹	13	0	0	2	21 ¹	0
		CBNMS24-Manta-13Sept2017	0	95	0	0	0	0	95	0
		GFNMS25-Manta-11Sept2017	3 ¹	257	2 ¹	1	0	0	254	0
		GFNMS26-Manta-12Sept2017	12	-	9	3	0	0		
		GFNMS26-Manta-DUP-12Sept2017	11	48 ¹	4 ¹	0	0	7	37 ¹	0
		GFNMS27-Manta-13Sept2017	1 ¹	78	1 ¹	0	0	0	75 ¹	2 ¹
		GFNMS28-Manta-13Sept2017	8	69 ¹	5	1	0	2	61 ¹	0
		MBNMS29-Manta-13Sept2017	4 ¹	32 ¹	4 ¹	0	0	0	28 ¹	0
		MBNMS30-Manta-27Sept2017	10	136	8	0	2	0	126	0
Sanctuary Samples	Wet Weather	MBNMS31-Manta-27Sept2017	5	-	3 ¹	0	2	0		
		MBNMS32-Manta-27Sept2017	3 ¹	51 ¹	2 ¹	0	1	0	47 ¹	1 ¹
		CBNMS22-Manta-30Mar2018	14	-	11	1	1	1		
		CBNMS23-Manta-29Mar2018	19	-	14	3	0	2		
		CBNMS24-Manta-29Mar2018	5	-	3 ¹	1	0	1		
		GFNMS25-Manta-30Mar2018	30	-	27	0	3	0		
		GFNMS26-Manta-29Mar2018	11	-	9	0	0	2		
		GFNMS27-Manta-29Mar2018	10	275	7	1	1	1	265	0
		GFNMS28-Manta-30Mar2018	16	-	13	1	1	1		

MANTA TRAWL RESULTS: PARTICLE COUNT (COUNT/SAMPLE) FOR MARINE SANCTUARY SAMPLES										
Location	Season	Sample ID	TOTAL particles (w/o fibers)	TOTAL particles (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
Sanctuary Samples	Wet Weather	MBNMS-29-Manta-17Nov17	33	159	20	5	1	7	125	1 ¹
		MBNMS29-Manta-11Jan18	60	207	43	7	6	4	146	1 ¹
		MBNMS29-Manta-30Mar2018	49	-	31	16	1	1		
		MBNMS29-Manta-DUP-30Mar2018	63	-	14	45	2	2		
		MBNMS30-Manta-31Mar2018	4 ¹	265	4 ¹	0	0	0	261	0
		MBNMS31-Manta-31Mar2018	7	-	7	0	0	0		
		MBNMS32-Manta-31Mar2018	10	-	10	0	0	0		

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation.

Appendices

Table A-4.3d. Manta trawl results (abundance) for marine sanctuary samples.

MANTA TRAWL RESULTS: ABUNDANCE (PARTICLES/KM ²) FOR MARINE SANCTUARY SAMPLES											
Location	Season	Sample ID	Area (km ²)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Sanctuary Samples	Dry Weather	CBNMS22-Manta-12Sept2017	0.0003221	6,208 ¹	3,104	0	6,208	117,959 ¹	0	15,521	133,480 ¹
		CBNMS23-Manta-12Sept2017	0.0010533	12,342	0	0	1,899	19,938 ¹	0	14,241	34,179 ¹
		CBNMS24-Manta-13Sept2017	0.0015268	0 ¹	0	0	0	62,222	0	0	62,222
		GFNMS25-Manta-11Sept2017	0.001397	1,432 ¹	716	0	0	181,820	0	2,147 ¹	183,967
		GFNMS26-Manta-12Sept2017	0.0008659	10,394	3,465	0	0			13,859	
		GFNMS26-Manta-DUP-12Sept2017	0.0007756	5,158 ¹	0	0	9,026	47,707 ¹	0	14,183	61,890 ¹
		GFNMS27-Manta-13Sept2017	0.0012947	772 ¹	0	0	0	57,928 ¹	1,545 ¹	772 ¹	60,245
		GFNMS28-Manta-13Sept2017	0.0013031	3,837	767	0	1,535	46,813 ¹	0	6,139	52,952 ¹
		MBNMS29-Manta-13Sept2017	0.0017388	2,300 ¹	0	0	0	16,103 ¹	0	2,300 ¹	18,404 ¹
		MBNMS30-Manta-27Sept2017	0.0013455	5,946	0	1,486	0	93,648	0	7,432	101,080
		MBNMS31-Manta-27Sept2017	0.0012254	2,448 ¹	0	1,632	0			4,080	
		MBNMS32-Manta-27Sept2017	0.0012315	1,624 ¹	0	812	0	38,165 ¹	812 ¹	2,436 ¹	41,413 ¹
		CBNMS22-Manta-30Mar2018	0.0014969	7,349	668	668	668			9,353	
		CBNMS23-Manta-29Mar2018	0.0015508	9,028	1,934	0	1,290			12,252	
		CBNMS24-Manta-29Mar2018	0.0018939	1,584 ¹	528	0	528			2,640	
Sanctuary Samples	Wet Weather	GFNMS25-Manta-30Mar2018	0.001376	19,622	0	2,180	0			21,802	
		GFNMS26-Manta-29Mar2018	0.0013023	6,911	0	0	1,536			8,447	
		GFNMS27-Manta-29Mar2018	0.0011921	5,872	839	839 ¹	839	222,291	0	8,388	230,679
		GFNMS28-Manta-30Mar2018	0.0016112	8,068	621	621 ¹	621			9,930	

