CLEAN WATER / RMP

NUMBER 741
JULY 2016

Pollutants of concern (POC) loads monitoring progress report, water years (WYs) 2012, 2013, and 2014

Prepared by:
Alicia Gilbreath, Jennifer Hunt, Jing Wu, Patrick Kim, and Lester McKee

For
Bay Area Stormwater Management Agencies Association (BASMAA)
Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) Sources Pathways and Loadings Workgroup (SPLWG) Small Tributaries Loading Strategy (STLS)

SAN FRANCISCO ESTUARY INSTITUTE | AQUATIC SCIENCE CENTER
4911 Central Avenue, Richmond, CA 94804 • p: 510-746-7334 (SFEI) • f: 510-746-7300 • www.sfei.org

Preface

This project was completed with funding provided by the Bay Area Stormwater Management Agencies Association (BASMAA) and the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP). This and two prior drafts of this report were reviewed by representatives of BASMAA prior to being submitted to the San Francisco Regional Water Quality Control Board (Water Board) in March 2013, 2014, and 2015 as an annual report for compliance with the Municipal Regional Stormwater NPDES permit (MRP). As such, three earlier versions of this report can also be downloaded from the Water Board website at:
http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/stormwater/Municipal/index.shtml

This final progress report builds upon the two prior annual draft reports and summarizes the results from the entire three winter seasons of monitoring.

Acknowledgements

We appreciate the support and guidance from members of the Sources, Pathways and Loadings Workgroup of the Regional Monitoring Program for Water Quality in San Francisco Bay. The detailed work plan behind this work was developed through the Small Tributaries Loading Strategy (STLS) during a series of meetings in the summer of 2011. Local members on the STLS at that time were Arleen Feng, Lucy Buchan, Khalil Abusaba, Jamison Crosby, and Chris Sommers (for BASMAA); and Richard Looker, Jan O'Hara, and Tom Mumley (for the Water Board). Project assistance during planning and implementation of the field components was also provided by ADH Environmental (Lower Marsh Creek and San Leandro Creek), Balance Hydrologics (Guadalupe River), and Kinnetic Laboratories, Inc. (Pulgas Pump Station - South) as contractors for ACCWP, CCCWP, SCVURPPP and SMCWPPP, respectively. SFEI field and logistical support over the three-year project was provided by David Gluchowski, Emily Novick, April Robinson, Patrick Kim, Don Yee, and Rachel Eastman. SFEI's data management team is acknowledged for their diligent delivery of quality assured well-managed data. Over the three-year duration of this project, this team included: Cristina Grosso, Amy Franz, John Ross, Don Yee, Adam Wong, and Michael Weaver. Pete Wilde (ADH Environmental), Chris Sommers, Bonnie DeBerry, Lisa Sabin (EOA Inc.), Arleen Feng (ACCWP), David Schoellhamer (USGS) and Kelly Moran (TDC Environmental) provided helpful written reviews of this report.

This progress report can be cited as:

Gilbreath, A.N., Hunt, J.A., Wu, J., Kim, P.S., and McKee, L.J., 2016. Pollutants of concern (POC) loads monitoring progress report, water years (WYs) 2012, 2013, and 2014. 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). Contribution No. 741. San Francisco Estuary Institute, Richmond, California.

Contents

1. Introduction

The San Francisco Regional Water Quality Control Board (Water Board) has determined that San Francisco Bay is impaired by mercury and PCBs due to threats to wildlife and human consumers of fish from the Bay. These contaminants persist in the environment and accumulate in aquatic food webs (SFRWRCB 2006; SFRWRCB, 2008). The Water Board has identified urban runoff from local watersheds as a pathway for pollutants of concern into the Bay, including mercury and PCBs. The Municipal Regional Stormwater Permit (MRP; SFRWRCB, 2009) contains several provisions requiring studies to measure local watershed loads of suspended sediment (SS), total organic carbon (TOC), polychlorinated biphenyl (PCB), total mercury (HgT), total methylmercury (MeHgT), nitrate-N (NO3), phosphate-P (PO4), and total phosphorus (TP) (provision C.8.e), as well as other pollutants covered under provision C.14. (e.g. legacy pesticides, PBDEs, and selenium).

Four Bay Area Stormwater Programs[1], represented by the Bay Area Stormwater Management Agencies Association (BASMAA), collaborated with the San Francisco Bay Regional Monitoring Program (RMP) to develop an alternative strategy allowed by Provision C.8.e of the MRP, known as the Small Tributaries Loading Strategy (STLS) (SFEI, 2009). An early version of the STLS provided an initial outline of the general strategy and activities to address four key management questions (MQs) that are found in MRP provision C.8.e:

> MQ1. Which Bay tributaries (including stormwater conveyances) contribute most to Bay impairment from POCs;
>
> MQ2. What are the annual loads or concentrations of POCs from tributaries to the Bay;
>
> MQ3. What are the decadal-scale loading or concentration trends of POCs from small tributaries to the Bay; and,
>
> MQ4. What are the projected impacts of management actions (including control measures) on tributaries and where should these management actions be implemented to have the greatest beneficial impact.

Since then, a Multi-Year-Plan (MYP) has been written (BASMAA, 2011) and updated twice (BASMAA, 2012; BASMAA, 2013). The MYP provides a comprehensive description of activities that will be implemented over the next 5-10 years to provide information and comply with the MRP. The MYP provides rationale for the methods and locations of proposed activities to answer the four MQs listed above. Activities include modeling using the regional watershed spreadsheet model (RWSM) to estimate regional scale loads (Lent and McKee, 2011; Lent et al., 2012; McKee et al., 2014), and pollutant characterization and loads monitoring in local tributaries beginning Water Year (WY) 2011 (McKee et al.,

[1] Alameda Countywide Clean Water Program, Contra Costa Clean Water Program, San Mateo Clean Water Pollution Prevention Program and Santa Clara Urban Runoff Pollution Prevention Program conduct monitoring and other activities on behalf of MRP Permittees in the four largest Bay Area counties.

2012), that continued in WY 2012 (McKee et al., 2013), WY 2013 (Gilbreath et al., 2014), and was largely completed in WY 2014 (this report).

The purpose of this report is to describe data collected during all three WYs (2012, 2013, and 2014) in compliance with MRP provision C.8.e., following the standard report content described in provision C.8.g.vi. The study design (selected watersheds and sampling locations, analytes, sampling methodologies and frequencies) as outlined in the MYP was developed to assess concentrations and loads in watersheds that are considered to likely be important watersheds in relation to sensitive areas of the Bay margin (MQ1):

- Lower Marsh Creek (Hg);
- North Richmond Pump Station (Hg and PCBs);
- San Leandro Creek below Chabot dam (Hg);
- Guadalupe River (Hg and PCBs);
- East Sunnyvale Channel (PCBs); and
- Pulgas Pump Station - South (PCBs).

Loads monitoring provides verification data for the RWSM (MQ2), and is intended to provide baseline data to assess long term loading trends (MQ3) in relation to management actions (MQ4). This report was structured to allow annual updates after each subsequent winter season of data collection. It should be noted that the sampling design described in this report (and modeling design: Lent and McKee, 2011; Lent et al., 2012; McKee et al., 2014) was focused mainly on addressing MQ2. During the next permit term (perhaps beginning in 2015), there will be an increasing focus towards finding high leverage watersheds and source areas within watersheds (MQ 1) for management focus (MQ4). A parallel report (the “POC synthesis report” (McKee et al., 2015)) is intended to document progress to date towards addressing management questions and the rationale for changed monitoring design going forward that more carefully addresses MQ1 and MQ4.

2. Field methods

2.1. Watershed physiography, sampling locations, and sampling methods

The San Francisco Bay estuary is surrounded by nine highly urbanized counties with a total population greater than seven million people (US Census Bureau, 2010). Although urban runoff from upwards of 300 small tributaries (note the number is dependent upon how the areas are lumped or split) flowing from the adjacent landscape represents only about 6% of the total freshwater input to the San Francisco Bay, this input has broadly been identified as a significant source of pollutants of concern (POCs) to the estuary (Davis et al., 2007; Oram et al., 2008; Davis et al., 2012; Gilbreath et al., 2012). Four watershed sites were sampled in WY 2012 and two additional watershed sites were added in WY 2013 and WY 2014 (Figure 1; Table 1). The sites were distributed throughout the counties where load monitoring was required by the MRP. The selected watersheds include areas with urban and industrial land uses, watersheds where stormwater programs are planning enhanced management actions to reduce PCB and mercury discharges, and watersheds with historic mercury or PCB occurrences or related management concerns.

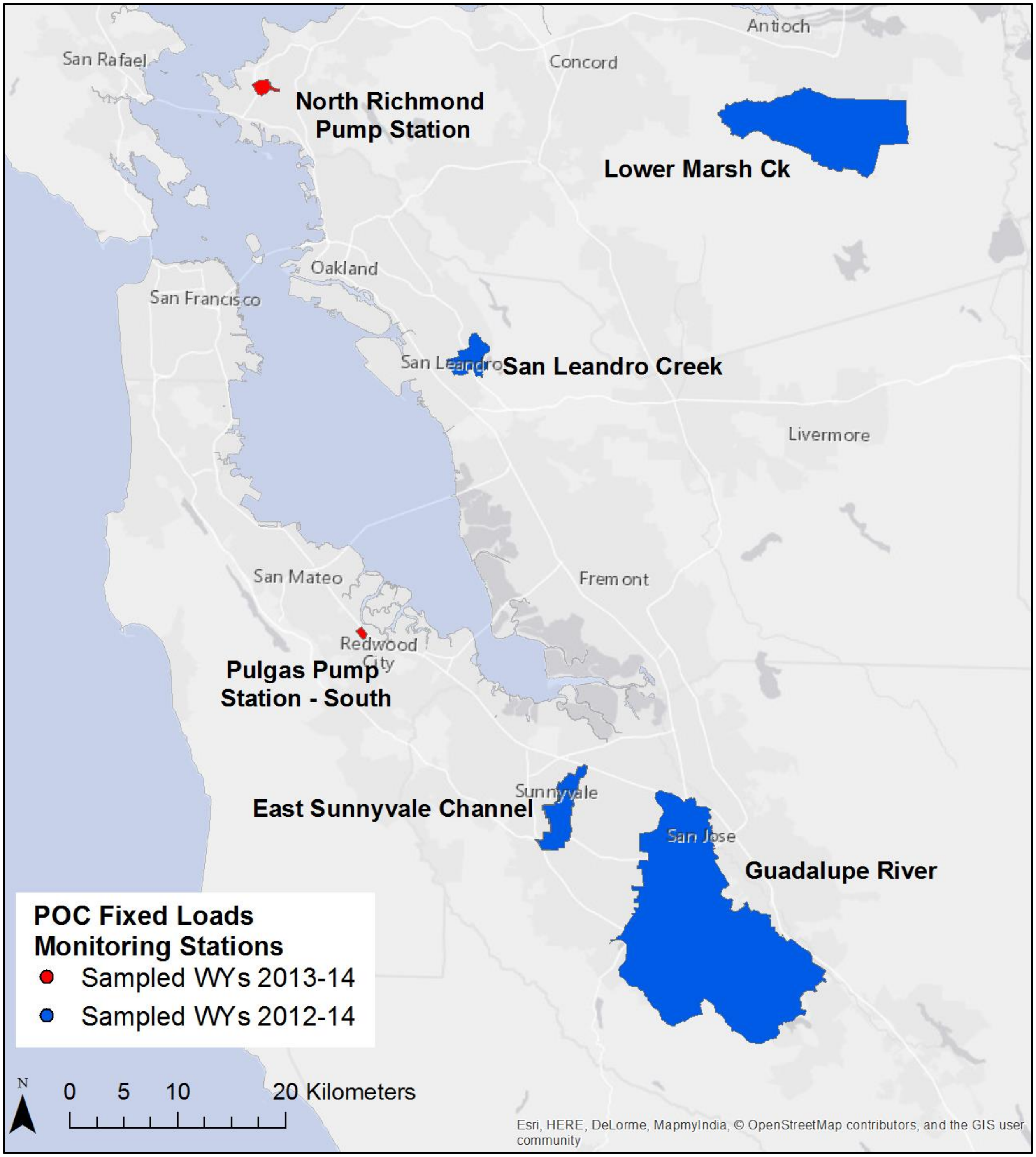

Figure 1. Water year 2012, 2013 and 2014 sampling watersheds.

Table 1. Sampling locations in relation to Countywide stormwater programs and sampling methods at each site.

County program	Watershed name	Water years sampled	Watershed area (km^2)[1]	Sampling location			Operator	Discharge monitoring method	Turbidity	Water sampling for pollutant analysis		
				City	Latitude (WGS1984)	Longitude (WGS1984)				Hg/MeHg collection	Discrete samples excluding Hg species	Composite samples
Contra Costa	Marsh Creek	2012-2014	99	Brentwood	37.990723	-122.16265	ADH	USGS Gauge Number: 11337600[2]; STLS creek stage applied to USGS discharge rating	OBS-500[4]	Manual grab	ISCO auto pump sampler[8]	ISCO auto pump sampler[8]
Contra Costa	North Richmond Pump Station	2013-2014	2.0	Richmond	37.953945	-122.37398	SFEI	Measurement of pump rotations/ interpolation of pump curve	OBS-500[4]	FISP US D95[7]	ISCO auto pump sampler[8]	ISCO auto pump sampler[8]
Alameda	San Leandro Creek	2012-2014	8.9	San Leandro	37.726073	-122.16265	SFEI WY2012 ADH WYs 2013-14	STLS creek stage/ velocity/ discharge rating	OBS-500**[4]	FISP US D95[7] WY 2012 ISCO pump sampler WY 2013-14	ISCO auto pump sampler[8]	ISCO auto pump sampler[8]
Santa Clara	Guadalupe River	2012-2014	236	San Jose	37.373543	-121.69612	SFEI WY2012 Balance WYs 2013-14	USGS Gauge Number: 11169025[3]	DTS-12[5]	FISP US D95[6]	FISP US D95[6]	FISP US D95[6]
Santa Clara	East Sunnyvale Channel	2012-2014	14.8	Sunnyvale	37.394487	-122.01047	SFEI	STLS creek stage applied to SCVWD discharge rating[6]	OBS-500*[4] WY 2012 DTS-12[5] WYs 2013-14	FISP US D95[8]	ISCO auto pump sampler[8]	ISCO auto pump sampler[8]
San Mateo	Pulgas Pump Station - South[9]	2013-2014	0.6	San Carlos	37.504583	-122.24901	KLI	ISCO area velocity flow meter with an ISCO 2150 flow module	DTS-12[5]	Pole sampler	ISCO auto pump sampler[8]	ISCO auto pump sampler[8]

[1]Area downstream from reservoirs
[2]USGS 11337600 MARSH C A BRENTWOOD CA
[3]USGS 11169025 GUADALUPE R ABV HWY 101 A SAN JOSE CA
[4]Campbell Scientific OBS-500 Turbidity Probe
[5]Forest Technology Systems DTS-12 Turbidity Sensor
[6]This rating curve was verified with discharge velocity measurements in WY 2012
[7]FISP US D-95 Depth integrating suspended hand line sampler
[8]Teledyne ISCO 6712 Full Size Portable Sampler
[9]Both the northern and southern catchments to the Pulgas Pump Station were sampled in the WY 2011 characterization study (McKee et al., 2012)
*OBS-500 malfunctioned during WY 2012 due to low flow water depth. A DTS-12 was installed during WY 2013
**OBS-500 malfunctioned during some WY2014 events

The monitoring design focused on winter season storms between October 1 and April 30 of each water year; the period when the majority of pollutant transport occurs in the Bay Area (McKee et al., 2003; McKee et al., 2006; Gilbreath et al, 2012). At all six sampling locations, measurement of continuous stage and turbidity at time intervals of 15 min or less was the basis of the chosen monitoring design (Table 1). At free flowing sites, stage was used along with a collection of discrete velocity measurements to generate a rating curve between stage and instantaneous discharge. Subsequently this rating curve was used to estimate a continuous discharge record over the wet season by either the STLS team or USGS depending on the sampling location (Table 1). At Richmond pump station, an optical proximity sensor (Omron, model E3F2) was used along with stage measurements and a pump efficiency curve based on the pump specifications to estimate flow. ISCO flow meters were deployed at the Pulgas Pump Station (Table 1). In the creek and channel sampling locations, turbidity[2] probes were mounted in the thalweg of each sampling location on an articulated boom that allowed turbidity sampling at approximately mid-depth under most flow conditions (McKee et al., 2004). At North Richmond Pump Station, the turbidity probe was mounted on a boom that extended into the center of the central well. At Pulgas Pump Station South, the turbidity probe was attached to the catch basin wall at a fixed height, which was selected to ensure the probe remained submerged.

Composite and discrete samples were collected for multiple analytes from the water column over the rising, peak, and falling stages of the hydrograph. The sampling design was developed to support the use of turbidity surrogate regression (TSR) during loads computations. This method is deemed one of the most accurate methods for the computation of loads of pollutants transported dominantly in particulate phase such as suspended sediments, mercury, PCBs and other pollutants (Walling and Webb, 1985; Lewis, 1996; Quémerais et al., 1999; Wall et al., 2005; Ruzycki et al., 2011; Gilbreath et al., 2012; Riscassi and Scanlon, 2013). The method involves logging a continuous turbidity record in a short time interval (15 min or less during the study) and collecting a number of discrete samples to support the development of pollutant-specific regressions. In this study, although not always achievable (see discussion later in the report), field crews aimed to collect 16 samples per water year during an early storm, several mid-season storms (ideally including one of the largest storms of the season) and a later season storm. The use of turbidity surrogate regression and the other components of this sampling design was recommended over a range of alternative designs (Melwani et al 2010), and was adopted by the STLS (BASMAA, 2011).