MANTA TRAWL RESULTS: ABUNDANCE (PARTICLES/KM ²) FOR MARINE SANCTUARY SAMPLES											
Location	Season	Sample ID	Area (km ²)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Sanctuary Samples	Wet Weather	MBNMS-29-Manta-17Nov17	0.0012201	16,391	4,098	820	5,737	102,447	820 ¹	27,046	130,312
		MBNMS29-Manta-11Jan18	0.0015925	27,002	4,396	3,768	2,512	91,681	628 ¹	37,677	129,987
		MBNMS29-Manta-30Mar2018	0.0017066	18,165	9,375	586	586			28,712	
		MBNMS29-Manta-DUP-30Mar2018	0.0016031	8,733	28,070	1,248	1,248			39,298	
		MBNMS30-Manta-31Mar2018	0.0010168	3,934 ¹	0	0	0	256,688	0	3,934 ¹	260,622
		MBNMS31-Manta-31Mar2018	0.0010573	6,621	0	0	0			6,621	
		MBNMS32-Manta-31Mar2018	0.0013989	7,148	0	0	0			7,148	

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation

Appendices

Table A-4.3e. Manta trawl results (volume and concentration) for Bay samples.

MANTA TRAWL RESULTS: VOLUME (L) AND CONCENTRATION (PARTICLES/L) FOR BAY SAMPLES											
Location	Season	Sample ID	Volume (L)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Bay Samples	Dry Weather	CB4-Manta-21Aug2017	201,279	0.00002	0	0	0	0.00042	0	0.00002	0.00044
		CB5-Manta-22Aug2017	137,528	0.00033	0.00006	0	0.00006	-	-	0.00045	
		CB5-Manta-5Nov2017	67,445	0.00037	0.00006	0.00001	0.00009	0.00408	0.00001 ¹	0.00053	0.00463
		CB5-Manta-DUP-5Nov2017	109,876	0.00006	0.00018	0	0	0.00248	0	0.00025	0.00273
		CB6-Manta-22Aug2017	82,007	0.00007	0.00001	0.00001	0	0.00178	0.00002 ¹	0.0001	0.0019
		CB7-Manta-22Aug2017	211,171	0.00006	0.0007	0	0.00002	0.00165	0	0.00078	0.00243
		CB8-Manta-25Aug2017	175,432	0.00176	0.00038	0.00005	0.00008	-	-	0.00227	
		CB8-Manta-DUP-25Aug2017	179,610	0.00089	0.00085	0.00003	0.00008	-	-	0.00185	
		CB9-Manta-22Aug2017	152,777	0.00017	0	0.00001	0.00001	0.00018 ¹	0	0.0002	0.00037 ¹
		LSB14-Manta-24Aug2017	167,410	0.00073	0.00002	0.0001	0.00014	0.00082	0.00001 ¹	0.00099	0.00181
		LSB15-Manta-24Aug2017	124,083	0.0003	0.00002	0	0.00005	-	-	0.00037	
		LSB16-Manta-24Aug2017	99,559	0.00012	0.00004	0	0	-	-	0.00016	
		SB10-Manta-23Aug2017	146,311	0.00011	0.00001	0	0	0.00022 ¹	0.00001 ¹	0.00012	0.00036 ¹
		SB11-Manta-23Aug2017	178,438	0.00014	0.00005	0	0	0.00031 ¹	0.00006	0.00019	0.00056
		SB12-Manta-23Aug2017	557,440	0.0001	0	0.00001	0.00002	-	-	0.00013	
		SB13-Manta-23Aug2017	106,681	0.00256	0.0001	0.00005	0.00021	0.00041 ¹	0.00005 ¹	0.00292	0.00337
		Alcatraz-Manta-18Sept2017	95,428	0.00031	0.00002	0.00007	0.00003	-	-	0.00044	
		SPB2-Manta-21Aug2017	108,929	0.00006	0	0	0	0.00079	0.00001 ¹	0.00006	0.00086
		SPB3-Manta-21Aug2017	185,691	0.00044	0.00004	0.00001	0.00016	0.00348	0	0.00065	0.00413
		SPB1-Manta-21Aug2017	142,902	0.00001 ¹	0.00007	0	0	-	-	0.00008	

Appendices

MANTA TRAWL RESULTS: VOLUME (L) AND CONCENTRATION (PARTICLES/L) FOR BAY SAMPLES											
Location	Season	Sample ID	Volume (L)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Bay Samples	Wet Weather	CB4-Manta-16Nov2017	102,764	0.00463	0.0007	0.00067	0.00013	-	-	0.00613	
		CB5-Manta-16Nov2017	133,796	0.0002	0.00156	0.00011	0.00002	-	-	0.0019	
		CB6-Manta-16Nov2017	126,039	0.00021	0.00019	0.00002	0.00006	-	-	0.00048	
		CB7-Manta-16Nov2017	136,451	0.00025	0.00036	0.00008	0.00005	0.00328	0	0.00074	0.00402
		CB7-Manta-DUP-16Nov2017	128,450	0.00062	0.00076	0.00035	0.00002	0.00149	0	0.00174	0.00323
		CB8-Manta-11Jan18	282,326	0.00359	0.00056	0.00004	0.00017	0.00497	0.00015	0.00435	0.00947
		CB9-Manta-11Jan18	75,435	0.05212	0.00831	0.00319	0.00211	-	-	0.06574	
		LSB14-Manta-06Mar2018	92,559	0.00022	0	0	0.00006	-	-	0.00028	
		LSB15-Manta-06Mar2018	87,425	0.00025	0	0	0.00001	-	-	0.00026	
		LSB15-Manta-DUP-06Mar2018	92,556	0.00078	0	0	0.00006	-	-	0.00084	
		LSB16-Manta-06Mar2018	90,061	0.0004	0	0.00001	0.00012	-	-	0.00053	
		SB10-Manta-19Mar2018	55,584	0.00011	0.00009	0.00004	0.00002	0.00756	0	0.00025	0.00781
		SB11-Manta-19Mar2018	135,953	0.00225	0.00009	0.00004	0.00024	0.00383	0	0.00262	0.00645
		SB12-Manta-19Mar2018	115,820	0.00037	0.00008	0.00003	0.00002	-	-	0.0005	
		SB12-Manta-DUP-19Mar2018	94,928	0.00038	0.00061	0.00015	0.00002	0.01828	0.00003 ¹	0.00116	0.01947
		SB13-Manta-19Mar2018	84,868	0.00088	0.00025	0	0.00027	0.00326	0	0.0014	0.00467
		SPB2-Manta-16Nov2017	125,002	0.00008	0	0	0.00002	0.00248	0	0.0001	0.00258
		SPB3-Manta-17Nov17	149,338	0.00133	0	0.00003	0.00006	-	-	0.00142	
		SUB1-Manta-17Nov17	127,544	0.00013	0.00001	0	0.00001	-	-	0.00014	

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation

Appendices

Table A-4.3f. Manta trawl results (volume and concentration) for marine sanctuary samples.

MANTA TRAWL RESULTS: VOLUME (L) AND CONCENTRATION (PARTICLES/L) FOR MARINE SANCTUARY SAMPLES											
Location	Season	Sample ID	Volume (L)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle	Total (w/o fibers)	Total (w/ fibers)
Sanctuary samples	Dry Weather	CBNMS22-Manta-12Sept2017	30,604	0.00007 ¹	0.00003	0	0.00007	0.00124 ¹	0	0.00016	0.00141 ¹
		CBNMS23-Manta-12Sept2017	100,062	0.00013	0	0	0.00002	0.00021 ¹	0	0.00015	0.00036 ¹
		CBNMS24-Manta-13Sept2017	145,045	0 ¹	0	0	0	0.00065	0	0	0.00065
		GFNMS25-Manta-11Sept2017	132,714	0.00002 ¹	0.00001	0	0	0.00191	0	0.00002 ¹	0.00194
		GFNMS26-Manta-12Sept2017	82,256	0.00011	0.00004	0	0	-	-	0.00015	
		GFNMS26-Manta-DUP-12Sept2017	73,679	0.00005 ¹	0	0	0.0001	0.0005 ¹	0	0.00015	0.00065 ¹
		GFNMS27-Manta-13Sept2017	122,998	0.00001 ¹	0	0	0	0.00061 ¹ 0.00002 ¹	0.00001 ¹	0.00063	
		GFNMS28-Manta-13Sept2017	123,791	0.00004	0.00001	0	0.00002	0.00049 ¹	0	0.00006	0.00056 ¹
		MBNMS29-Manta-13Sept2017	165,182	0.00002 ¹	0	0	0	0.00017 ¹	0	0.00002 ¹	0.00019 ¹
		MBNMS30-Manta-27Sept2017	127,819	0.00006	0	0.00002	0	0.00099	0	0.00008	0.00106
Sanctuary samples	Wet Weather	MBNMS31-Manta-27Sept2017	116,409	0.00003 ¹	0	0.00002	0	-	-	0.00004	
		MBNMS32-Manta-27Sept2017	116,993	0.00002 ¹	0	0.00001	0	0.0004 ¹ 0.00001 ¹	0.00003 ¹	0.00044 ¹	
		CBNMS22-Manta-30Mar2018	142,206	0.00008	0.00001	0.00001	0.00001	-	-	0.0001	
		CBNMS23-Manta-29Mar2018	147,326	0.0001	0.00002	0	0.00001	-	-	0.00013	
		CBNMS24-Manta-29Mar2018	179,922	0.00002 ¹	0.00001	0	0.00001	-	-	0.00003	
		GFNMS25-Manta-30Mar2018	130,722	0.00021	0	0.00002	0	-	-	0.00023	
		GFNMS26-Manta-29Mar2018	123,717	0.00007	0	0	0.00002	-	-	0.00009	
Sanctuary samples	Wet Weather	GFNMS27-Manta-29Mar2018	113,252	0.00006	0.00001	0.00001	0.00001	0.00234	0	0.00009	0.00243
		GFNMS28-Manta-30Mar2018	153,066	0.00008	0.00001	0.00001	0.00001	-	-	0.0001	
		MBNMS-29-Manta-17Nov17	115,914	0.00017	0.00004	0.00001	0.00006	0.00108 0.00001 ¹	0.00028	0.00137	