Discrete samples for analytes used for loads computations (except water samples collected for mercury, methylmercury and a simultaneously collected sample for suspended sediment analysis) were collected using the ISCO autosampler as a slave pump at all the sites except the Guadalupe River site. At the Guadalupe River location, all discretely collected samples were collected using a Teflon coated Federal Interagency Sediment Program (FISP) D-95 depth-integrating water quality sampler due to the large

[2] Turbidity is a measure of the “cloudiness” in water caused by suspension of particles, most of which are less than 62.5 µm in size and, for most creeks in the Bay Area, virtually always less than 250 µm (McKee et al., 2003). In natural flowing rivers and urban creeks or storm drains, turbidity usually correlates with the concentrations of suspended sediments and hydrophobic pollutants.

distance between the overhead structure (a road bridge) and the water surface. Discrete samples for analysis of mercury and methylmercury and a simultaneously collected sample for SSC analysis were collected with the D-95 at Guadalupe, East Sunnyvale Channel, North Richmond Pump Station, and San Leandro Creek (WY 2012 only), using a pole sampler at Pulgas Pump Station - South, by manually dipping an opened bottle from the side of the channel at Lower Marsh Creek (all WYs), and by ISCO manual pump at San Leandro (in WY 2013-2014) (Table 1).

Tubing for the ISCO autosamplers was installed using the clean hands technique, as was the 1 L Teflon bottle for use with the D-95. Composite samples made up of a number of discrete sub-samples were collected using the ISCO autosampler at all of the sites except Guadalupe River. Composite samples and the timing of each individual sub-sample were collected with the intent of representing the average concentrations during a storm runoff hydrograph for each storm event sampled. The concentration of a particular analyte of interest obtained from laboratory analysis of such a composite sample is usually referred to as an event mean concentration (Stone et al., 2000; Ma et al., 2009). However, as will be discussed later for each of the individual sites, the composites collected during this study rarely captured sub-samples from the entire hydrograph. Additionally, these composites were time-weighted (except at North Richmond Pump Station where collection times were limited to times of pump outs) rather than flow-weighted, chosen to better represent the average conditions that an organism would be exposed to over a period of time, which was advantageous to the interpretation of toxicity. At the Guadalupe site, a FISP D-95 depth integrating water quality sampler was used to collect multiple discrete samples over the hydrograph which were manually composited on-site.

All water samples were collected in pre-labeled appropriately sized and cleaned sample bottles and placed on ice in coolers either during the sampling procedure or as soon as practically possible. Samples were transported back to the office and labels were rechecked as they were logged in prior to and in preparation for shipment to the laboratories.

2.2. Loads computational methods

It has been recognized since the 1980s that different sampling designs and corresponding loads computation techniques generate computed loads of differing magnitude and of varying accuracy and precision (e.g. Walling and Webb, 1985). Therefore, how can we know which methodology generates the most accurate load? Generally, techniques that maintain high resolution variability in concentration and flow data during the field collection and subsequent computation process result in high-resolution loads estimates. Less accurate loads are generated by sampling designs that do not account for (or adequately describe) the concentration variability (e.g. a daily or weekly sampling protocol would not work for a semi-arid environment like the Bay Area where storm hydrographs are flashy even in larger watersheds) or that use some kind of mathematical average concentration (e.g. simple mean; geometric mean; flow weighted mean) combined with monthly or annual time interval flows (again would not work in the semi-arid environment since 95% of flow occurs during storms). While maintaining respect to the goal that the objective of environmental data interpretation is to neither over nor under interpret the available data, loads computation techniques may often be improved with extra effort to stratify the data. Stratification can be done in relation to environmental processes such as seasonality, flow regime, or data quality. In a general sense, the more resolved the data are in relation to the processes of concentration or flow variation, the more likely it is that computations will result in loads with high accuracy and precision. The data collection protocol implemented through the Small Tributaries Loading Strategy (STLS) was designed to allow for data stratification in the following manners:

1. Early-season (“1st storm”) storm flow sampled for pollutants
2. Mid-season (“largest flood”) storm flow sampled for pollutants
3. Later-season storm flow sampled for pollutants
4. Early-, mid-, and later-season storm flow when no pollutant sampling took place
5. Dry weather flow

Loads computation techniques differ for each of these strata in relation to pollutants that are primarily transported in dissolved or particulate phase. As subsequent samples were collected each year at the STLS monitoring sites, our knowledge about how concentrations varied with season and flow (improvements of the definition of the strata) and thus about how best to apply loads computation techniques gradually improved. Therefore, with each additional annual reporting year, the loads were recomputed. This occurred in relation to both improved flow information as well as an improved understanding of concentration variation in relation to seasonal characteristics and flow. The loads and interpretations presented here therefore supersede those reported in previous annual reports for WY 2012 (McKee et al., 2013), and 2013 (Gilbreath et al., 2014).

During the study, concentrations either measured or estimated were multiplied with the continuous estimates of flow (1-15 minute interval) to compute the load on a 1 to 15 minute basis and summed to monthly and wet season loads. Laboratory measured data were retained in the calculations and assumed real for that moment in time. The techniques for estimating concentrations were applied in the following order of preference (and resulting accuracy of loads) as appropriate for each analyte (see summary in Table 2):

Linear interpolation: Linear interpolation was the primary technique used for interpolating concentrations between measured data points when storms were well sampled. It is the most accurate loads computation method for such storms and retains the maximum amount of information about how concentration and flow varies during the storm of interest (Young and DePinto, 1988; Kronvang and Bruhn, 1996). Two linear interpolation approaches were applied:

> **Linear Interpolation using water concentrations (LI_{WC}):** Linear interpolation using water concentrations is the process by which the interpreter estimates the concentrations mathematically between observed measurements using a linear time step (Kronvang and Bruhn 1996). It was appropriately used for pollutants which occur mainly in dissolved phase because it does not incorporate varying turbidity or SSC (Table 2). It can be used for analytes that are primarily transported in particulate phase; although during this study a superior method using particle ratios was applied to those analytes (Table 2).

Linear Interpolation using particle ratios (LI_{PR}): Linear interpolation using particle ratios can be thought of as locally derived regression in three-dimensional space. It is superior to linear interpolation using water concentrations (see above) for pollutants which occur mainly in particulate form because it ensures that the relationship between the derived concentration and varying turbidity that occurs between the two laboratory pollutant measurements results in particle ratios that, at all times, are reasonable (simpler linear interpolation of concentrations between samples may lead to unreasonable particle ratios for example if samples are collected on either side of a turbidity peak leading to lower particle ratios estimated at the turbidity peak). The use of this method was decided upon in concert with the field sampling design and was only possible because of the collection of continuous turbidity measurements. It was ideal for PCBs and Hg (two of the analytes of most interest) as well as other particulate phase analytes like total phosphorus (Table 2).

Regression Estimators: Regression estimator methods for loads calculations involve developing relationships between limited sample concentration data and an unlimited surrogate measure (e.g. turbidity or flow). These relationships are then applied to the unlimited surrogate measure record (e.g. the short time interval records of flow or turbidity) to calculate short time interval estimates of pollutant concentrations. This loads calculation method has been widely applied to estimating suspended sediment loads throughout the world (e.g., Walling and Webb, 1985; Lewis, 1996), demonstrated by SFEI and others to work well for metals (e.g., Quémerais et al., 1999; Wall et al., 2005; David et al. in press; McKee et al., 2010; Ruzycki et al., 2011; Riscassi and Scanlon, 2013), and more recently been demonstrated by SFEI to work well for organic pollutants (McKee et al., 2006; Gilbreath et al., 2012). This study was designed specifically to apply this method for loads calculations of discretely sampled analytes.

Interpolation using unique POC-flow based regression equations (FSR): The flow based surrogate regression interpolation method was applied for pollutants transported dominantly in the dissolved phase and forming a good relationship with flow, or to the more particle associated pollutants during periods when a turbidity probe failed to deliver quality data (yet the relationship with flow was preferred over resorting to a simple ratio or averaging method.

Interpolation using unique POC-turbidity based regression equations (TSR): Turbidity surrogate regression can be considered the default standard for pollutants of concern that are primarily transported in a particulate form. These types of pollutants (for example PCBs and mercury) form strong linear relationships with either turbidity or SSC. For the particle associated pollutants, turbidity surrogate regression was applied to all unsampled flood flow conditions observed at each monitoring site except under rare circumstances when turbidity data were not available due to probe malfunction. This interpolation method is superior to FSR for particle-associated pollutants because it takes into account hysteresis in relation to flow (Walling and Webb, 1985; Lewis, 1996). For example concentrations of suspended sediment and pollutants that are strongly associated with suspended sediment often have greater concentrations during the rising stage of the hydrograph for a given flow as compared with concentrations at the same flow magnitude but on the falling stage of a given hydrograph. This occurs because there is more energy in the water column and typically no transport or source limitations during the rising stages of the hydrograph and earlier phases of a storm. Conversely, water transported during the falling stages is typically less turbulent and sources may have been washed clean by this time or, for the larger watersheds or those that have nonurban land-use in the upstream areas, lower concentrations can occur purely because the origin of the water has evolved to include upstream or less impervious components of the watershed. Note: TSR was used to estimate SSC during non-sampled flood flow periods, but the specific methods applied to developing the turbidity-SSC regression relationships were based on the USGS methods for this analysis following Rasmussen et al., 2009. The method involves developing a simple linear regression model for SSC and turbidity pairs on untransformed or log-log transformed data, and in cases where a simple linear model does not meet a minimum standard (model standard percentage error (MSPE) < 20%), a multiple linear model including both turbidity and flow is evaluated.

Ratios and Averages: During unsampled periods of the record and in cases where pollutants did not form strong relationships with surrogate measures (turbidity, flow and other measured pollutants were all explored), or during periods when the surrogate measure record was unavailable, a simple ratio or average estimator method was applied.

Flow Weighted Mean Concentration (FWMC): In the event that flow or turbidity/SSC does not adequately explain the variation in pollutant concentrations, a flow weighted mean concentration can be calculated and applied to the appropriate flow classes. This is a simple ratio method that averages the concentration data but weighted more heavily towards the greatest flow and thus is an improvement over a simple average (Walling and Web, 1985, Birgand et al., 2010). If warranted, the data may be stratified first with a different FWMC applied to each stratum. Stratification in this manner has been previously applied for Chesapeake Bay tributaries and found to improve the accuracy of loading estimates (Lawson et al., 2001). Using a FWMC is the lowest accuracy method applied in this study for estimating storm flow concentrations.

Interpolation assuming a representative concentration (e.g. "dry weather lab measured" or "lowest measured"): To apply this method, an estimate of average concentrations under certain flow conditions is combined with discharge. This is, in effect, a simple average estimator and is the least accurate and precise of all the loads calculation methods. Because this sampling program focuses on characterizing concentration during storm flows, it may be desirable to use this method in addition to one or more of the previously mentioned methods (e.g. this method may better characterize lower flows alongside use of the FWMC to better characterize storm flows).

3. Continuous data quality assurance

3.1. Continuous data quality assurance methods

Prior to the start of WY 2012, the STLS monitoring teams developed the continuous monitoring protocols for the study collaboratively. Basic quality assurance methods were applied to the WY 2012 dataset. In WY 2013, a better documented method for quality assurance was developed and applied to continuous data (turbidity, stage, and rainfall) collected at the POC loads monitoring stations (McKee et al., 2013). QA was performed on WY 2012 data though not as systematically as later years. Quality of the continuous data record for each monitoring location for all three years are highlighted in the text below and summarized in Tables 3 and 4.

Table 2. Methods predominantly used for loads computations in relation to each pollutant of concern.

Computation method[a]	SS	TOC	PCBs	HgT	MeHgT	NO3	PO4	TP
Linear interpolation water concentrations (LI_{WC})		✓				✓	✓	
Linear interpolation particle ratios (LI_{PR})	✓		✓	✓	✓			✓
Turbidity surrogate regression (TSR)	✓	✓	✓	✓	✓			✓
Flow surrogate regression (FSR)		✓				✓	✓	
Flow weighted mean concentration (FWMC)		✓				✓	✓	
Assumed representative concentration (for dry weather flow)	✓	✓	✓	✓	✓	✓	✓	✓

[a] Exceptions to the methods listed for each analyte include: FWMCs were used for all analytes at Pulgas Pump Station - South. Flow Surrogate Regression was used for most analytes at San Leandro Creek when the turbidity sensor was malfunctioning or had been removed to protect it from vandalism (FWMCs had to be used during these periods for TOC, NO3 and PO4), and at East Sunnyvale Channel to estimate SSC during all of WY 2012 and portions of WY 2013 when the turbidity record was impacted by vegetation collecting at the sensor. The estimated SSC was then used in regressions with particulate associated pollutants.

Throughout the season, field staff were responsible for data verification checks after data were downloaded during site visits. The field staff reviewed the data and completed a data transmission record. During the data validation process, individual records were flagged if they didn't meet the criteria developed in the continuous QA protocol. Datasets were evaluated in relation to the validation criteria, including: accuracy of the instruments through calibration, accuracy of the instruments in relation to comparison with manual measurements, dataset representativeness relative to logging interval and the degree of change from one measurement to the next, completeness of the dataset relative to the target monitoring period (October 1 – April 30) and finally our confidence in the corrections applied to the data records (Table 3 and Table 4). For more information on the quality assurance procedures developed and applied for continuous data, the reader is referred to the current version of the draft "*Quality Assurance Methods for Continuous Rainfall, Run-off, and Turbidity Data*" (McKee et al., 2015).

3.2. Continuous data quality assurance summary

The targeted monitoring period for this study was October 1 through April 30 each wet season (totaling 212 days each season). Especially in the first year of monitoring at each location in which the STLS team installed equipment (this excepts all equipment at Guadalupe as well as stage/flow equipment at Lower Marsh for WYs 2012 and 2013), there were often delays to start the season. The delay to start was the sole reason for missing stage data at all sites except for North Richmond Pump Station in WY 2013 when there was a 7 day period of missing record in October 2012 for unknown reasons. In addition to delayed starts, occasionally the rain gauges clogged, leading to data gaps in the rainfall records, and the expensive turbidity sensor at San Leandro Creek was often removed during periods when no rain was expected in order to prevent vandalism. A complete review of the number of days missing (out of 212) for each continuous record is provided in Table 3.

Overall the continuous rainfall data were acceptable. Rain data were collected at all the sites except for Guadalupe (Note, SCVWD collects high quality rainfall data throughout the Guadalupe River watershed), and the data were collected on the same time interval as stage and turbidity (except at North Richmond and Pulgas pump stations where rainfall data were collected on the 5 minute interval but stage and turbidity intervals were variable). Rain gauges were cleaned before and periodically during the season, but not calibrated. All sites except for the North Richmond Pump Station and Lower Marsh Creek

Table 3. Continuous data quality assurance summary for record completeness and accuracy for each monitoring location. Missing days for all three monitoring years are provided, but quality ratings for accuracy of comparison were only developed for WYs 2013 and 2014. When only one rating is provided, it is relevant for both WYs. "NR" indicates that the QA procedure was not completed and "NA" indicates that the QA procedure was not applicable.

	Missing Days in Period of Record[a]			Accuracy of Comparison[b]		
	Rainfall	Stage	Turbidity	Rainfall	Stage	Turbidity
Lower Marsh	58/ 31/ 61	0 / 0 / 36	58 / 31 / 36	Excellent	NR	NR
Richmond	NA/ 61 / 0	NA / 7 / 17	NA / 0 / 0	Poor[1] / Excellent	NR / Excellent	Good[2] /Excellent
San Leandro	38 / 48 / 30	38 / 42 /23	38 / 42 / 37[3]	Excellent	Excellent	Excellent
Guadalupe	Complete	Complete	5 / 5 / 21	NA	NR / Excellent	Excellent
Sunnyvale	61 / 0 / 1	61 / 0 / 0	61 / 0 / 0	Excellent	Excellent	Excellent
Pulgas	NA / 9 / 21	NA / 41 / 21	NA / 117 / 72	Excellent	NR	Poor[4]

[a] Number of missing days is out of total target of 212 days. Number of missing days is provided for each monitoring year (WY 2012 - 2014)

[b] Accuracy of comparison is provided for WYs 2013 and 2014, the years for which this metric was evaluated systematically.

[1] Rainfall tipping bucket clogged during portions of December and January, leading to a poor relationship between the site record and other nearby rain gauge records.

[2] Regression between sensor and manual measurement data $R^2 = 0.85$.

[3] In total, 158 days of this record were missing turbidity in WY 2014. However, much of that time stages were low enough that no flow occurred. The 37 days noted includes the 23 days at the beginning of the record in which stage was not recorded plus 14 days in which flow did occur yet turbidity was not recorded. This equates to approximately half of the storms in WY 2014 which have no turbidity data.