Appendices

Sanctuary samples	Wet Weather	MBNMS29-Manta-11Jan18	151,285	0.00028	0.00005	0.00004	0.00003	0.00097	0.00001 ¹	0.0004	0.00137
		MBNMS29-Manta-30Mar2018	162,127	0.00019	0.0001	0.00001	0.00001	-	-	0.0003	
		MBNMS29-Manta-DUP-30Mar2018	152,297	0.00009	0.0003	0.00001	0.00001	-	-	0.00041	
		MBNMS30-Manta-31Mar2018	96,596	0.00004 ¹	0	0	0	0.0027	0	0.00004	0.00274
		MBNMS31-Manta-31Mar2018	100,439	0.00007	0	0	0	-	-	0.00007	
		MBNMS32-Manta-31Mar2018	132,898	0.00008	0	0	0	-	-	0.00008	

¹ Numbers highlighted in pink are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation

Table A-4.4. Manta trawl results (particle counts) for field and laboratory blanks: calculation of the threshold for data qualification (average of blanks plus two times the standard deviation).

		Count (count/sample)								
		Sample ID	Total particle (w/o fibers)	Total particle (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
Field Blanks	SPB3-Manta-Blank-21Aug2017	0	69	0	0	0	0	66	3	
	SB13-Manta-Blank-23Aug2017	6	53	6	0	0	0	43	4	
	GFNMS25-Manta-Blank-11Sept2017	3	58	2	1	0	0	47	8	
	SPB3-Manta-Blank-17Nov17	0	-	0	0	0	0	-	-	
	MBNMS29-Manta-BLANK-11Jan18	2	14	2	0	0	0	12	0	
	CB9-Manta-Blank-11Jan18	2	-	2	0	0	0	-	-	
	LSB14-Manta-Blank-06Mar2018	3	-	3	0	0	0	-	-	
	CBNMS22-Manta-Blank-30Mar2018	0	-	0	0	0	0	-	-	
Field Blank Stats	Average	2	48.5	1.9	0.1	0	0	42	3.8	
	StDev	2.1	24	2	0.4	0	0	22.4	3.3	
	Average + 2*StDev	6.1	96.4	5.9	0.8	0	0	86.8	10.4	
Lab Blanks	Manta_LabBlank1	0	1	0	0	0	0	1	0	
	Manta_LabBlank2	0	2	0	0	0	0	2	0	
	Manta_LabBlank3	0	2	0	0	0	0	2	0	
	Manta_LabBlank4	0	-	0	0	0	0	-	-	
	Manta_LabBlank5	0	-	0	0	0	0	-	-	
	Manta_LabBlank6	0	-	0	0	0	0	-	-	
	Manta_LabBlank7	0	-	0	0	0	0	-	-	

Appendices

		Count (count/sample)								
		<i>Sample ID</i>	Total particle (w/o fibers)	Total particle (w/ fibers)	Fragment	Foam	Sphere	Film	Fiber	Fiber Bundle
<i>LB Stats</i>	Average		0	1.7	0	0	0	0	1.7	0
	StDev		0	0.6	0	0	0	0	0.6	0
	Average + 2*StDev		0	2.8	0	0	0	0	2.8	0
<i>All Blank Stats</i>	Average		1.1	28.4	1	0.1	0	0	24.7	2.1
	StDev		1.7	30.2	1.7	0.2	0	0	26.7	3.1
	Average + 2*StDev		4.5	88.9	4.3	0.6	0	0	78.2	8.3

Appendices

Table A-4.5a. One-liter grab results (count and concentration) for Bay samples.

Location	Season	Sample ID	1 Liter Results (Count)				1 Liter Results (Concentration)	
			Fibers	Film	Foam	Fragment	Total Particles	Total Concentration (particles/L)
Bay Samples	Dry Weather	CB-4-Nano-21Aug2017	5 ¹				5 ¹	4.95 ¹
		CB5-Nano-22Aug2017	4 ¹	2		3	9 ¹	8.92 ¹
		CB6-Nano-22Aug2017	3 ¹				3 ¹	3.2 ¹
		CB7-Nano-1-22Aug2017	2 ¹			1 ¹	3 ¹	3.22 ¹
		CB7-Nano-DUP-22Aug2017	3 ¹				3 ¹	3.05 ¹
		CB8-Nano-25Aug2017	1 ¹	2			3 ¹	2.86 ¹
		CB9-Nano-22Aug2017			1	5	6 ¹	6.58 ¹
		LSB16-Nano-24Aug2017	6 ¹	1 ¹		2 ¹	9 ¹	8.57 ¹
		SB10-Nano-23Aug2017	3 ¹				3 ¹	2.99 ¹
		SB11-Nano-23Aug2017	3 ¹	1 ¹			4 ¹	3.82 ¹
Bay Samples	Wet Weather	SB13-Nano-23Aug2017	6 ¹				6 ¹	6.51 ¹
		SPB1-Nano-21Aug2017	4 ¹				4 ¹	4.74 ¹
		SPB2-NANO-21Aug2017	4 ¹			1 ¹	5 ¹	5 ¹
		SPB3-Nano-21Aug2017	7 ¹	1 ¹			8 ¹	9.39 ¹
		CB4-Nano-16Nov2017	6 ¹				6 ¹	5.79 ¹
		CB5-Nano-16Nov2017	3 ¹				3 ¹	2.82 ¹
		CB6-Nano-16Nov2017	8 ¹				8 ¹	7.48 ¹
		CB7-Nano-16Nov2017	2 ¹				2 ¹	1.9 ¹
		CB8-Nano-11Jan18	3 ¹				3 ¹	3.06 ¹

Appendices

Location	Season	Sample ID	1 Liter Results (Count)				1 Liter Results (Concentration)	
			Fibers	Film	Foam	Fragment	Total Particles	Total Concentration (particles/L)
Bay Samples	Wet Weather	CB9-Nano-11Jan18	15	1 ¹		5	21	21.83
		LSB14-Nano-06Mar2018	3 ¹			2 ¹	5 ¹	4.89 ¹
		LSB15-Nano-06Mar2018	2 ¹				2 ¹	1.92 ¹
		LSB16-Nano-06Mar2018	2 ¹				2 ¹	2.63 ¹
		SB10-Nano-19Mar2018	2 ¹	3		1 ¹	6 ¹	6.91 ¹
		SB11-Nano-19Mar2018	2 ¹	1 ¹			3 ¹	3.19 ¹
		SB12-Nano-19Mar2018	4 ¹	1 ¹			5 ¹	5 ¹
		SB13-Nano-19Mar2018	12				12	11.32
		SPB1-Nano-17Nov17	6 ¹				6 ¹	5.69 ¹
		SPB2-Nano-16Nov2017	4 ¹	1 ¹			5 ¹	5 ¹
		SPB3-Nano-17Nov17	6 ¹				6 ¹	5.72 ¹

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation

Appendices

Table A-4.5b. One-liter grab results (count and concentration) for marine sanctuary samples.

Location	Season	Sample ID	1 Liter Results (Count)			1 Liter Results (Concentration)			
			Fibers	Film	Foam	Fragment	Total Particles	Total Concentration (particles/L)	Fiber Concentration (fibers/L)
Dry Weather	Sanctuary Samples	CBNMS22-Nano-12Sept2017	3 ¹			1 ¹	4 ¹	4 ¹	3 ¹
		CBNMS23-Nano-12Sept2017	2 ¹			1 ¹	3 ¹	2.95 ¹	1.97 ¹
		CBNMS24-Nano-13Sept2017	10 ¹				10 ¹	10.41 ¹	10.41 ¹
		GFNMS25-Nano-11Sept2017	1 ¹			1 ¹	2 ¹	1.9 ¹	0.95 ¹
		GFNMS26-Nano-12Sept2017	1 ¹				1 ¹	1.06 ¹	1.06 ¹
		GFNMS26-Nano-DUP-12Sept2017	2 ¹				2 ¹	2.23 ¹	2.23 ¹
		GFNMS27-Nano-13Sept2017	3 ¹				3 ¹	2.84 ¹	2.84 ¹
		GFNMS28-Nano-13Sept2017	2 ¹	2		1 ¹	5 ¹	4.78 ¹	1.91 ¹
		MBNMS29-Nano-13Sept2017	4 ¹				4 ¹	4.16 ¹	4.16 ¹
		MBNMS31-Nano-27Sept2017	7 ¹				7 ¹	6.69 ¹	6.69 ¹
Wet Weather	Sanctuary Samples	MBNMS30-Nano-27Sept2017	3 ¹				3 ¹	2.89 ¹	2.89 ¹
		MBNMS32-Nano-27Sept2017	5 ¹	1 ¹		2 ¹	8 ¹	7.75 ¹	4.84 ¹
		GFNMS25-Nano-30Mar2018	1 ¹				1 ¹	0.95 ¹	0.95 ¹
		GFNMS26-Nano-29Mar2018	10 ¹	3		4	17	16.1	9.47 ¹
		GFNMS28-Nano-30Mar2018	4 ¹				4 ¹	3.81 ¹	3.81 ¹
		MBNMS-29-Nano-17Nov17	36	3		3	42	39.29	33.68
		MBNMS29-Nano-11Jan18	1 ¹				1 ¹	0.98 ¹	0.98 ¹
		MBNMS29-Nano-31Mar2018				1 ¹	1 ¹	0.94 ¹	0 ¹
		MBNMS30-Nano-31Mar2018	4 ¹				4 ¹	3.77 ¹	3.77 ¹
		MBNMS30-Nano-DUP-31Mar2018	7 ¹			1 ¹	8 ¹	7.56 ¹	6.62 ¹
		MBNMS31-Nano-31Mar2018	2 ¹	1 ¹		3	6 ¹	5.69 ¹	1.9 ¹
		MBNMS32-Nano-31Mar2018	10 ¹				6	16	15.27
									9.54 ¹

¹ Numbers are below the threshold determined by the average field and laboratory blanks multiplied by two times the standard deviation

Appendices

Table A-4.6. Field blanks and laboratory blanks: calculation of threshold for data qualification (average of field and lab blanks plus two times the standard deviation).