[4] Manual turbidity measurements against sensor measurements had an $R^2 = 0.25$ in WY 2013 and 0.09 in WY 2014; this record fluctuated dramatically and cyclically (presumably in relation to pump outs); additional review of these data is recommended by BASMAA as they believe application of additional smoothing techniques may improve correlation between manual and sensor turbidity readings.

Table 4. Continuous data quality assurance summary for representativeness and confidence in corrections for each monitoring location. Quality ratings were only developed for WYs 2013 and 2014. When only one rating is provided, it is relevant for both WYs. "NA" indicates that the QA procedure was not applicable.

	Representativeness of the Population[c]			Confidence in Corrections[c]		
	Rainfall	Stage	Turbidity	Rainfall	Stage	Turbidity
Lower Marsh	Excellent	Excellent	Excellent	Excellent / Poor[1]	Excellent	Excellent
Richmond	Excellent	Excellent	Poor / Good[2]	Good[3] / Excellent	Excellent	Excellent
San Leandro	Excellent	Excellent	Excellent	Excellent	Excellent	Poor[4]
Guadalupe	NA	Excellent	Excellent	NA	USGS maintained	Excellent
Sunnyvale	Excellent	Good[5] / Excellent	Excellent	Excellent	Excellent	Poor/Good[6]
Pulgas	Excellent	Excellent / Poor[7]	Good[8]	Excellent	Poor/Good[9]	Poor[10]

[c] Representativeness of the Population and Confidence in Corrections metrics are provided for WYs 2013 and 2014, the years for which this metric was evaluated systematically

[1] During WY 2014, data from 59% of the actual rain days were rejected due to clogging of the tipping bucket. The data were substituted with records from nearby local stations (Weather Underground). The regression of daily total rainfall between one of these substituted gages and the site gage for days when the tipping bucket was working had a coefficient of variation of 0.61; the other site has since been decommissioned and the relationship could not be evaluated.

[2] In WY 2013, 4.2% of the population (251 records) had > 20 NTU absolute value change and ≥15% relative change from the preceding record; 2.9% (171 records) had > 20 NTU absolute value change and >50% relative change from the preceding record. In WY 2014, 3.7% of the population had >20 NTU absolute value change and ≥15% relative change from the preceding record; 2.1% had >20 NTU absolute value change and >50% relative change from the preceding record.

[3] Data missing due to clogging was corrected with the nearby Richmond City Hall gage; the regression of daily total rainfall between the Richmond City Hall gage and the site gage for days when the tipping bucket was working had an $R^2 = 0.91$.

[4] Turbidity could not be measured at flows <0.4 ft. Generally, however, these were likely periods of very low turbidity anyway. However, during WY 2013, 23% of records for stages > 1ft were missing turbidity, and in WY 2014, several entire storms were missed due to the sensor not being installed to prevent vandalism or malfunctioning. For WY 2014, 43% of record for which there was flow did not have corresponding turbidity records.

[5] 4.7% of records at Sunnyvale in WY 2013 showed a >15% change between consecutive readings.

[6] The sensor installed during WY 2012 was not adequate for measuring turbidity at lower flows and the entire record was rejected (noted here but not reflected in the Table 4 rating since only WYs 2013 and 2014 are rated. During the subsequent water years, vegetation frequently got caught on the boom structure within the channel and fouled the turbidity record. During WY 2013, 8.3% of the record was rejected and could not be corrected. In WY 2014, 7% of records required correction but this time there was relatively clear evidence for the method used to fill data gaps.

[7] 14% of the records at Pulgas showed a >15% change between consecutive readings, 7% were >25% change, and 1.3% were >100%change.

[8] In WY 2013, 1.9% of the population (483 records) had > 20 NTU absolute value change and ≥15% relative change from the preceding record; 1.3% (328 records) had > 20 NTU absolute value change and >50% relative change from the preceding record. Recommended action for improvement is to shorten the recording interval from 5 minutes to 1 minute. In WY 2014, 1.6% of the population had > 20 NTU absolute value change and ≥15% relative change from the preceding record; 1.0% had > 20 NTU absolute value change and >50% relative change from the preceding record.

[9] During WY 2013, a large portion of the record was on intervals > 15 minutes and we often had no confidence in a method to correct the data. Equipment issues were improved in WY 2014 and the recording interval was set to 15 min except during times of flow, when it switched to logging on the 1 min interval. However, back-ups into the stormdrain led to zero-flow conditions prompting the measurement interval back to 15 minute intervals. It is unknown what the flow was between these occurrences. In total, this scenario appeared to have happened between 15-25 times and back-of-the-envelope calculations suggest that 2-4 % of the total flow volume was likely not recorded as a result.

[10] The turbidity sensor was placed in a catchbasin near the pump station and the runoff in the catchbasin was vigorously pumped out when the pump station turned on. This led to cyclical large variations in the turbidity record, and BASMAA is currently investigating the pump station on/off times to determine if spikes due to pumping can be identified and discerned from erroneous spikes. Pending additional review, the current comparison to the manual turbidity measurements was poor, and we have little confidence in the corrections that were applied to the dataset. Furthermore, the recording interval for WY 2013 was set to 5 min. This was also the case for WY 2014, except during times of flow, when it logged on the 1 min interval consistent with the stage record. However, back-ups into the stormdrain led to zero-flow conditions prompting the measurement interval back to 5 minutes. It is unknown what the flow and turbidity was between these occurrences.

compared well to nearby rain gauges. Clogging of the tipping buckets at these two sites led to discrepancies in the record compared with nearby gauges. The daily data of the site gage was regressed with the daily data of a nearby gage during periods when the site gage was working, and the regression was used to correct the site gage record. The regression was strong for North Richmond (R^2 = 0.91) but poor for San Leandro Creek (R^2 = 0.61). All sites had rainfall totals during 5-, 10- and 60-minute intervals that aligned with 1-, 2- and 5-year rainfall returns in their respective regions.

Overall the continuous stage data were acceptable. When collected, manual stage measurements compared well with the corresponding record from the pressure transducer ($R^2 > 0.99$ at all sites all years where it was measured). Percent differences between consecutive records were reasonable at all sites with the exceptions of Sunnyvale in WY 2013 and Pulgas in WY 2014 when there were nearly 5 and 14% of the records at each station in which consecutive records showed greater than a 15% difference in stage measurement. Manual stage measurements were not collected at Pulgas Pump Station - South at all during the study, and could not be used to verify the accuracy or precision of those stage records.

At the creek and channel sites, flow was calculated from the continuous stage record and therefore the accuracy of the estimated flows was dependent on a quality stage record as well as a quality discharge rating curve. At Lower Marsh Creek and Guadalupe River, the USGS had already developed discharge rating curves. The Santa Clara Valley Water District (SCVWD) provided a discharge rating curve for Sunnyvale Channel, and through measurements over a broad stage range, the STLS team verified the quality of the SCVWD curve. The San Leandro location was a challenging cross section to rate given no bed control, seasonally variable vegetation on the banks, variation in the cross-section morphology within and just upstream of the measurement point under the bridge and a near-field side channel entry just upstream. Given these issues, a flow rating for the site would likely take many years under a very wide variety of storms to verify with certainty. With these challenges in mind, the STLS team began development of a discharge rating curve at San Leandro Creek, which was well-measured in WY 2012 and 2014 at stages <2 feet and with three measurements in WY 2012 at approximately 3.5 feet of stage. Due to the large gap in measurements between 2 and 3.5 feet of stage, as well as no measurements for flows between 3.5 and 4 feet of stage (the maximum stage recorded during that study on 12/23/2012), we could have at best moderate confidence in the flow estimates for this site. Compounding this uncertainty, flow volumes estimated during storms of similar sizes between monitoring years were substantially different from year to year perhaps associated with morphological changes that were not documented. Therefore, despite excellent QA ratings for the continuous stage record at San Leandro, our overall confidence in the flow record for this site is low.

The pump station sampling locations employed alternate methods of flow estimation and therefore additional QA procedures were applied to the flow records. The stage records were evaluated for these sites in the same manner as for the creek and channel locations. Additionally, at North Richmond Pump Station, the optical proximity sensor record was reviewed for consistency of the pump shaft rates during times of operation. At both North Richmond and Pulgas pump stations, the storms during each water year monitored were isolated and total flow and precipitation volumes were calculated. Relationships between these metrics were evaluated and used to identify eight storms at Pulgas from WY 2013 when the flow meter was malfunctioning. After censorship of these storms, the rainfall-runoff relation at each site was excellent (r^2=0.96 and r^2=0.98 for North Richmond and Pulgas, respectively).

Continuous turbidity data were rated excellent at Lower Marsh Creek and Guadalupe River throughout the monitoring periods. The San Leandro Creek dataset was relatively free from spikes requiring censorship or correction but had a large portion of missing records due to failure to install the sensor prior to some storm events and delays in correcting sensor loss or malfunction. East Sunnyvale Channel's entire WY 2012 record was censored because the numerous spikes that resulted from the OBS-500 reading the bottom of the channel during low flows could not be corrected. The turbidity record for East Sunnyvale Channel also had numerous spikes in the subsequent two years of monitoring due to vegetation catching on the boom structure and interfering with the turbidity measurement; this record could not be corrected for small portions of WY 2013 but because more frequent maintenance was implemented in WY 2014 to address this problem, the entire record could be used after correction of some records. The two pump station monitoring sites were the most dynamic in terms of turbidity magnitude changes from record to record and presented the most challenging logistics for turbidity measurement, which resulted in diminished quality. At North Richmond Pump Station, for example, the regression between sensor and manual measurements in WY 2013 was slightly less than ideal (r^2 = 0.85) and despite the frequent 1-minute logging interval, 4.2% of the WY 2013 records during pump outs had relative changes in turbidity magnitudes from record to record greater than 15% and 20 NTU, leading to a quality ranking of "Poor" for WY 2013. Field staff noted throughout the season large amounts of trash in the pump station well where monitoring occurred, and this could be the cause of the turbidity fluctuations, though it is also conceivable that the small urban system and unique monitoring configuration could have been so dynamic as to result in these relative changes. At Pulgas Pump Station - South the turbidity sensor was placed in a catchbasin near the inlet to the pump station and the runoff in the catchbasin was vigorously pumped out when the pumps turned on. This led to cyclical large variations in the turbidity record, and it was not always possible to discern erroneous spikes in the data record as opposed to the cyclical spikes resulting from the pump outs. BASMAA is undertaking further review of the pump on/off times to determine if spikes due to pumping can be identified and if somehow this information will be useful to estimating loads. Furthermore, the recording interval for WY 2013 was set to 5 min, which was long in duration relative to the dynamically changing system. The logging interval was improved in WY 2014, such that during times of flow turbidity was recorded on the 1 min interval consistent with the stage record. However, the programming logic set to accomplish this changing interval created some periods in which flow and turbidity were likely not recorded on the shorter intervals. The current comparison to the manual turbidity measurements at Pulgas Creek was poor in both water years, and we have little confidence in the corrections that were applied to the dataset. Ultimately, the turbidity record was not used to estimate continuous loads at Pulgas Creek, and a flow-based or flow-weighted mean concentration approach was adopted instead. BASMAA has suggested they may undertake further review of this dataset, including application of smoothing functions to better fit the pollutant data to the turbidity record and potentially improve the usability of these data.

4. Laboratory analysis and quality assurance

4.1. Sample preservation and laboratory analysis methods

All samples were labeled, placed on ice, transferred back to the respective site operator's headquarters, and refrigerated at 4 °C until transport to the laboratory for analysis. Laboratory methods were chosen to ensure the highest practical ratio between method detection limits, accuracy and precision, and costs (BASMAA, 2011; 2012). No changes were made between WYs 2013 and 2014 in laboratories conducting the chemical analyses (Table 5).

An inter-comparison study, started in WY 2013 and continued in WY 2014, was designed to assess any impacts of laboratory change during the study. A subset of samples were collected in replicate in the field and sent to the previous and replacement laboratories for analysis. Nutrients, copper, mercury, methylmercury, selenium and pyrethroid samples were analyzed as part of the inter-comparison study. Individual laboratory QA summaries for the WY 2014 inter-comparison analyses are presented in section 5.2 of this report. A review of the inter-comparison study results and laboratory QA can be found in Attachment 2.

Table 5. Laboratory analysis methods for WY 2014 samples.

Water Year	Analyte	Method	Field Filtration	Field Acidification	Laboratory
WY2012	Carbaryl	EPA 632M	No	No	DFG WPCL
WY2013	Carbaryl	EPA 632M	No	No	DFG WPCL
WY2014	Carbaryl	EPA 632M	No	No	DFG WPCL
WY2012	Copper[1]	EPA 1638M	No	No	Brooks Rand Labs LLC
WY2013	Copper[1]	EPA 1638M	No	No	Caltest Analytical Laboratory
WY2014	Copper[1]	EPA 1638M	No	No	Caltest Analytical Laboratory
WY2012	Dissolved OrthoPhosphate	EPA 300.1	Yes	No	EBMUD
WY2013	Dissolved OrthoPhosphate	SM20 4500-P E	Yes	No	Caltest Analytical Laboratory
WY2014	Dissolved OrthoPhosphate	SM20 4500-P E	Yes	No	Caltest Analytical Laboratory
WY2012	Fipronil[2]	EPA 619M	No	No	DFG WPCL
WY2013	Fipronil[2]	EPA 619M	No	No	DFG WPCL
WY2014	Fipronil[2]	EPA 619M	No	No	DFG WPCL
WY2012	Nitrate	EPA 300.1	Yes	No	EBMUD
WY2013	Nitrate	EPA 353.2/SM20 4500-NO3 F	Yes	Yes	Caltest Analytical Laboratory
WY2014	Nitrate	EPA 353.2/SM20 4500-NO3 F	Yes	No	Caltest Analytical Laboratory
WY2012	PAHs	AXYS MLA-021 Rev 10	No	No	AXYS Analytical Services Ltd.
WY2013	PAHs	AXYS MLA-021 Rev 10	No	No	AXYS Analytical Services Ltd.
WY2014	PAHs	AXYS MLA-021 Rev 10	No	No	AXYS Analytical Services Ltd.
WY2012	PBDEs	AXYS MLA-033 Rev 06	No	No	AXYS Analytical Services Ltd.
WY2013	PBDEs	AXYS MLA-033 Rev 06	No	No	AXYS Analytical Services Ltd.
WY2014	PBDEs	AXYS MLA-033 Rev 06	No	No	AXYS Analytical Services Ltd.
WY2012	PCBs	AXYS MLA-010 Rev 11	No	No	AXYS Analytical Services Ltd.
WY2013	PCBs	AXYS MLA-010 Rev 11	No	No	AXYS Analytical Services Ltd.

Water Year	Analyte	Method	Field Filtration	Field Acidification	Laboratory
WY2014	PCBs	AXYS MLA-010 Rev 11	No	No	AXYS Analytical Services Ltd.
WY2012	Pyrethroids	AXYS MLA-046 Rev 04	No	No	AXYS Analytical Services Ltd.
WY2013	Pyrethroids	EPA 8270Mod (NCI-SIM)	No	No	Caltest Analytical Laboratory
WY2014	Pyrethroids	EPA 8270Mod (NCI-SIM)	No	No	Caltest Analytical Laboratory
WY2012	Selenium[1]	EPA 1638M	No	No	Brooks Rand Labs LLC
WY2013	Selenium[1]	EPA 1638M	No	No	Caltest Analytical Laboratory
WY2014	Selenium[1]	EPA 1638M	No	No	Caltest Analytical Laboratory
WY2012	Suspended Sediment Concentration	ASTM D3977	No	No	EBMUD
WY2013	Suspended Sediment Concentration	ASTM D3977-97B	No	No	Caltest Analytical Laboratory
WY2014	Suspended Sediment Concentration	ASTM D3977-97B	No	No	Caltest Analytical Laboratory
WY2012	Total Hardness	EPA 1638M	No	Yes	Brooks Rand Labs LLC
WY2013	Total Hardness	SM 2340	No	Yes	Caltest Analytical Laboratory
WY2014	Total Hardness	SM 2340 C	No	Yes	Caltest Analytical Laboratory
WY2012	Total Mercury	EPA 1631EM	No	Yes	Moss Landing Marine Laboratories
WY2013	Total Mercury	EPA 1631EM Rev 11	No	Yes	Caltest Analytical Laboratory
WY2014	Total Mercury	EPA 1631EM Rev 11	No	Yes	Caltest Analytical Laboratory
WY2012	Total Methylmercury	EPA 1630M	No	Yes	Moss Landing Marine Laboratories
WY2013	Total Methylmercury	EPA 1630M Rev 8	No	Yes	Caltest Analytical Laboratory
WY2014	Total Methylmercury	EPA 1630M Rev 8	No	Yes	Caltest Analytical Laboratory
WY2012	Total Organic Carbon	SM 5310 C	No	Yes	Delta Environmental Lab LLC
WY2013	Total Organic Carbon	SM20 5310B	No	Yes	Caltest Analytical Laboratory
WY2014	Total Organic Carbon	SM20 5310B	No	Yes	Caltest Analytical Laboratory
WY2012	Total Phosphorus	EBMUD 488 Phosphorus	No	Yes	EBMUD
WY2013	Total Phosphorus	SM20 4500-P E	No	Yes	Caltest Analytical Laboratory
WY2014	Total Phosphorus	SM20 4500-P E/SM 4500-P F	No	Yes	Caltest Analytical Laboratory
WY2012	Toxicity[3]	See Table note 3 below	No	No	Pacific Eco-Risk Labs
WY2013	Toxicity[3]	See Table note 3 below	No	No	Pacific Eco-Risk Labs
WY2014	Toxicity[3]	See Table note 3 below	No	No	Pacific Eco-Risk Labs

[1] Dissolved selenium and dissolved copper samples were field filtered and field acidified (HNO_3) at the Lower Marsh Creek (WY 2012, 2013, 2014) and San Leandro Creek stations (WY 2013, 2014).
[2] The DFG laboratory filtered these Fipronil samples. This procedure likely biases the results low as compared with fipronil analyses in which the samples were not filtered. At this time, we have no estimate for how low this bias may be. Also note, because the scope of this work was defined prior to more recent scientific information about the importance of fipronil degradates, it did not include the measurement of degradates, some of which we now know may be equally or even more toxic than the parent chemical.
[3] Toxicity testing includes: chronic algal growth test with *Selenastrum capricornutum* (EPA 821/R-02-013), chronic survival & reproduction test with *Ceriodaphnia dubia* (EPA 821/R-02-013), chronic survival and growth test with fathead minnows, *Pimephales promelas* (EPA 821/R-02-013), and 10-day survival test with *Hyalella azteca* (EPA 600/R-99-064M).