	<i>Sample ID</i>	<i>Count (count/sample)</i>				
		<i># Fibers</i>	<i># Film</i>	<i># Foam</i>	<i># Fragment</i>	<i>Total Particles</i>
Field Blanks	GFNMS25-Nano-Blank-11Sept2017	8	0	0	0	8
	SB13-Nano-Blank-19Mar2018	2	0	0	0	2
	SB13-Nano-BLANK-23Aug2017	1	0	0	0	1
	LabBlank-1	1	1	0	0	2
	LabBlank-2	1	0	0	0	1
Lab Blanks	LabBlank-3	2	0	0	0	2
	LabBlank-4	8	1	0	1	10
	LabBlank-5	8	0	0	0	8
	LabBlank-6 (redone)	3	0	0	3	6
	Average Blanks	3.8	0.2	0.0	0.4	4.4
Stats	StDev Blanks	6.5	0.9	0.0	2.0	7.1
	Average + 2*StDev	10.2	1.1	0.0	2.5	11.5

Sediment

Table A-5.1. Mass of sediment analyzed for microparticles and microplastics.

Sample ID	Mass (g dw)
17MMP-S-TB102-MP	150.2
17MMP-S-SUB53-MP	150.1
17MMP-S-SB056-MP	150
17MMP-S-SB074-MP	150
17MMP-S-SPB128-MP	150
RMP-14SC-1153	150
17MMP-S-TB101-MP	149.9
15RMPMC-CB32-MP1	118.1
15RMPMC-CB15-MP1	95.7
17MMP-S-SPB15-MP-1	95
15RMPMC-CB37-MP1	93.2
17MMP-S-SPB15-MP-2	81
17MMP-S-SB051-MP	66.6
RMP-14SC-1270	47.4
17MMP-S-SUB52-MP	44
15RMPMC-CB10-MP1	43.4
17MMP-S-LSB04-MP	29.2
17MMP-S-SOSL16-MP-2	25.7
17MMP-S-SOSL16-MP-1	24
17MMP-S-LSB06-MP	23
17MMP-S-LSB02-MP-1	16
17MMP-S-SOSL40-MP	4.4

Appendices

Table A-5.2. Microparticles counts in sediment samples (microparticles/sample). Average of field duplicates are reported at SOSL16 and SPB15.

Site	Subembayment	Site Type	Fiber	Fiber Bundle	Film	Foam	Fragment	Sphere	Total
LSB02	Lower South Bay	Margin - Wastewater	782	2	13	10	124	32	963
SOSL40	Southern Sloughs	Margin - Stormwater and Wastewater	122 ¹	0	4	1	32	3	162 ¹
LSB06	Lower South Bay	Margin - Stormwater	284	2	0	0	29	15	330
SOSL16	Southern Sloughs	Margin - Stormwater and Wastewater	215.5	0	10.5	11.5	97	13	347.5
LSB04	Lower South Bay	Margin	235	0	5	7	79	17	343
CB15	Central Bay	Margin - Stormwater	280	0	6	2	116	1	405
CB37	Central Bay	Margin - Stormwater	68 ¹	0	29	6	255	5	363 ¹
CB10	Central Bay	Margin - Stormwater	83 ¹	0	10	7	62	1	163 ¹
SPB15	San Pablo Bay	Margin - Stormwater	199	1.5	17.5	17.5	77.5	9.5	322.5
CB32	Central Bay	Margin - Stormwater	183	0	7	8	186	0	384
SB002S	South Bay	Ambient	113 ¹	0	0	0	15	0	128 ¹
SUB52	Suisun Bay	Margin	110 ¹	1	0	0	6 ¹	1	118 ¹
SB051	South Bay	Margin	111 ¹	0	1 ¹	0	62	1	175 ¹
CB001S	Central Bay	Ambient	210	0	8	1	44	1	264
SPB128	San Pablo Bay	Margin - Stormwater	109 ¹	0	0	0	94	0	203 ¹
TB102	Tomales Bay	Reference Site	101 ¹	0	1 ¹	8	15	0	125 ¹
SB074	South Bay	Margin - Stormwater	112 ¹	0	0	0	2 ¹	6	120 ¹
TB101	Tomales Bay	Reference Site	97 ¹	0	0	0	19	0	116 ¹
SB056	South Bay	Margin - Stormwater	67 ¹	0	0	0	8 ¹	0	75 ¹
SUB53	Suisun Bay	Margin - Wastewater	56 ¹	0	0	0	19	0	75 ¹
LSB04-FB	Lower South Bay	Margin (Field blank)	143	0	1	0	7	0	151
LABQA	LABQA	LABQA	17	0	0	0	1	0	18
LABQA	LABQA	LABQA	1	0	0	0	0	0	1
LABQA	LABQA	LABQA	0	0	0	0	0	0	0

¹ Flagged samples are below the conservative data qualification threshold for each morphology. Total microparticle counts are flagged if any of the contributing morphology counts are flagged.

Appendices

Table A-5.3. Microparticles concentrations in sediment samples (microparticles/g dw).

Site	Subembayment	Site Type	Fiber	Fiber Bundle	Film	Foam	Fragment	Sphere	Non-fiber
LSB02	Lower South Bay	Margin - Wastewater	49	0.1	0.8	0.6	7.8	2.0	11.2
SOSL40	Southern Sloughs	Margin - Stormwater and Wastewater	27.7 ¹	0	0.9	0.2	7.3	0.7	9.1
LSB06	Lower South Bay	Margin - Stormwater	12.3	0.1	0	0	1.3	0.7	1.9
SOSL16	Southern Sloughs	Margin - Stormwater and Wastewater	8.7	0	0.4	0.5	3.9	0.5	5.4
LSB04	Lower South Bay	Margin	8.0	0	0.2	0.2	2.7	0.6	3.7
CB15	Central Bay	Margin - Stormwater	2.9	0	0.1	0.02	1.2	0.01	1.3
CB37	Central Bay	Margin - Stormwater	0.7 ¹	0	0.3	0.06	2.7	0.05	3.2
CB10	Central Bay	Margin - Stormwater	1.9 ¹	0	0.2	0.2	1.4	0.02	1.8
SPB15	San Pablo Bay	Margin - Stormwater	2.2	0	0.2	0.2	0.9	0.1	1.4
CB32	Central Bay	Margin - Stormwater	1.5	0	0.1	0.07	1.6	0	1.7
SB002S	South Bay	Ambient	2.4 ¹	0	0	0	0.3	0	0.3
SUB52	Suisun Bay	Margin	2.5 ¹	0	0	0	0.1 ¹	0.02	0.1 ¹
SB051	South Bay	Margin	1.7 ¹	0	0.02 ¹	0	0.9	0.02	0.9
CB001S	Central Bay	Ambient	1.4	0	0.05	0.007	0.3	0.007	0.4
SPB128	San Pablo Bay	Margin - Stormwater	0.7 ¹	0	0	0	0.6	0	0.6
TB102	Tomales Bay	Reference Site	0.7 ¹	0	0.007 ¹	0.05	0.1	0	0.2
SB074	South Bay	Margin - Stormwater	0.7 ¹	0	0	0	0.01 ¹	0.04	0.05 ¹
TB101	Tomales Bay	Reference Site	0.6 ¹	0	0	0	0.1	0	0.1
SB056	South Bay	Margin - Stormwater	0.4 ¹	0	0	0	0.05 ¹	0	0.05 ¹
SUB53	Suisun Bay	Margin - Wastewater	0.4 ¹	0	0	0	0.1	0	0.1

¹ Flagged samples are below the conservative data qualification threshold for each morphology.

Appendices

Table A-5.4a. Number of microparticles and microplastics in sediment.

	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>
Plastic	90	126	50	22	52
Anthropogenic unknown	39	13	1	4	0
Natural-based	139	19	5	9	1
Unknown	20	161	21	37	17
Not identified (no spectroscopy)	3672	1197	63	36	58
Total	3960	1516	140	108	128

Table A-5.4b. Percentage represents composition of particles in each morphology that underwent spectroscopy. For example, Plastic Fiber % indicates that among fibers that underwent spectroscopy, 31% were plastic.

	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>
Plastic	31%	39%	65%	31%	74%
Anthropogenic unknown	14%	4%	1%	6%	0%
Natural-based	48%	6%	6%	13%	1%
Unknown	7%	50%	27%	51%	24%

Table A-5.5a. Number of microparticles identified among most commonly identified plastic polymers. Other identified compositions include other plastic polymers, non-plastic materials, and unknown.

	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>
Polystyrene	2	18	10	9	52
Polyethylene	0	31	11	5	0
Polypropylene	8	23	4	2	0
Polyester/Polyethylene terephthalate	28	7	5	0	0
Acrylic	14	10	2	1	0
Cellulose acetate	20	4	1	0	0
Unknown potentially rubber	0	126	0	0	0

Appendices

Table A-5.5b. Percentage represents composition of particles in each morphology that underwent spectroscopy for the most commonly identified plastic polymers. For example, polystyrene sphere % indicates that among sphere microparticles that underwent spectroscopy, 74% were polystyrene. Other identified compositions include other plastic polymers, non-plastic materials, and unknown.

	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>
Polystyrene	1%	6%	13%	13%	74%
Polyethylene	0%	10%	14%	7%	0%
Polypropylene	3%	7%	5%	3%	0%
Polyester/Polyethylene terephthalate	10%	2%	6%	0%	0%
Acrylic	5%	3%	3%	1%	0%
Cellulose acetate	7%	1%	1%	0%	0%
Unknown potentially rubber	0%	39%	0%	0%	0%

Prey Fish

Table A-6.1. Total microparticle counts in fish samples from San Francisco Bay and Tomales Bay.