4.2. Quality assurance methods for pollutants of concern concentration data

The data quality was reviewed using protocols applied to samples collected for the SF Bay Regional Monitoring Program for Water Quality. Data handling procedures and acceptance criteria may differ among programs. However, underlying data are never discarded; results even for “censored” data are maintained, so impacts of applying different protocols can be assessed if desired.

4.2.1. Holding Times

Holding times are the length of time a sample can be stored after collection and prior to analysis without significantly affecting the analytical results. Holding times vary with the analyte, sample matrix, and analytical methodology used to quantify concentration. Holding times can be extended if preservation techniques are employed to reduce biodegradation, volatilization, oxidation, sorption, precipitation, and other physical and chemical processes.

4.2.2 Sensitivity

The sensitivity review evaluated the percentage of field samples that were non-detects (NDs) as a way to evaluate if the analytical methods employed were sensitive enough to detect expected environmental concentrations of the targeted parameters. In general, if more than 50% of the samples were ND, then the method may not be sensitive enough to detect ambient concentrations. However, review of historical data from the same project/matrix/region (or a similar one) helped to put this evaluation into perspective; in most cases the lab was already using a method that is as sensitive as is possible.

4.2.3 Blank Contamination

Blank contamination was assessed to quantify the amount of targeted analyte in a sample from external contamination in the lab or field. This metric was performed on a lab-batch basis. Lab blanks within a batch were averaged. When the average blank concentration was greater than the method detection limit (MDL), the field samples within each batch were qualified as blank contaminated. If the field sample result (including any reported as ND) was less than 3 times the average blank concentration those results were “censored” and not reported or used for any data analyses. All censored data are made available but are qualified as exceeding QAQC thresholds.

4.2.4 Precision

Rather than evaluation by lab batch, precision was reviewed on a project or dataset level (e.g., a year or season’s data) so that the review took into account variation across batches. Only results that were greater than 3 times the MDL were evaluated, as results near MDL were expected to be highly variable. The overarching goal was to review precision using sample results that were most similar in characteristics and concentrations to field sample results. Therefore the priority of sample types used in this review was as follows: lab-replicates from field samples or field replicates (but only if the field replicates are fairly homogeneous which is unlikely for wet-season runoff event samples unless collected simultaneously from a location). Replicates from Certified Reference Materials (CRMs), matrix spikes, or spiked blank samples were reviewed next with preference to select the samples that most resembled the targeted ambient samples in matrix characteristics and concentrations. Results outside of the project management quality objective (MQO) but less than 2 times the MQO (e.g., ≤50% if the MQO is ≤25% relative percent difference (RPD) or relative standard deviation (RSD)) were qualified; those outside of 2 times the MQO were censored. All censored data are made available but are qualified as exceeding QAQC thresholds.

4.2.5. Accuracy

Accuracy was also reviewed on a project or dataset level (rather than a batch basis) so that the review takes into account variation across batches. Only results that were greater than 3 times the MDL were evaluated. Again, the preference was for samples most similar in characteristics and concentrations to field samples. Thus the priority of sample types used in this review was as follows: CRMs, then Matrix Spikes (MS), then Blank Spikes. If CRMs and MS were both reported in the same concentration range, CRMs were preferred because of external validation/certification of expected concentrations, as well as better integration into the sample matrix (MS samples were often spiked just before extraction). If both MS and blank spike samples were reported for an analyte, the MS was preferred due to its more similar and complex matrix. Blank spikes were used only when preferred recovery sample types were not available (e.g., no CRMs, and insufficient or unsplittable material for creating an MS). Results outside the MQO were qualified, and those outside 2 times the MQO (e.g., >50% deviation from the target concentration, when the MQO is ≤25% deviation) were censored for poor recovery. All censored data are made available in all public data displays but are qualified as exceeding QAQC thresholds.

4.2.6. Comparison of dissolved and total phases

This review was only conducted on water samples that reported dissolved and total fractions. In most cases the dissolved fraction was less than the particulate or total fraction. Some allowance is granted for variation in individual measurements, e.g. with a precision MQO of RPD or RSD<25%, a dissolved sample result might easily be higher than a total result by that amount.

4.2.7. Average and range of field sample versus previous years

Comparing the average range of the field sample results to comparable data from previous years (either from the same program or other projects) provided confidence that the reported data do not contain egregious errors in calculation or reporting (errors in correction factors and/or reporting units). Comparing the average, standard deviation, minimum and maximum concentrations from the past several years of data aided in exploring data, for example if a higher average was driven largely by a single higher maximum concentration.

4.2.8. Fingerprinting summary

The fingerprinting review evaluated the ratios or relative concentrations of analytes within an analysis. For this review, we looked at the reported compounds to find out if there are unusual ratios for individual samples compared to expected patterns from historic datasets or within the given dataset.

Since analyses of organic contaminants at trace levels are often susceptible to biases that may not be detected by conventional QA measures, additional QA review helps ensure the integrity of the reported data. Based on knowledge of the chemical characteristics and typical relative concentrations of organic contaminants in environmental samples, concentrations of the target contaminants are compared to results for related compounds to identify potentially erroneous data. Compounds that are more abundant in the original technical mixtures and are more stable and recalcitrant in the environment are expected to exist in higher concentrations than the less abundant or less stable isomers. For example, PCB congener concentrations follow general patterns of distribution based on the original concentrations in Aroclor mixtures. If an individual congener occurs at concentrations much higher than usual relative to more abundant congeners, the result warrants further investigation.

Furthermore, several contaminants chemically transform into other toxic compounds and are usually measured within predicted ranges of concentrations compared to their metabolites (e.g. heptachlor epoxide/heptachlor), so deviations from such expectations are also further investigated. However, great care should be exercised in using information on congener ratios of common Aroclor mixtures and other such heuristic methods, for some of the same reasons that interpreting environmental PCBs only as mixtures of Aroclors has limitations. Over-reliance on such patterns in data interpretation may lead to inadvertent censoring of data, e.g., for contributions from unknown or unaccounted sources.

When results are reported outside the range of expected relative concentrations, and the laboratory cannot identify the source of variability, values are qualified to indicate uncertainty in the results. If the reported values do not deviate much from the expected range, they are generally allowed to stand and are included in calculations of "sums" for their respective compound classes. However, if the reported concentrations deviate greatly from the expected range and are clearly higher than observed in past analyses or current sample splits, it can be reasonably concluded that the results are erroneous. Again, even "censored" data records are maintained, so any impact of censoring can be reviewed or reversed.

5. Results

The following sections present results from the six monitored tributaries. In the first sub-section, a summary of data quality is initially presented. This is then followed by sub-sections that synthesize climate and flow across the six locations, concentrations of POCs across the six locations, loads across the six locations, and a graphical summary of particle concentrations across the six locations.

5.1 Project Quality Assurance Summary

The section below reports on WY 2014 data; for the WY 2012 and 2013 quality assurance summaries refer to previous reports.

Nutrients

Overall the nutrient data were acceptable. Methods were sufficiently sensitive to detect ambient environmental concentrations. Analytes were not detected in any lab blanks, so field samples did not need qualifying for blank contamination. Some analytes (orthophosphate and phosphorus) were detected in field blanks, with the lowest field samples usually at least about 3x higher than the maximum field blank, except for phosphorus, where the field blank (.057) was only ~20% less than the lowest field sample (.067), and only 6x lower than the average field result. Field blank samples with analyte detection were qualified, but field blanks were not included in all field sample batches so were not used for flagging field results on a batch or whole project basis. However, field blanks should be considered in the interpretation of low concentration samples even if not included in all analytical batches.

Precision on field replicates (generally blind) was good, with RSDs on field replicate samples averaging 15% or better for all the nitrogen analytes and <10% for the phosphorus analytes. Results for matrix spikes and blank spikes were consistent, averaging <10% RSD for all analytes. Recoveries were also good, averaging within 10% of expected values or better for all analytes on matrix spikes, and within ~5% or better for all analytes on laboratory control samples (LCS).

Nutrients - Inter-comparison Study

Overall the data were marginal, with moderate to large deviations for some analytes. Method detection limits were acceptable with no NDs reported. Data were reported not blank corrected. No contamination was measured in any of the method blanks. QC sample types were evaluated according to the preferences noted previously (with greatest preference to sample types most similar in matrix and concentration range as reported field samples, if results for those QC sample types were available in a reportable quantitative range). Lab replicates of field samples were used to evaluate precision for nutrients other than total phosphorus. Average RSDs were good, all less than their respective target MQOs (Nitrate 15%; orthophosphate, and total phosphorus 10%). LCS replicates were used to evaluate the precision of Total Phosphorus results, with average RSD of <1% well within the target MQO (10%). LCS recovery RSDs were examined for nitrate as N and orthophosphate as P but not used for qualifying precision on these analytes, since unspiked lab replicates were quantified for those. Orthophosphate (mean RSD 4.28%) was less than the target MQO (10%), but nitrate as N (mean RSD 22.31%) exceeded 15%, with much of the variation due to different spiking levels on different LCS.

Matrix spikes were used to evaluate the accuracy of total phosphorus results. Recoveries were good with the average recovery errors all within their target MQOs (nitrate 15%, orthophosphate, and total phosphorus 10%). LCS samples were used to assess the accuracy of Nitrate as N and orthophosphate, since these analytes were not in matrix spikes. Recoveries were fair for nitrate (mean error 23.25%, qualified with the non-censoring qualifier of “VIU”) and poor for orthophosphate (mean error 33.95% qualified with the censoring qualifier of “VRIU”). LCS samples were examined for total phosphorus, but not used for qualifying. Total phosphorus (mean error 9.74%) was less than the target MQO (10%).

Suspended Sediment Concentration

Overall the data were acceptable. Method detection limits (MDLs) were sufficient for estimation or quantitation of most samples, with only ~3% of the results reported as NDs. Data were reported not blank corrected. No blank contamination was found in the field or method blanks. LCS replicates were used to evaluate precision, with the average RSD (3.62%) being well below the target method quality objective (MQO) of 10%. The average RSD for field replicates was not used in the evaluation, but was examined and found to be 7.5%. No qualifiers were added. LCS were used to assess accuracy as they were the only spiked samples analyzed. Recoveries measured were good with the average recovery error of 2.93% being well below the target 10% MQO. No qualifiers were needed.

Suspended Sediment Concentration - Inter-comparison Study

Method detection limits were acceptable with no NDs reported. Data were reported not blank corrected. No contamination was measured in the method blanks. Lacking other sample types analyzed in replicate, CRM recoveries were used to evaluate the precision of Suspended Sediment Concentration results, and had an average RSD of 23.6%. Although this was more than double the target MQO of 10%, they were qualified with the non-censoring qualifier of “VIL” since the CRMs were certified at different target values and thus might not be expected to show similar recoveries. CRMs were used to assess the accuracy of the suspended sediment concentration results. Recoveries measured were fair with the average recovery error of 16.23% being greater than the target MQO of 10%, but less than 20%, so were qualified with the non-censoring qualifier of “VIU”.

Total Organic Carbon

The TOC data were acceptable. MDLs were sufficient with zero NDs reported. Data were reported not blank corrected. Blank contamination was not measured in the method blanks. Equipment and field blanks were examined, but not used in qualifying field sample results in the database. Blank contamination was found in one of the seven equipment blanks at a level ~3% of those found in the field samples (equipment blank contamination 0.51 mg/L compared to mean field sample concentration 15.74 mg/L). No blank contamination was measured in the field blank.

Precision was evaluated using matrix spike replicates. The RSD was good averaging 0.47%; less than the MQO of 10%. No qualifiers were needed. LCS replicates and blind field replicates had an average RSD of 3.87% and 2.97%, respectively. Matrix spike samples were used to assess accuracy as no CRMs were analyzed. Recoveries measured were good with the average recovery error of 6.88% being less than the target MQO of 10%. No qualifiers were needed. LCS recoveries were good with an average recovery error of 2.22%.

Copper, Selenium, and Total Hardness

The copper, selenium, and total hardness data were acceptable. Samples were either field filtered, or lab filtered within 24 hours except for 1 field blank and one sample, qualified for being slightly over (25-26 hours) the target filtering hold time. MDLs were sufficient with zero NDs reported. Data were reported not blank corrected. Blank contamination was not measured in the method blanks.

Equipment and field blanks were examined, but not used in the qualifying of field samples. Blank contamination was found in several of the field blanks for copper (dissolved and total) at a level ~20% of those found in the field samples for dissolved copper (mean field blank contamination 1.4 ug/L compared to mean field sample concentration 7 ug/L), and at a level ~2% of those found in the field samples for total copper (mean field blank contamination 0.6 ug/L compared to mean field sample concentration of 28 ug/L). No blank contamination was measured in the equipment blanks.

Precision was evaluated using the matrix spike replicates, with the average RSDs being well less than the target MQOs (selenium 35%, copper 25%, and hardness 5%); all <2%. Average RSDs for LCS replicates were also less than the target MQOs; all <5%. The average RSDs for field replicates were not used in qualified, but were examined and found to be less than the target MQOs; all <5%. No precision qualifiers were added. Matrix spike samples were used to assess accuracy as no CRMs were analyzed. Recoveries measured were good with average recovery errors less than the target MQOs (selenium 35%, copper 25%, and hardness 5%). LCS recoveries were also good with average recovery errors all less than the target MQOs. No recovery qualifiers were needed. Dissolved and total fractions were reported for copper and selenium. Dissolved/Total ratios were all < 1.35, within the propagated accepted error for precision and accuracy on individual results.

Copper, Selenium, and Total Hardness - Inter-comparison Study

Overall the data were acceptable. MDLs were sufficient with zero NDs reported. One batch had selenium detected in blanks slightly over the MDL, but still well below most field sample concentrations. The data were blank corrected and the blank standard deviation was less than the MDL so no blank qualifiers were added. Lab replicates were used to evaluate precision, with the average RSDs being all <4%, well below the target MQOs (selenium 35%; calcium, copper, and magnesium 25%). Average RSDs for matrix spike/matrix spike replicate samples were all <4%, also less than the target MQOs. No precision qualifiers were added. CRMs were used to assess accuracy. Recoveries measured were good with average recovery errors less than the target MQOs (selenium 35%; calcium, copper, and magnesium 25%); the highest recovery error was 12% for calcium (to calculate hardness). Matrix spike and LCS recoveries were good with average recovery errors all less than the target MQOs. No added qualifiers were needed. Dissolved and total fractions were reported for copper and selenium. Dissolved/Total ratios were all < 1.35, within precision expected propagated error.

Mercury and Methylmercury

The total mercury (Hg) and methylmercury (MeHg) data overall are acceptable. All were analyzed within the recommended 28 day hold time aside from one mercury sample analyzed slightly beyond (35 days) that was qualified for hold time. The methods were sufficiently sensitive to detect MeHg or Hg in nearly all samples, with only 2 MeHg analyses reported not detected. Blank concentrations of MeHg and total Hg were below detection limits for all blank sample types (field, equipment, and lab), so no blank qualifiers were needed.

Precision on field replicates was acceptable, averaging 16% RSD for both total and methyl mercury. Matrix spike/MSD precision averaged 2% RSD, and LCS (spiked blank) precision was similarly good, averaging 4% and 12% for total and methyl mercury, respectively. No CRMs were analyzed, so matrix spikes were the best indicators of recovery available. Although a few individual sample recoveries were outside of the target range (due to spiking less than 2x native concentrations), recovery errors averaged 11% or better for MeHg and total Hg matrix spikes and spike duplicates spiked higher than 2x, and averaged 9% or better for blank spikes, well within target errors of +/-35%. No added qualifiers were needed. The ratios of methyl to total mercury were within an expected reasonable range, with methyl mercury (around 0.2 ng/L) near 1% or less of total mercury (0.05 ug/L = 50 ng/L, around 250x higher).

Mercury and Methylmercury – Inter-comparison Study

Overall the data were quite good. MDLs were sufficient that there were no NDs for field samples. Methylmercury (MeHg) and mercury (Hg) were not detected in most blanks, except 1 just at its MDL, although the blank average for that batch was still <MDL. Precision on an un-spiked lab replicate was good, with an RSD <3%. Precision on repeated measures of CRMs, MS and LCS were similarly good, all averaging <3%, well within the target 35% MQO for Hg and MeHg. Recoveries on CRMs, MSs, and LCS were all good, with average errors <5% for Hg, and <15% for MeHg, well within the target <35%. The ratios of mercury and methylmercury were pretty typical, with methylmercury <1% of total mercury (although they weren't necessarily reported as pairs for a given site and event in the IC samples).

Carbaryl and Fipronil

Overall the carbaryl and fipronil data were acceptable. Methods were sufficient to detect at least some target analytes in most samples. Fipronil was always detected. None of the target analytes were detected in blanks. Precision on field replicates was generally good, with RSDs <35% target for all analytes. Carbaryl had the highest variation (30%) due to concentrations near the MDL. Precision on MS/MSD and LCS replicates was better yet, <20% RSD for all analytes. Recovery errors on all reported analytes averaged less than the 35% target so no added qualifiers were needed

PAHs

Overall the PAH data were marginally acceptable. MDLs were sufficient with 5 of the 44 reported PAHs having NDs (ranging from 6 to 53% ND per PAH congener), with only 1, Benz(a)anthracene having >=50% ND. Blank contamination was measured in at least one of the seven method blanks for many analytes with blank contamination high enough (>1/3 of the field sample result) to qualify many results (88% of Biphenyl, but 29% or less for other PAHs and alkylated PAHs) with the censoring contamination qualifier of “VRIP”. Many of these censored results were the alkylated PAHs, not used in generating sums of PAHs; the other censored LPAH and HPAH results typically account for about 10% of total PAHs.

Field blanks were examined, but not used in qualifying field samples in the database. Contamination in the field blanks was found at concentrations mostly 1-4 times that found in the lab blanks, except for 2,6-Dimethylnaphthalene, 2-Methylnaphthalene, 1-Methylnaphthalene, C1-Naphthalenes, and Naphthalene, which were respectively 5, 6, 7, 8 and 8 times greater in the field blanks than the lab blanks. Average field blank contaminant concentrations were generally less than 10% of the average concentrations found in the field samples, notable exceptions were 1-Methylnaphthalene, C1-Naphthalenes, 2-Methylnaphthalene, Biphenyl, and Naphthalene, which were 22%, 24%, 26%, 27% and 58% of the average field sample concentrations, respectively.

Replicates on field samples were used to evaluate precision and were good, less than the target 35% average RSD. LCS replicates were examined and were also all less than the target 35% average RSD (all <10%). The average RSD combining field and lab duplicates were not used in qualifying, but were less than the target MQO of 35%. No precision qualifiers were added. LCS were used to assess the accuracy of PAHs as no CRMs or matrix spikes were reported. Recoveries measured in the LCS were good with recovery errors less than the target 35% for all 44 PAHs measured (all <20%). No recovery qualifiers were added. Alkyalted PAHs were not included in the LCS or other recovery samples so were qualified with the QA code of “VBS” and batch verification code of “VQI” for partial/unknown recovery QA.

PBDEs

The PBDE data were overall acceptable. MDLs were sufficient with 29 of the 49 reported total fraction PBDE congeners having NDs (ranging from 6 to 100% ND), and 27% (13 out of 49) having >=50% ND. PBDE congeners 28, 47, 49, 71, 85, 99, 100, 116, 119, 126, 140, 153, 154, 155, 183, 190, 197, 205, 206, 208, and 209 had some contamination in at least one method blank, but the blank contamination was only bad enough to qualify 58% of PBDE 190 and 205, 41% of PBDE 126, 29% of PBDE 116, 24% of PBDE 140, 12% of PBDE 155, and 6% of PBDE 71 and 199 results with the censoring contamination qualifier of “VRIP” (results with reported concentrations <3x the blank results (by batch) being censored for contamination).

Field blanks were examined, but not used in qualifying. Blank contamination was found in at least one field blank for PBDE congeners 17, 28, 47, 49, 85, 99, 100, 119, 140, 153, 154, 155, 203, 206, 207, and 209. Field blank contamination was found at concentrations mostly 1-4 times that found in the lab blanks, except for PBDE 049 and 085, which were respectively 13 and 10 times greater in the field blanks than the lab blanks. However, this was still well below the concentrations found in the field samples; average field blank contaminant concentrations at most were 2.3% (PBDE 049) of the average concentrations found in the field samples.

Lab replicates on field samples were used to evaluate precision and were generally good, less than the target 35% average RSD (PBDE 008 was just below at 34.9%). Replicates of the eight usable LCS were examined and were all <35% average RSD (all <16%). The average RSD combining all field and lab duplicates were not used in qualifying (since lab replicates alone are more representative of purely analytical issues) but were examined and found to be less than the target MQO of 35%, except for PBDE 138 (RSD 35.6%). No precision qualifiers were added. LCS results were used to assess the accuracy of PBDEs as no CRMs or matrix spikes were reported. Recoveries for the eight PBDEs measured in the LCS were good with recovery errors less than the target 35% for all reported analytes (all <15%). LCS results for PBDE 33 were unusable. No additional qualifiers were needed.

PCBs

Overall the PCB data were acceptable. MDLs were sufficient with NDs being reported for 15.5% (11 out of 71) PCB congeners ranging from 1% to 3.5% ND; none were extensive (>=50% ND). Blank contamination was measured in at least one method blank for many PCBs (8, 18, 28, 31, 33, 44, 49, 52, 56, 60, 66, 70, 87, 95, 99, 101, 105, 110, 118, 128, 132, 138, 149, 151, 153, 156, 158, 170, 174, 177, 180, 183, and 187). Contamination was over 1/3 of the field sample result in 1% to 11% of PCB 8, 18, 28, 31, 33, 44, 49, 52, 56, 60, 66, 70, 87, 95, 99, 101, 105, 110, 118, 151, and 177 samples and qualified with the censoring qualifier of “VRIP”.

Field blanks were examined, but not used in qualifying results in the database. Blank contamination was found in the field blanks at levels generally less than in the method blanks and at levels well below those found in the field samples (< 1%). Lab replicates of field samples were used to evaluate precision, with the average RSD being less than the target MQO (35%); all <30%. Average RSD for LCS replicates were examined, and were less than the target MQO of 35%; all <10%. The average RSD for field replicates were not used in qualifying, but were examined and found to be less than the target MQO; all <22%. No precision qualifiers were added. LCS results were used to assess accuracy as no CRMs, or matrix spikes were analyzed. Recoveries measured were good with recovery errors less than the target MQO (35%); all <8%. No additional recovery qualifiers were needed.

Pyrethroids

Overall the pesticide data were acceptable. NDs were reported for all 11 pyrethroids ranging from 7% to 100% ND; NDs for Allethrin, Total Esfenvalerate/Fenvalerate, Fenpropathrin, Tetramethrin, and T-Fluvalinate were extensive (>=50% ND). Data were reported not blank corrected. Blank contamination was measured in at least one method blank for Total lambda-Cyhalothrin. Contamination was extensive enough so that 20% of Total lambda-Cyhalothrin results were qualified with the censoring qualifier of "VRIP" (results with reported concentrations <3x the blank results (by batch) being censored for contamination). Field blanks were examined, but not used in qualifying. Blank contamination was found in the field blank for Total lambda-Cyhalothrin at levels ~40% of those found in the method blanks (0.11 ng/L compared to 0.26 and 0.28 ng/L), and at a level below those found in the field samples (average field sample concentration 0.62 ng/L, field blank contamination 0.11 ng/L).

Matrix spike replicates were used to evaluate precision, with the average RSD being well less than the target MQO (35%); all <12%. Average RSD for LCS replicates were examined, and were less than the target MQO of 35%; all <14%. The average RSD for field replicates were not used in qualifying, but were examined and found to be less than the target MQO (35%); all <30%. No precision qualifiers were added. Accuracy was assessed using the matrix spike samples as no CRMs were analyzed. Recoveries measured were generally good with average recovery errors less than the target MQO (35%); except for Total lambda-Cyhalothrin (42%) and T-Fluvalinate (41%) which were qualified with the non-censoring qualifier of "VIU". LCS recoveries were good with average recovery errors all less than 30%.

Pyrethroids – Inter-comparison Study

Overall the data were acceptable. Most pyrethroids were 100% ND, except for Bifenthrin, Deltamethrin/Tralomethrin, and Total Permethrin (Tetramethrin was qualified by the laboratory as an unreportable estimate). Data were reported not blank corrected. No contamination was measured in the one method blank. No replicates of any kind were analyzed so precision could not be evaluated; results were qualified with the QA code of "VBS" for incomplete QC. The LCS was used to assess accuracy as no CRMs or matrix spikes were analyzed. Recoveries measured were generally good with average recovery errors less than the target MQO (35%); all were <24%.

Toxicity

The 36 hour recommended hold times were exceeded for some sets of *Hyalella azteca* (up to 53 hour hold time) and *Pimephales promelas* (up to 74 hours), and up to 1-2 hour slight exceedances for the other species. Results exceeding the recommended 36 hour hold time were qualified. Control survival was acceptable with a minimum 80% survival just meeting the 80% requirement in one batch. Other batches had higher survival up to 100%. Water quality limits for the test species were not exceeded in any tests. Reference toxicant control EC_{50}/LC_{50} were within the mean+/-2stdev of previous control results ("typical response" range).

5.2 Climate and flow at the sampling locations during water years 2012, 2013, and 2014

The climatic conditions under which observations are made of pollutant concentrations in flowing river systems have a large bearing on concentrations and loads observed. It has been argued that a 30 year period is needed in California to capture the majority of climate related variability of a single site (Inman and Jenkins, 1999; McKee et al., 2003). Given monitoring programs for concentrations or loads do not normally continue for such a long period (except for rare occasions for turbidity and suspended sediment (e.g. Santa Anna River, Southern California: Warrick and Rubin 2007; Casper Creek, northern California: Keppeler, 2012; Alameda Creek at Niles (data for WYs 1957-73 and 2000-present (30 years)), the objective of sampling is usually to try to capture sufficient components of the full spectrum of variability to make inferences from a smaller dataset. When such data are available, they usually reveal complex patterns in relation to rare large events or several periods of rare drought and decadal scale changes to climate and land use or water management (Inman and Jenkins, 1999; McKee et al., 2003; Warrick and Rubin 2007; Keppeler, 2012; Warrick et al., 2013). However, in general for pollutant, data sets are rarely longer than a few years and high magnitude (high intensity or long duration) events occur infrequently and thus are usually poorly represented. Unfortunately, these types of events usually transport the majority of a decadal scale loads (Inman and Jenkins, 1999; Warrick and Rubin 2007). This occurs because the discharge-load relation spans 2-3 orders of magnitude on the discharge axis and often 3-4 orders of magnitude on the sediment load axis and is described best by a power function ($Q_s=aQ_w^b$) where a and b are constants that describe pollutant sources and the erosive power of water. Therefore storms and wet years with larger discharge, if measured, have a profound influence on the estimate of mean annual load for a given site and would likely confound any comparisons of loads between sites unless adequately characterized. However, if it is assumed that this is consistently true for all sites, or loads measured during dry years can be "climatically adjusted", the validity of loads comparisons between sites will be increased.

Conceptually, watersheds that are more impervious, or smaller in area, or have lower pollutant production variability (and lower source complexity) should exhibit lower inter-annual variability (lower slope of the power function) and therefore require less sampling to adequately quantify pollutant source-release-transport processes (an example in this group is Marsh Creek which has rural and recent urbanization land uses and few suspected source areas for PCBs). In contrast, a longer sampling period spanning a wider climatic variability would be more ideal to adequately describe pollutant source-release-transport processes in watersheds that are larger, or less impervious, or have large and known pollutant sources. The quintessential example of this category within this study is Guadalupe River in relation to Hg sources, release mechanisms, and loads but San Leandro Creek (both Hg and PCBs) and East Sunnyvale Channel and Pulgas Pump Station - South (PCBs) also appear to be in this category. Marsh Creek also appears to be in this category in relation to suspended sediment. Concentration variability relative to first flush and storm magnitude-frequency-duration will probably remain unexplainable for these analytes, even after three years of sampling. This will be one factor that may lead to lower confidence in annual loads computations and average annual loads estimates.

Unfortunately, during the three year study, winter seasons have been very dry relative to average annual conditions with all observations to-date made during years of between 38-85% mean annual precipitation and 22-82% mean annual flow (Table 6). For example, San Leandro Creek experienced 75% of mean annual runoff (MAR) in WY 2012, 67% MAR in WY 2013, and 52% MAR in WY 2014. However, there have been some notable storms, particularly those occurring during late November and December of WY 2013, an intense first flush in November 2013 (WY 2014) and another relatively intense storm in late February 2014 (WY 2014). For example, approximately 52% of the total wet season rainfall fell at the East Sunnyvale Channel rain gauge over 11 days during November and December of WY 2013 and 13% on February 28, 2014 (WY2014). Loads of pollutants were disproportionately transported during such events; at East Sunnyvale Channel, 96%, 91% and 84% of the WY 2013 total wet season sediment, PCBs and mercury loads were transported during those larger November and December storms and 30%, 58% and 24% of the total wet season sediment, PCBs, and mercury loads were transported in a single day on February 28 in WY 2014. However, despite these larger individual storm events, the overall drought conditions during the study may result in estimated long-term averages for each site that are biased low due relatively benign flow production, sediment erosion, and transport conditions in all six watersheds. The bias may not be as severe in those watersheds that received slightly wetter conditions and/or that are more impervious.

Table 6. Climate and flow during sampling years at each sampling location.

Water Year (WY)		Marsh Creek[2]	North Richmond Pump Station[3]	San Leandro Creek[4]	Guadalupe River[5]	East Sunnyvale Channel[6]	Pulgas Pump Station - South[7]
Rainfall (mm) (% mean annual)	2012	320 (71%)	NA	486 (75%)	179 (47%)	224 (60%)	NA
	2013	344 (76%)	493 (85%)	437 (67%)	223 (59%)	307 (82%)	378 (78%)
	2014	260 (58%)	327 (57%)	338 (52%)	161 (43%)	207 (55%)	183 (38%)
	Mean Annual	457	578	627	378	387	484
Runoff (Mm^3) (% mean annual)	2012	1.87 (22%)	NA	7.30	38.0 (68%)	1.07	NA
	2013	6.23 (73%)	0.74	7.21	45.45 (82%)	1.51	0.22
	2014	1.17* (15%)	0.50	0.24	16.75* (30%)	1.01	0.08
	Mean Annual	8.0	No long term data	No long term data	55.6	No long term data	No long term data

[1] Unless otherwise stated, averages are for the period Climate Year (CY) (Jul-Jun) (rainfall) or Water Year (WY) (Oct-Sep) (runoff) 1971-2010.
[2] Rainfall gauge: Concord Wastewater treatment plant (NOAA gauge number 041967) (CY 1991-2013); Runoff gauge: Marsh Creek at Brentwood (gauge number 11337600) (WY 2001-2013).
[3] Rainfall gauge: This study with mean annual from modeled PRISM data; Runoff gauge: This study.
[4] Rainfall gauge: Upper San Leandro Filter (gauge number 049185); Runoff gauge: This study.
[5] Rainfall gauge: San Jose (NOAA gauge number 047821); Runoff gauge: Guadalupe River at San Jose (gauge number 11169000) and at Hwy 101 (gauge number 11169025).
[6] Rainfall gauge: Palo Alto (NOAA gauge number 046646); Runoff gauge: This study
[7] Rainfall gauge: Redwood City NCDC (gauge number 047339-4); Runoff gauge: This study.
* indicates data missing for the latter few months of the season

5.3 Concentrations of pollutants of concern

Understanding the concentrations of pollutants in the watersheds is important to both directly answer MQ2 of the Small Tributary Loading Strategy management questions as well as form the basis from which to answer all of the other key management questions identified by the Strategy. The three year sampling program has provided data that, in some cases, indicate surprisingly high concentrations (e.g. Hg in San Leandro Creek; PCBs in East Sunnyvale Channel; PBDEs in North Richmond Pump Station); in other cases indicate surprisingly low concentrations (Hg in Marsh Creek). While in still other cases, sampling has somewhat verified what was expected (North Richmond (PCBs and Hg), Guadalupe (PCBs and Hg) and Pulgas Pump Station - South (PCBs)). In some cases NDs and quality assurance issues confound robust interpretations. This section explores these issues through synthesis of data collected across all six sampling locations over the three years.

Concentrations of pollutants typically vary over the course of a storm and between storms of varying magnitudes, and are dependent on antecedent rainfall, soil moisture conditions, related discharge, sediment supply and transport, and pollutant source-release-transport processes. Although these can be fully understood over a long period of sampling that covers a wide range of conditions, shorter sampling programs will fail to capture this variability and therefore concentrations may appear complex or even chaotic and interpretation may remain difficult. Thus, it is important, even during shorter sampling programs, to sample over a wide range of flow conditions both within and between storms to adequately characterize concentrations of pollutants in a watershed.