Region and Species	Total Microparticles	Fiber	Fragment	Film	Foam	Sphere
San Francisco Bay Anchovies (n = 54)	752	667	76	8	1	0
San Francisco Bay Topsmelt (n = 58)	869	723	106	38	1	1
San Francisco Bay Anchovies + Topsmelt (n = 112)	1,621	1,390	182	46	2	1
Tomales Bay Anchovies (n = 20)	96	91	3	2	0	0
Tomales Bay Topsmelt (n = 20)	202	192	8	2	0	0
Tomales Bay Anchovies + Topsmelt (n = 40)	298	283	11	4	0	0
San Francisco Bay and Tomales Bay Anchovies + Topsmelt (n = 152)	1,919	1,673	193	50	2	1

Table A-6.2. Average microparticles/fish in each species from San Francisco Bay and Tomales Bay (no blank correction).

Region and Species	Total Microparticles	Fiber	Fragment	Film	Foam	Sphere
San Francisco Bay Anchovies (n = 54)	13.9	12.4	1.4	0.1	0.02	0.0
San Francisco Bay Topsmelt (n = 58)	15	12.5	1.8	0.7	0.02	0.02
San Francisco Bay Anchovies + Topsmelt (n = 112)	15	12	1.6	0.4	0.02	0.02
Tomales Bay Anchovies (n = 20)	4.8	4.6	0.2	0.1	0	0
Tomales Bay Topsmelt (n = 20)	10.1	9.6	0.4	0.1	0	0
Tomales Bay Anchovies + Topsmelt (n = 40)	7.5	7.1	0.3	0.1	0	0
San Francisco Bay and Tomales Bay Anchovies + Topsmelt (n = 152)	13	11	1.3	0.3	0	0

[Appendices](#)

Table A-6.3a. Number of microparticles and microplastics in San Francisco Bay anchovies.

Composition	Total Microparticles	Fiber	Fragment	Film	Foam	Sphere
Plastic	102	91	7	4	0	0
Anthropogenic unknown	278	271	4	3	0	0
Natural-based	56	56	0	0	0	0
Unknown	53	17	34	1	1	0
Not identified (no spectroscopy)	263	232	31	0	0	0
Total	752	667	76	8	1	0

Table A-6.3b. Percentage represents composition of particles in each morphology that underwent spectroscopy. For example Plastic Fiber % represents among fibers that underwent spectroscopy, 21% were plastic. NA = not applicable.

Composition	Fiber	Fragment	Film	Foam	Sphere	Nonfiber (Fragment + Film + Foam + Sphere)
Plastic	21%	16%	50%	0%	NA	20%
Anthropogenic unknown	62%	9%	38%	0%	NA	13%
Natural-based	13%	0%	0%	0%	NA	0%
Unknown	4%	76%	13%	100%	NA	67%

Table A-6.4a. Number of microparticles and microplastics in San Francisco Bay topsmelt.

Composition	Total Microparticles	Fiber	Fragment	Film	Foam	Sphere
Plastic	134	109	17	6	1	1
Anthropogenic unknown	280	242	16	22	0	0
Natural-based	48	42	3	3	0	0
Unknown	80	32	46	2	0	0
Not identified (no spectroscopy)	327	298	24	5	0	0
Total	869	723	106	38	1	1

Table A-6.4b. Percentage represents composition of particles in each morphology that underwent spectroscopy. For example Plastic Fiber % represents among fibers that underwent spectroscopy, 26% were plastic.

Composition	Fiber	Fragment	Film	Foam	Sphere	Nonfiber (Fragment + Film + Foam + Sphere)
Plastic	26%	21%	18%	100%	100%	21%
Anthropogenic unknown	57%	20%	67%	0%	0%	32%
Natural-based	10%	4%	9%	0%	0%	5%
Unknown	8%	56%	6%	0%	0%	41%

[Appendices](#)

Table A-6.5a. Number of microparticles and microplastics in San Francisco Bay fish (anchovies and topsmelt).

<i>Composition</i>	<i>Total Microparticles</i>	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>
Plastic	236	200	24	10	1	1
Anthropogenic unknown	558	513	20	25	0	0
Natural-based	104	98	3	3	0	0
Unknown	133	49	80	3	1	0
Not identified (no spectroscopy)	590	530	55	5	0	0
Total	1,621	1,390	182	46	2	1

Table A-6.5b. Percentage represents composition of particles in each morphology that underwent spectroscopy. For example Plastic Fiber % represents among fibers that underwent spectroscopy, 23% were plastic.

<i>Composition</i>	<i>Fiber</i>	<i>Fragment</i>	<i>Film</i>	<i>Foam</i>	<i>Sphere</i>	<i>Nonfiber (Fragment + Film + Foam + Sphere)</i>
Plastic	23%	19%	24%	50%	100%	21%
Anthropogenic unknown	60%	16%	61%	0%	0%	26%
Natural-based	11%	2%	7%	0%	0%	4%
Unknown	6%	63%	7%	50%	0%	49%