The monitoring design for this project aimed to collect pollutant concentration data from 12 storms over the span of three years (except for North Richmond Pump Station and Pulgas Pump Station - South, each with a target of 8 storm events), with priority pollutants sampled at an average of four samples per storm for a total of 48 discrete samples collected during the monitoring term. In order to capture as much variability as possible, the program aimed to sample earlier season storms, several larger (preferably) or “mid-season” storms, and a later season storm each year for each site (Melwani et al 2010; BASMAA, 2011), However, due to dry conditions, these aims were not easily met. Sampling at the six locations over the three water years has included sampling between 7-10 storm events at each location (Table 7). North Richmond Pump Station was the only site where the full allotment of storm events was completed (n=8). Given the small sample size and varying sample sizes between sites, and the failure in some cases to collect a full sample set across the desired storm conditions, the following synthesis represents the best available knowledge about these sites; and areas where gaps in knowledge remain are identified.

Overall, detections of concentrations in the priority pollutants (suspended sediment, total PCBs, total mercury, total methylmercury, total organic carbon, total phosphorous, nitrate, and phosphate) were all 90% or higher, as were detections of several of the “tier II” pollutants (total and dissolved copper and selenium, PAHs and PBDEs) (Table 8). Numerous pyrethroids were not detected at any of the sites; whereas, Delta/Tralomethrin, Cypermethrin, Cyhalothrin lambda, Permethrin, Bifenthrin as well as Carbaryl and Fipronil were all detected in one or more samples at each sampling location.

The two highly urban and impervious sampling locations added in WY 2013 and also sampled in WY 2014 (North Richmond and Pulgas pump stations), have the lowest mean SSC; whereas, pollutant concentrations are relatively high for these watersheds (e.g. PCBs at Pulgas Pump Station - South). In contrast, East Sunnyvale Channel has high PCB concentrations but also relatively high SSC. As a result,

Table 7. Number of storms sampled and number of discrete samples collected at each location relative to the program objectives as recommended (Melwani et al 2010) and codified in the multi-year-plan (e.g. BASMAA, 2011).

Water Year	Storm category	Marsh Creek	North Richmond Pump Station	San Leandro Creek	Guadalupe River	East Sunnyvale Channel	Pulgas Pump Station – South
2012	Early season or "first flush	No	Study not yet begun	No	No	No	Study not yet begun
	Larger or mid-season	Yes		Yes	Yes	Yes	
	Later season	Yes		Yes	Yes	Yes	
2013	Early season or "first flush	Yes	Yes	Yes	Yes	Yes	No
	Larger or mid-season	Yes	Yes	Yes	Yes	Yes	No
	Later season	Yes	Yes	Yes	Yes	No	Yes
2014	Early season or "first flush	No	Yes	No	Yes	No	Yes
	Larger or mid-season	Yes	Yes	Yes	Yes	Yes	Yes
	Later season	No	No	No	Yes	Yes	Yes
	Total number of discrete samples	31 out of 48	32 out of 32	44 out of 48	39 out of 48	40 out of 48	28 out of 32

the particle ratio (turbidity or SSC to pollutant; discussed further in section 5.5) rank shows a differing order to the water concentration ranking. Given the high imperviousness and small size of the North Richmond and Pulgas Pump Station - South watersheds, although fewer storms have been sampled at these locations, it is unlikely greater variation in SSC would be observed even if they were to be sampled again in the future.

Mean HgT concentrations across the six sites varied between 18.2 – 212 ng/L (Table 8, Figure 2). Guadalupe River contains historic mercury mines in the upper watershed and is a known mercury source to the San Francisco Bay, explaining the relatively high mercury and, possibly, methylmercury concentrations (Thomas et al., 2002; Conaway et al., 2003; Davis et al., 2012). Less clear is what the source of mercury is in San Leandro Creek, which has mercury and methylmercury concentrations nearly as high as Guadalupe River. If sampling in San Leandro Creek were to continue at some point in the future, under more variable storm and climatic conditions, an improved understanding of source-release-transport processes of mercury in this watershed may be generated that would help to isolate natural or anthropogenic mercury sources. Although HgT and MeHgT concentrations did not even near the USEPA Criteria for Water or the EPA California Toxics Rule Freshwater Aquatic Life Protection criteria (respectively), the San Francisco Bay has more stringent criteria related to the San Francisco Bay TMDL. The median particle ratio concentrations of HgT exceed the San Francisco Bay TMDL target (0.2 mg/kg)

Table 8. Synthesis of concentrations of pollutants of concern based on all quality assured data collected over the three sampling years at each location. Table notes on following page.

			Lower Marsh Creek		Richmond Pump Station		San Leandro Creek		Guadalupe River		East Sunnyvale Channel		Pulgas Creek		
Analyte Name	Unit	Reporting Limit	Number (% detect)	Mean (std.error)	Number (% detect)	Mean (std.error)	Number (% detect)	Mean (std.error)	Number (% detect)	Mean (std.error)	Number (% detect)	Mean (std.error)	Number (% detect)	Mean (std.error)	Water Quality Benchmark
SSC	mg/L	3	101 (94%)	204 (23.5)	108 (97%)	56.8 (5.57)	117 (95%)	115 (13.8)	136 (100%)	157 (12.3)	137 (98%)	232 (31.4)	96 (99%)	56.5 (6.27)	NA
ΣPCB	ng/L	NA	22 (100%)	1.25 (0.258)	32 (100%)	13.8 (1.57)	44 (100%)	8.01 (1.16)	39 (100%)	14.3 (2.4)	40 (100%)	104 (27.5)	29 (100%)	505 (261)	14 ng/L (a); 0.17 ng/L (b) 1 ug/kg * SF Bay TMDL
Total Hg	ng/L	0.526	31 (100%)	38.4 (9.62)	32 (100%)	39.6 (7.8)	44 (100%)	106 (24.2)	39 (100%)	212 (35.9)	40 (100%)	47.6 (6.68)	31 (100%)	18.2 (2.39)	2400 ng/L (c) 0.2 mg/kg * SF Bay TMDL
Total MeHg	ng/L	0.0401	20 (90%)	0.291 (0.0741)	16 (100%)	0.208 -0.0633	30 (100%)	0.397 (0.0663)	27 (100%)	0.504 (0.0677)	27 (93%)	0.295 (0.0376)	20 (100%)	0.189 (0.033)	1400 ng/L (a)
TOC	mg/L	0.481	30 (100%)	7.13 (0.34)	32 (100%)	11.2 (1.82)	44 (100%)	8.24 (0.462)	40 (100%)	12.2 (1.96)	40 (100%)	10.1 (1.1)	28 (100%)	20.5 (5.54)	NA
NO3	mg/L	0.0488	28 (96%)	0.569 (0.0402)	32 (100%)	0.976 (0.143)	45 (100%)	0.425 (0.0659)	36 (100%)	0.917 (0.099)	41 (100%)	0.472 (0.0872)	28 (100%)	0.466 (0.0864)	10 mg/L (d) 1.1 mg/L (e)
Total P	mg/L	0.01	30 (100%)	0.415 (0.0441)	32 (100%)	0.384 (0.0256)	44 (100%)	0.288 (0.024)	40 (100%)	0.414 (0.0376)	41 (100%)	0.411 (0.0429)	28 (100%)	0.29 (0.047)	0.03 mg/L (f)
PO4	mg/L	0.0112	30 (100%)	0.0987 (0.0074)	31 (100%)	0.218 (0.0141)	45 (100%)	0.1 (0.00412)	40 (100%)	0.15 (0.0156)	41 (100%)	0.128 (0.00905)	28 (100%)	0.124 (0.0189)	NA
Hardness	mg/L	NA	4 (100%)	176 (19.3)	5 (100%)	129 (38.6)	8 (100%)	56.5 (4.94)	7 (100%)	138 (12.7)	8 (100%)	124 (32.6)	6 (100%)	69.8 (12)	NA
Total Cu	ug/L	0.527	8 (100%)	13.7 (3.59)	8 (100%)	22.5 (4.49)	11 (100%)	16.2 (3.07)	10 (100%)	21.6 (2.87)	10 (100%)	17.9 (1.88)	7 (100%)	43.9 (10.1)	13 00 ug/L (b)
Dissolved Cu	ug/L	0.5	8 (100%)	2.74 (0.588)	8 (100%)	8.45 (1.53)	11 (100%)	5.98 (0.682)	10 (100%)	5 (0.939)	10 (100%)	5.5 (1.09)	7 (100%)	18.6 (3.91)	13 ug/L (g)
Total Se	ug/L	0.0925	8 (100%)	0.742 (0.103)	8 (100%)	0.409 (0.0638)	11 (100%)	0.223 (0.019)	10 (100%)	1.31 (0.252)	10 (100%)	0.606 (0.147)	7 (100%)	0.292 (0.0632)	20 ug/L (g)
Dissolved Se	ug/L	0.124	8 (100%)	0.647 (0.0886)	8 (100%)	0.366 (0.0586)	11 (100%)	0.166 (0.0149)	10 (100%)	1.07 (0.266)	10 (100%)	0.519 (0.146)	7 (100%)	0.244 (0.0526)	3.1 ug/L (h)
Carbaryl	ng/L	20	8 (25%)	3.63 (2.39)	8 (88%)	21.6 (4.72)	12 (50%)	5.82 (2.11)	10 (90%)	29.5 (6.87)	10 (40%)	6.5 (2.78)	7 (100%)	105 (26.3)	850 ng/L (i) 2530 ng/L (j)
Fipronil	ng/L	4.34	8 (100%)	12.2 (1.19)	8 (75%)	6.31 (1.92)	11 (91%)	10.1 (1.89)	10 (100%)	11.3 (1.56)	10 (90%)	6.5 (1.13)	7 (86%)	3.29 (0.68)	110 ng/L (i)
ΣPAH	ng/L	NA	4 (100%)	140 (46.5)	4 (100%)	527 (279)	5 (100%)	1260 (494)	11 (100%)	416 (116)	6 (100%)	1350 (455)	6 (100%)	1660 (1070)	8.8 ng/L (d)
ΣPBDE	ng/L	NA	4 (100%)	27 (10.1)	5 (100%)	789 (644)	5 (100%)	28.5 (11.7)	5 (100%)	60.8 (18.3)	6 (100%)	47 (16)	6 (100%)	45.6 (13.1)	NA
Delta/ Tralomethrin	ng/L	3.05	8 (75%)	1.5 (0.637)	8 (75%)	2.29 (0.818)	10 (40%)	0.391 (0.207)	10 (50%)	0.852 (0.328)	9 (89%)	1.77 (0.469)	7 (43%)	0.386 (0.205)	55 ng/L (i)
Cypermethrin	ng/L	1.53	8 (88%)	11.7 (8.24)	8 (100%)	4.84 (1.38)	11 (55%)	0.368 (0.115)	10 (70%)	1.49 (0.512)	10 (80%)	3.29 (0.63)	7 (100%)	2.42 (0.663)	195 ng/L (i)
Cyhalothrin lambda	ng/L	1.5	7 (86%)	1.23 (0.486)	7 (100%)	1.1 (0.228)	9 (56%)	0.616 (0.376)	10 (70%)	0.556 (0.174)	8 (75%)	0.656 (0.296)	6 (83%)	0.35 (0.12)	3.5 ng/L (i)
Permethrin	ng/L	15	8 (75%)	6.08 (2.29)	8 (100%)	17.7 (5.91)	11 (55%)	3.59 (1.24)	10 (80%)	10.5 (2.34)	10 (100%)	21.8 (3.61)	7 (86%)	10.7 (3.03)	10.6 ng/L (i)
Bifenthrin	ng/L	1.53	8 (100%)	75.2 (29.9)	8 (100%)	5.88 (0.796)	11 (91%)	8.08 (2.69)	10 (90%)	5.29 (1.18)	10 (90%)	8.01 (1.95)	7 (100%)	5.14 (1.81)	75 ng/L (i)

Table Notes:

Analyzed but not detected: Fenpropathrin, Esfenvalerate/Fenvalerate, Cyfluthrin, Allethrin, Prallethrin, Phenothrin, Resmethrin.
All Hardness results in WY 2013 were censored.
Sources:
(a) EPA California Toxics Rule, California Inland Surface Waters - Freshwater Aquatic Life Protection (4-day average); http://www.waterboards.ca.gov/water_issues/programs/water_quality_goals/search.shtml
(b) EPA California Toxics Rule, California Inland Surface Waters - Human Health Protection (30-day average); http://www.waterboards.ca.gov/water_issues/programs/water_quality_goals/search.shtml
(c) USEPA criteria for Water 1986 (EPA 440/5-86-001) - referenced in Basin Plan; http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/basinplan/web/bp_ch3.shtml
(d) EPA National Recommended Water Quality Criteria - Human Health and Welfare Protection; http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/basinplan/web/bp_ch3.shtml
(e) McKee and Krottje, 2008. This is not a recognized water quality benchmark, but the review by these authors indicated that nitrate (NO3-) may be chronically toxic to aquatic life, especially fish and amphibian eggs, at concentrations as low as 1,100 ug/L.
(f) EPA Nutrient criteria for Level III Eco-Region 6 (EPA, 2000); discussed in McKee and Krottje, 2008.
(g) National Toxics Rule - referenced in Basin Plan; http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/planningtmdls/basinplan/web/bp_ch3.shtml
(h) EPA Draft Aquatic Life Ambient Water Quality Criterion for Selenium (Freshwater) 2015 (a 30-day average); https://www.epa.gov/wqc/aquatic-life-criterion-selenium-documents
(i) USEPA Office of Pesticide Programs' Aquatic Life Benchmarks (USEPA, 2014) – invertebrates; https://www.epa.gov/pesticide-science-and-assessing-pesticide-risks/aquatic-life-benchmarks-pesticide-registration#benchmarks
(j) EPA National Recommended Water Quality Criteria - Freshwater Aquatic Life Protection; http://www.waterboards.ca.gov/water_issues/programs/water_quality_goals/search.shtml

in all watersheds except for Lower Marsh Creek (Figure 2). San Francisco Bay TMDL targets for MeHgT are associated with fish tissue concentrations, not stormwater, and therefore were not evaluated here.

Consistent with our conceptual model that PCB sources are more variable in the landscape than mercury, the variation in mean PCB concentrations across the six watersheds was much greater than HgT, varying between 1.25 and 505 ng/L. The maximum PCB concentration observed during the three year program (6,669 ng/L) was collected in Pulgas Pump Station - South, which also has the greatest mean PCB concentration of the six locations. This maximum result was collected during a storm when concentrations were an order of magnitude higher than results from any other storm sampled at the station, and it is unclear why this storm in particular mobilized such high concentrations given that the storm was relatively small in magnitude (0.42 inches), intensity (maximum 1 hour rainfall 0.11 inches) and peak flow rate (8.6 cfs relative to other PCB samples collected at flows as high as 17 cfs). Also notable is that sampling at Pulgas Pump Station - South during WYs 2013 and 2014 characterized relatively small storm events (one during WY 2013) and during a very dry period; given that PCBs are dominantly associated with particles and that particle transport is correlated with rainfall magnitude and intensity (as seen at Zone 4 Line A[3] (Gilbreath et al., 2012)) it is possible that additional sampling during a wetter period and more intense storm events could show even greater concentrations. When evaluated against the California Toxics Rule for California Inland Surface Waters (Freshwater Aquatic Life Protection) 4-day average criterion (14 ng/L), the median concentration for all six stations falls just

[3] Zone 4 Line A is a 4.2 km^2 100% urban tributary located in Hayward, CA. This creek was monitored extensively by the RMP between WYs 2007-2010 using a similar study approach to estimate loads as the one reported here. It presents one of the most robust datasets available in the Bay Area for multiple pollutants.

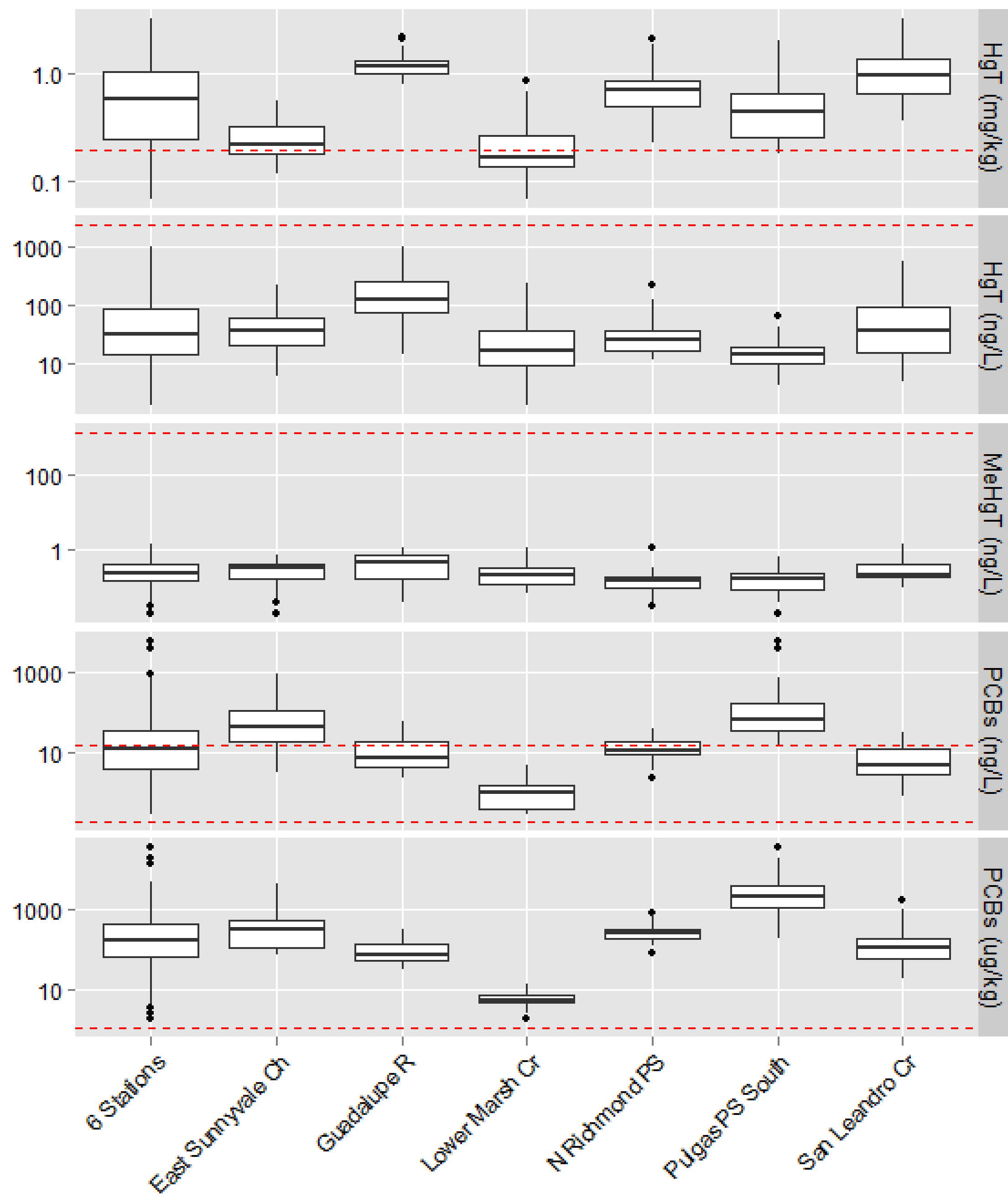

Figure 2. Summary boxplots of data collected across the six sampling stations for select analytes. Boxplots for other analytes available in Attachment 3. Dashed red lines denote TMDL target, water quality criteria or other benchmark noted in Table 8.

below the criterion, with Pulgas Pump Station – South and East Sunnyvale Channel having virtually all sampled concentrations exceed the criterion. The San Francisco Bay TMDL criterion for PCBs in suspended sediment is exceeded by the particle ratios of all samples collected in the study. The exceedance in March Creek is particularly notable since this represents the "cleanest" least urban watershed measured in the Bay Area so far; this observation suggests the intractable nature of reducing PCBs loads to a level that will meet the TMDL objectives.

With regard to the tier I priority analytes, it is also worth noting that phosphorus concentrations in most of the six watersheds appear greater than elsewhere in the world under similar land use scenarios, perhaps attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005). For example, Dillon and Kirchner (1975) found that watersheds of differing geology under the same land use could exhibit loads differing by an order of magnitude. Bay Area watersheds with geological sources of phosphorus such as appetite minerals may naturally release greater amounts of phosphorus. All total phosphorous concentrations sampled exceed the EPA Nutrient criteria for Level III Eco-Region 6 (Attachment 3 Figure 21).

Selenium and PBDE concentrations, two analytes being collected at a lesser frequency in this study (intended only for characterization) are particularly notable. In the Guadalupe River, mean selenium concentrations were 2 to 6-fold greater than the other five locations (Attachment 3 Figure 22). Elevated groundwater concentrations have been observed in Santa Clara County previously (Anderson, 1998). Across all six sites, Se concentrations averaged 0.6 µg/L. If these concentrations are representative and combined with average annual flow entering the Bay from the nine-county Bay Area (1.5 km^3 based on the RWSM: Lent et al., 2012), the total average annual Se load would be estimated to be 900 kg. Although this is less than the estimated average annual load entering the Bay from the Central Valley Rivers (16,000 kg/yr; David et al., in press), it is still a large component of the Se mass balance for the Bay. Maximum PBDE concentrations in North Richmond Pump Station were 33 to 60-fold greater than the PBDE maxima observed in the five other locations of this current study (Attachment 3 Figure 21). These are the highest PBDE concentrations measured in Bay Area stormwater to-date (see section 8.2 for details). Additional investigation into the source-release processes of PBDE that are specific to Richmond, and lacking in the other watersheds, would be needed to better understand this result.

Relative to the USEPA Office of Pesticide Programs' Aquatic Life Benchmarks, the median concentration of permethrin across all six sites exceeds the acute criterion for invertebrates with East Sunnyvale Channel exceeding the criteria in nearly every sample (Attachment 3 Figure 20). Lower Marsh Creek had some exceedances for bifenthrin and San Leandro Creek for cyhalothrin. There were no exceedances at any station for delta/tralomethrin (Attachment 3 Figure 21), carbaryl or fipronil. However, as noted previously, fipronil samples were filtered and therefore have a low bias. Additionally, fipronil degradates were not analyzed for this study yet it has been more recently identified that fipronil degradates can be as or more toxic than the parent chemical.

In summary, concentration sampling during the three water years at the six locations has in part confirmed previously known or suspected high leverage watersheds (i.e. mercury in Guadalupe, PCBs in East Sunnyvale Channel and Pulgas Pump Station - South). Concentration results have also raised some questions about certain pollutants in other watersheds (e.g. upper versus lower watershed Hg concentrations in San Leandro Creek, PBDE concentrations in North Richmond Pump Station). More sampling under a broader range of storm events (early season and first flush, larger storms during the mid-season and later season storms) would improve characterization of pollutants in those watersheds and increase confidence in the relative magnitude between watersheds and average annual loads estimates (baseline concentrations) that might form the basis for assessing trends (MQ3) at some future time. Although not the subject of this report, the RMP has provided funding to support the development of a POC loadings synthesis document (McKee et al. 2015) and a trends strategy document (slated for preparation in spring 2016). A more thorough evaluation of existing data as a baseline for the trends management questions will be completed through those efforts.

4.1 Loads of pollutants of concern computed for each sampling location

One of the primary goals of this project and a key management question of the Small Tributary Loading Strategy was to estimate the annual loads of POCs from tributaries to the Bay (MQ2). In particular, large loads of POCs entering sensitive Bay margins are likely to have a disproportionate impact on beneficial uses (Greenfield and Allen, 2013). As described in the climatic section (5.2), given that the relationship between climate (manifested as either rainfall or resulting discharge) and watershed loads follows a power function, estimates of long-term average loads for a given watershed are highly influenced by samples collected during wetter than average conditions and rare high magnitude storm events. Comparing loads estimates between the sites was confounded by relatively small sample datasets collected during climatically dry years. However, based on data collected, average annual loads estimates for each sampling location have now been computed. Accepting these caveats, the following observations are made on the total wet season loads estimates at the six locations.

The magnitude of the total loads between watersheds is largely driven by drainage area of each watershed. In terms of total wet season loads from each of the six watersheds, the largest watershed sampled is the Guadalupe River, which also has the largest load for every pollutant estimated in this study. Conversely, Pulgas Pump Station - South is the smallest watershed in the study and has the lowest total wet season load for all analytes except PCBs. As another example, methylmercury in San Leandro Creek (8.9 km2) and Guadalupe River (236 km^2) have similar concentrations but Guadalupe River discharges more than 10x the total mass of methylmercury given the much greater overall discharge of runoff volume and sediments. There is one significant exception. As mentioned, Pulgas Pump Station - South exports a disproportionately large PCB load, greater than Lower Marsh Creek (up to 10-fold larger), North Richmond Pump Station (up to 3-fold larger), and San Leandro Creek (up to 6-fold larger) (Table 9).

Comparison of total wet season loads between water years at the sites highlights how loads estimates can be highly variable even during three drier than average years. Additionally, the size and intensity of the storm events in the different regions where the sampling sites were located greatly impacted the load variation from year-to-year and between sampling locations. For example, SS loads in Guadalupe River and San Leandro Creek were approximately 1-to-2-fold greater in WY 2013 than in WY 2014, whereas SS loads were 12- and 8-fold larger in WY 2013 relative to WY2012 in Lower Marsh Creek and East Sunnyvale Channel, where the late November and December 2012 (WY2013) storms were

Table 9. Loads of pollutants of concern during the sampling years at each sampling location.

Site	Water Year	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)	Loads Confidence	Loads Quality
Marsh Creek[a]	2012	1.61	233	11,380	1.34	64.0	0.262	956	175	578	Moderate (PCBs); Low (Hg)	Lack of sample data during storms that cause runoff and sediment transport through the upper watershed reservoir and data during a wet year.
	2013	5.82	2,703	39,500	16.0	408	2.78	3,474	666	4,212		
	2014	1.34	202	9,257	1.20	30.7	0.217	786	148	479		
North Richmond Pump Station[b]	2012	-	-	-	-	-	-	-	-	-	Moderate	Lack of data during wet year.
	2013	0.795	35.7	6,353	8.14	16.0	0.200	761	161	215		
	2014	0.499	20.4	6,197	4.76	15.8	0.117	478	101	186		
San Leandro Creek[c]	2012	7.30	232	40,483	16.4	221	1.57	1,973	571	1,404	Low	Lack of a robust discharge rating curve for higher flows; lack of data during reservoir release and during a wet year.
	2013	7.21	230	52,274	15.0	213	1.58	2,801	674	1,334		
	2014	0.243	27.9	1,840	1.93	25.4	2.89	97.1	23.4	70.6		
Guadalupe River	2012	25.8	2,106[d]	154,379	123	2,039	6.13	20,879	2,498	6,023	High (PCBs); Low (Hg)	Lack of long duration and high intensity storms sampled for Hg release from upper watershed. Confidence in PCB data supported by previous studies.
	2013	35.5	4,464[d]	238,208	309	5,476	13.6	25,775	3,771	10,829		
	2014	16.75	1,668[d]	106,141	97.2	1,519	4.29	13,182	1,723	4,172		
East Sunnyvale Channel[e]	2012	1.31	55.0	8,227	50.6	25.9	0.404	335	139	386	Moderate	Lack of data during wet year. High variability in PCB concentrations between storm events.
	2013	1.51	430	8,685	81.9	81.9	2.64	369	159	628		
	2014	1.01	90.4	12,040	76.8	27.5	1.13	336	135	347		
Pulgas Pump Station – South[f]	2012	-	-	-	-	-	-	-	-	-	Low	A lower quality (FWMC) approach applied to loads calculations. Lack of data during a wet year. High variability in PCB concentrations between storm events.
	2013	0.165	10.9	1,539	21.8	3.07	0.0291	41.1	12.8	33.0		
	2014	0.08	5.31	764	11.8	1.48	0.0141	20.1	6.31	16.1		

[a] Marsh Creek wet season loads are reported for the period of record 12/01/11 – 4/26/12, 10/19/12 – 4/18/13 and 11/06/13 – 4/30/14.

[b] North Richmond Pump Station wet season loads are reported for the period of record 11/01/12 – 4/30/13 and 10/16/13 – 4/30/14.

[c] San Leandro Creek wet season loads are reported for the period of record 12/01/11 – 4/30/12, 11/01/12 – 4/18/13 and 11/01/13 – 4/30/14.

[d] SS loads for Guadalupe River were computed by the USGS.

[e] East Sunnyvale Channel wet season loads are reported for the period of record 12/01/11 – 4/30/12, 10/01/12 – 4/30/13 and 10/01/13 – 4/30/14.

[f] Pulgas Pump Station - South loads are estimates provided for the entire wet seasons (10/01/12 – 4/30/13 and 10/01/13 – 4/30/14) however monitoring only occurred during the period 12/17/2012 – 3/15/2012 and 10/22/13 – 4/30/14. Monthly loads for the non-monitored period were extrapolated using regression equations developed for the monthly rainfall and corresponding monthly (or partial month) contaminant load.

comparatively larger events. Even when normalized to total discharge (in other words, the flow-weighted mean concentration [FWMC]), East Sunnyvale Channel transported 7-fold as much sediment in WY 2013 than WY 2012, whereas the FWMC of suspended sediment in San Leandro Creek was the same in WYs 2012 and 2013. The relationship between FWMC and discharge (either at the annual or individual flood scale) can be used as an indicator of when enough data have been collected to characterize the site adequately to answer our management questions. FWMC should continue to increase relative to storm magnitude until watershed sources are exhausted; locations and analytes that reach that maximum will have sufficient data to compute reliable long term average annual loads. With the data currently in hand, attempts to estimate average annual loads will be biased low.

In light of these climatic considerations as well as the known data quality considerations and challenges at each of the sampling locations, the two far-right columns in Table 9 note the remaining level of confidence in the annual loads estimates as well as the main issues at each site which warrant the confidence level rating. Any future sampling at each of these locations should seek to alleviate these issues and to raise the quality of the data in relation to answering management questions.

5.5. Comparison of regression slopes and normalized loads estimates between watersheds

One of our key activities in relation to the Small Tributary Loading Strategy is improving our understanding of which Bay tributaries (including stormwater conveyances) contribute most to Bay impairment from pollutants of concern (MQ1) and therefore potentially represent watersheds where management actions should be implemented to have the greatest beneficial impact (MQ4). Multiple factors influence the treatability of pollutant loads in relation to impacts to San Francisco Bay. Conceptually, a large load of pollutant transported on a relatively small mass of sediment is more treatable than less polluted sediment. Therefore, the graphical function between pollutants and either sediment concentration or turbidity provides a first order mechanism for ranking relative treatability of watersheds (Figure 3A). This method is valid for pollutants that are dominantly transported in a particulate form (total mercury and the sum of PCBs are good examples but pyrethroid pesticides and PBDEs may also be considered in this group) and when there is relatively little variation in the particle ratios between water years or storms or at least less variation than seen between watersheds. Note data presented at the October 2013 SPLWG meeting demonstrated that this assumption is sometimes violated.

These issues accepted, based on the ratios between turbidity and Hg, runoff derived from less urbanized upper portions of San Leandro Creek watershed and runoff from the Guadalupe River watershed exhibit the greatest particle ratios for HgT (Figure 3). East Sunnyvale Channel, Marsh Creek and Pulgas Pump Station - South appear to have relatively low particle ratios for HgT, although, Marsh Creek has not been observed under wet conditions when the possibility of mercury release from historic mining sources exists. The relative nature of these rankings has not changed in relation to the previous reports (McKee et al., 2013; Gilbreath et al., 2014).

In contrast, for the sum of PCBs, Pulgas Pump Station - South and East Sunnyvale Channel exhibit the highest particle ratios among these six watersheds, with urban sourced runoff from Guadalupe River

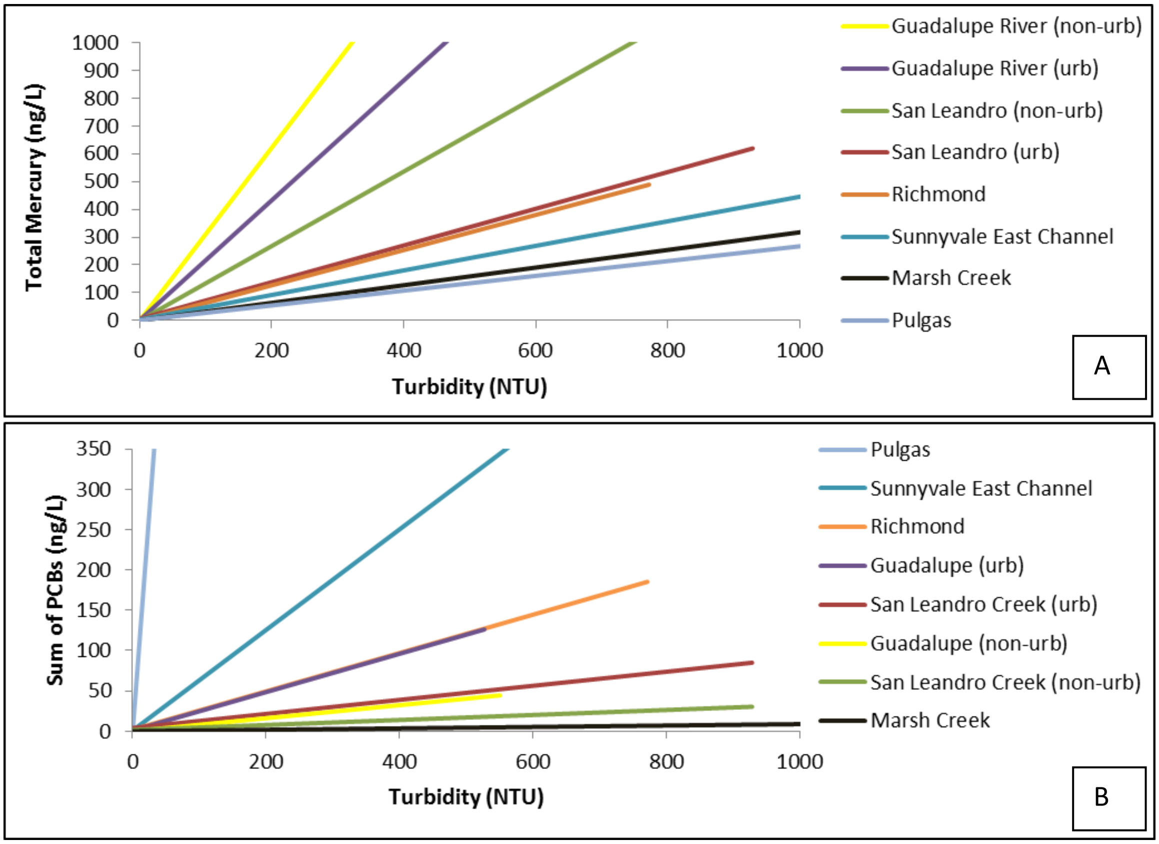

Figure 3. Comparison of regression slopes between watersheds based on data collected during sampling for A) Total Mercury and B) PCBs. Turbidity range shown on graphs represents minimum and maximum turbidities for entire sampling period

and North Richmond Pump Station ranked 3rd and 4th as indicated by the turbidity-PCB graphical relation (Figure 3B). Marsh Creek exhibits very low particle ratios for PCBs, an observation that is unlikely to change with additional samples given the likelihood of relatively few sources and low variability of release-transport processes. Unlike for Hg, new data collected during WYs 2013 and 2014 alters the relative PCB rankings based on this graphical analysis providing an example of the influence of either low sample numbers or the random nature of sample capture on the resulting interpretation of particle ratios (as discussed in the October 2013 SPLWG meeting). Given the relatively wide confidence intervals around these lines (not shown) and the collection during dry years, the relative nature of these regression equation rankings may change if there are any future samples completed.

Another influence on potential treatability is the size of the watershed. Conceptually, a large load that is transported from a smaller watershed - and therefore in association with a smaller volume of water - is more manageable. Efforts to manage flows from the North Richmond Pump Station watershed exemplify this type of opportunity. Thus, area normalized loads (yields) provide another useful mechanism for first order ranking of watersheds (Table 10) in relation to ease of management. This method is more highly subject to climatic variation than the turbidity function/particle ratio method for ranking and therefore was done on climatically averaged loads. Despite these challenges, in a general

Table 10. Climatically averaged area normalized loads (yields) ranked in relation to PCBs based on free flowing areas downstream from reservoirs (See Table 1 for areas used in the computations).

	Unit runoff (m)	SS (t/km^2)	TOC (mg/m^2)	PCBs ($\mu g/m^2$)	HgT ($\mu g/m^2$)	MeHgT ($\mu g/m^2$)	NO3 (mg/m^2)	PO4 (mg/m^2)	Total P (mg/m^2)
Marsh Creek	0.11	68.0	770	0.40	12	0.036	67	13	65
North Richmond Pump Station	0.56	21	5300	4.7	21	0.15	540	110	170
San Leandro Creek	0.95	66	6000	3.4	55	0.26	320	82	220
Guadalupe River	0.24	47	1800	5.7	380	0.14	170	28	120
East Sunnyvale Channel	0.17	25	1200	9.0	5.6	0.12	45	19	60
Pulgas Pump Station - South	0.59	40	5700	83	11	0.11	150	47	120

sense, the relative rankings for PCBs exhibit a similar ranking to the particle ratio method; Pulgas Pump Station - South watershed ranked highest and Marsh Creek watershed ranked lowest. However the relative ranking of the other watersheds is not similar. In the case of mercury, Guadalupe River and San Leandro Creek exhibit the highest currently estimated yields corroborating the evidence from the particle ratio method. Similar to PCBs, the relative ranking of the other four watersheds is not similar to the particle ratio method. Given all our observations were during dry years, it is difficult to know the certainty of the climatically averaged yields. For example, the relative rankings for suspended sediment loads normalized by unit area would likely change substantially with the addition of data from a water year that exceeds the climatic normal for each watershed; total phosphorus unit loads would also respond in a similar manner. For pollutants such as PCBs and HgT that are found in specific source areas such as industrial and mining areas (Hg only), release processes will likely be influenced by both climatic factors and sediment transport off impervious surfaces; also factors that are not likely well captured by the sampling that has occurred under dry conditions.

5.6. How to access the sampling data

Data for this project is housed by and publicly available for download at the California Environmental Data Exchange Network (CEDEN). CEDEN can be accessed at the following url: http://www.ceden.org/ Follow these directions to find the data for this study:

1) Once at the above url, click on: “Find Data” towards left of screen.
2) Everything can be left as the default except the following:
    a. Approximately midway down screen, click “Select Programs” and highlight by clicking “SF Bay STLS Monitoring”. Then click “Done”.
    b. Down one tab, click “Select Projects” and highlight “STLS Monitoring WY 2012”, “STLS Monitoring WY 2013”, and “STLS Monitoring WY 2014” by holding the shift key while you click on each. Then click “Done”.

3) Scroll down to bottom and click "Retrieve Data".

This enables you to retrieve all of the data but use of the other search tabs will allow you to narrow the results to a more specific selection.

6. Conclusions and next steps

6.1. Current and future uses of the data

The monitoring program implemented during the study was designed primarily to improve estimates of watershed-specific and regional loads to the Bay (MQ2) and secondly, to provide baseline data to support evaluation of trends towards concentration or loads reductions in the future (conceptually one or two decades hence) (MQ3) (see introduction section) in compliance with MRP provision C.8.e. (SFRWRCB, 2009). Multiple metrics have been developed and presented in this report to support these management questions:

- Pollutant loads: Pollutant loading estimates can help measure relative delivery of pollutants to sensitive Bay margin habitats and support calibration and verification of the Regional Watershed Spreadsheet Model and resulting regional scale loading estimates.
- Flow Weighted Mean Concentrations: FWMC can help to identify when sufficient data has been collected to adequately characterize watershed processes in relation to a specific pollutant in the context of management questions.
- Sediment-pollutant particle ratios: Particle ratios can help identify relative watershed pollution levels on a particle basis and relates to treatment potential.
- Pollutant area yields: Pollutant yields can help identify pollutant sources and relates to treatment potential.
- Correlation of pollutants: Finding co-related pollutants helps identify those watersheds with multiple sources and provides additional cost/benefit for management actions.

As discussed briefly in the introduction (section 1), as management effort focuses more and more on locating high leverage watersheds and patches within watersheds, the monitoring (and modeling) design is evolving.

6.2. What data gaps remain at current loads stations?

With regard to addressing the main management endpoints (single watershed and regional watershed loads and baseline data for trends) that influenced the monitoring design recommended by Melwani et al 2010 and described in each iteration of the MYP (BASMAA, 2011; BASMAA, 2012; BASMAA, 2013), an important question that managers are asking is how to determine when sufficient data have been collected. Several sub-questions are important when trying to make this determination. Are the data representative of climatic variability; have storms and years been sampled well enough relative to expected climatic variation? Are the data representative of the source-release-transport processes of the pollutant of interest? In reality, these factors tend to juxtapose and after three years of monitoring during relatively dry climatic conditions, some data gaps remain for each of the monitoring locations.

- Marsh Creek watershed has been sampled for three WYs. Continuous turbidity data were rated excellent at Lower Marsh Creek. Ample lower watershed stormwater runoff data are now available at Lower Marsh Creek, but this site is lacking information on high intensity upper watershed rain events where sediment mobilization from the historic mercury mining area could occur. Any future sampling would ideally be focused on Hg and for storms of greater intensity preferably when spillage is occurring from the upstream reservoir. No further PCB data are recommended. The sampling design to achieve these goals could be revisited with the objective of increased cost efficiency for data gathering to support remaining unanswered management questions.
- North Richmond Pump Station watershed has been sampled for two WYs (although data exist from a previous study [Hunt et al., 2012]). Additional data in relation to early season (seasonal 1st flush or early season storms) would help improve estimates of loads that could be averted from diversion of early season storms to wastewater treatment. Further data collection in relation to high concentrations of PBDEs would increase our understanding of PBDE source(s) in this watershed.
- San Leandro Creek watershed has been sampled for three WYs. San Leandro Creek received poor ratings on the quality of discharge information and completeness of turbidity data. The largest weakness is the scarcity of velocity measurements to adequately describe the stage-discharge rating curve for stages >2 feet and generate a continuous flow record. Additional velocity measurements are necessary to increase the accuracy and precision of discharge data for the site and support the computation of loads. There is currently no information on pollutant concentrations during reservoir releases, yet volumetrically, reservoir releases during WYs 2012 and 2013 were proportionally large but may have been atypical. Sample collection during release would help elucidate pollutant load contributions from the reservoir. Data collection during more intense rainstorms are also desirable for this site given the complex sources of PCBs and mercury in the watershed and the existence of areas of less intense land use and open space lending to likely relatively high inter-annual variability of water and sediment production.
- Guadalupe River watershed has been sampled at the Hwy 101 location during nine water years (WY 2003-2006, 2010-2014) to-date, but data are still lacking to adequately describe high intensity upper watershed rain events when mercury may still be released from sources in relation to historic mining activities. This type of information could help estimate the upper range of mercury loads from the mercury mining district and continue to help focus management attention. Further data collection in Guadalupe River watershed should focus on Hg sampling during high intensity storms. Further sampling of relatively frequent smaller runoff events is unnecessary and transport processes for PCBs are well supported by currently available data. The current sampling design is not cost-effective for gathering improved information to support management decisions in this watershed.
- East Sunnyvale Channel initially received poor quality data ratings for turbidity but this improved substantially in WYs 2013 and 2014. However, more storm event POC data are needed for establishing higher confidence in particle ratios, pollutant loads, FWMCs, and yields. A PCB source was apparently mobilized during the February 28, 2014 storm which had very high PCB

concentrations, and this source seemed to continue to flush through the system in subsequent events. Because of this, our PCB regression with turbidity is not strong, creating uncertainty around the accuracy of the total PCB load estimate (e.g. what PCB sources might have moved through the system when we were not sampling?). Further data are needed in this watershed to better understand source-release-transport processes for PCBs.

- Based on the current review of the data, Pulgas Pump Station - South received a poor data quality rating for turbidity. Monitoring at this site was complicated by the logistical limitations of monitoring in a highly dynamic storm drain system. The challenging logistics of this site led to delays in the initiation of monitoring in WY 2013 as BASMAA/KLI worked to establish a monitoring plan and functional instrumentation configuration (e.g., during WY 2013, turbidity data were only collected during three of the seven wet season months due to these challenges). In addition, because this site was located within a storm drain and vault adjacent to a pump station, the periodic operation of the pumps likely contributed to turbidity spikes and generally noisy nature of the data. Following review of WY 2014 observations, it was decided to reject the whole turbidity data set from this site. Although not feasible under the scope of this project, BASMAA has suggested they may undertake further review of this dataset, including application of smoothing functions to better fit the pollutant data to the turbidity record and potentially improve the usability of these data. KLI collected a robust manual turbidity sample set in combination with the pollutant sampling. Although they did not accurately record the times of this sample collection and therefore a relationship between manual turbidity and the sensor turbidity record for discrete times cannot be developed, a relationship between manual collection and smoothed sensor data (e.g. smoothed over 15-30 minutes) may be possible. This could then validate the data quality of the smoothed turbidity data, and allow future use of these data for the development of turbidity-pollutant regressions. However, because of the dynamic nature of this system (e.g. the sensor record showed changes >500 NTU in a 15 minute period), the likelihood of forming acceptable regressions between pollutant data and smoothed turbidity data seems low. More importantly, the cyclical spiking of the turbidity record suggests resuspension of settled sediments during pump outs. If the turbidity sensor was measuring resuspension of sediment in the vault, the record then includes sediment that was twice-measured (e.g. turbidity caused by sediment when it initially entered the vault, as well as when it was resuspended). Therefore, the continuous turbidity record likely does not accurately represent the turbidity within the system, and consequently an accurate, continuous record for any pollutant likely cannot be established using the turbidity surrogate regression method even in the event that a pollutant-turbidity regression could be developed through smoothing.

  The sampling program began at this location (and North Richmond Pump Station) in WY 2013 as compared to WY 2012 at the other sites, and so despite being one of the most logistically challenging sites to set up for monitoring, BASMAA/KLI also had the least amount of time to execute it (arguably North Richmond Pump Station was also logistically challenging but SFEI had already completed two years of sampling at this location for another project, during which some of the instrumentation set-up challenges had been worked through). Due to both the delay in monitoring initiation at Pulgas combined with the very low rainfall in WY 2013, only a single storm was monitored and therefore very little data was available from WY 2013 in which to assess these issues. In short, although this has been a three-year project, this is really the first year that a substantial dataset has been available to evaluate for the Pulgas Ck Pump Station site. On the positive side, there are nearly two full wet seasons of flow data as well as seven storms worth of pollutant data, including the highest PCB concentrations observed to-date in the Bay Area. Despite challenges with the continuous turbidity record, these other data are valuable and less robust estimates of load are possible based on the FWMC approach. Additionally, because KLI also collected manual turbidity samples during pollutant sample collection, the pollutant data could potentially still be used to estimate loads using turbidity surrogate regression if a high quality relationship between the manually collected turbidity record and a continuous record could be established. Now that the monitoring challenges for this site are better understood, additional effort to improve the continuous turbidity monitoring at this location would be desirable to increase confidence in particle ratios, pollutant loads, FWMCs, and yields.

6.3. Next Steps

Recent discussions between BASMAA and the Region 2 Regional Water Quality Control Board in relation to reissuing the MRP (and discussion at the October 2013 and May 2014 SPLWG meetings) have highlighted the increasing focus towards finding watersheds and land areas within watersheds for management focus (MQ4). The monitoring design described in this report is not appropriate for this increasing management focus. There are various alternative monitoring designs that are more cost-effective for addressing the increasing focus in the second MRP permit term towards finding watersheds and land areas within watersheds for management attention while still supporting the other STLS management questions in a programmatic manner. The challenge for the STLS and SPWLG is finding the right balance between the different alternatives within budget constraints. Sampling during WY 2015 is using the following reconnaissance characterization design:

- Collaboration with stormwater Countywide programs to identify locations with possible PCB and/or mercury sources (based on a GIS based analysis)
- Focused sampling in older industrial drainages (some of which are tidally influenced)
- Composite sampling: 1 composite per storm/per analyte for PCB, total mercury, total metals, SSC, grain size, TOC/DOC; 5-15 aliquots per composite sample
- Pilot testing passive sediment samplers

The advantage of the reconnaissance sampling design is flexibility and given recent advances on the development of the RWSM (SFEI in preparation) have indicated the value of the data collected previously using the reconnaissance design (McKee et al., 2012), it seems likely that the reconnaissance design may end up being the most cost-effective going forward over the next three or more years. Data and information gathered over the last 10+ years guided by the SPLWG and STLS will continue to help guide the development of a cost effective monitoring design to adapt to changing management needs.

7. References

Amweg, E., D.P. Weston and N.M. Ureda. Use and toxicity of pyrethroid pesticides in the central valley, California, USA. Environmental Toxicology and Chemistry. vol. 24, no. 4, pp. 966–972.

Anderson B., Hunt, J., Markiewicz, D., Larsen, K. 2010. Toxicity in California Waters. Surface Water Ambient Monitoring Program. California State Water Resources Control Board. Sacramento, CA.

Anderson, D.W., 1998. Natural levels of nickel, selenium, and arsenic in the South San Francisco Bay area. Report prepared for the City of San Jose, Environmental Services Department by the Institute for Research in Environmental Engineering and Science, San Jose, Ca. 15pp.

Balogh, S. J., Y. Huang, H. J. Offerman, M. L. Meyer, and D. K. Johnson. 2002. Episodes of Elevated Methylmercury Concentrations in Prairie Streams. Environmental Science and Technology **36**:1665 - 1670.

Barringer, J. L., M. L. Riskin, Z. Szabo, P. A. Reilly, R. Rosman, J. L. Bonin, J. M. Fischer, and H. A. Heckathorn. 2010. Mercury and Methylmercury Dynamics in a Coastal Plain Watershed, New Jersey, USA. Water Air and Soil Pollution **212**:251-273.

Bartley, R., W. J. Speirs, T. W. Ellis, and D. K. Waters. 2012. A review of sediment and nutrient concentration data from Australia for use in catchment water quality models. Marine Pollution Bulletin **65**:101-116.

BASMAA, 2011. Small Tributaries Loading Strategy Multi-Year Plan (MYP) Version 2011. A document developed collaboratively by the Small Tributaries Loading Strategy Work Group of the Regional Monitoring Program for Water Quality in the San Francisco Estuary (RMP): Lester McKee, Alicia Gilbreath, Ben Greenfield, Jennifer Hunt, Michelle Lent, Aroon Melwani (SFEI), Arleen Feng (ACCWP) and Chris Sommers (EOA/SCVURPPP) for BASMAA, and Richard Looker and Tom Mumley (SFRWQCB). Submitted to the Water Board, September 2011, in support of compliance with the Municipal Regional Stormwater Permit, provision C.8.e. http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/stormwater/MRP/2011_AR/BASMAA/B2_2010-11_MRP_AR.pdf

BASMAA, 2012. Small Tributaries Loading Strategy Multi-Year Plan (MYP) Version 2012A. A document developed collaboratively by the Small Tributaries Loading Strategy Work Group of the Regional Monitoring Program for Water Quality in the San Francisco Estuary (RMP): Lester McKee, Alicia Gilbreath, Ben Greenfield, Jennifer Hunt, Michelle Lent, Aroon Melwani (SFEI), Arleen Feng (ACCWP) and Chris Sommers (EOA/SCVURPPP) for BASMAA, and Richard Looker and Tom Mumley (SFRWQCB). Submitted to the Water Board, September 2011, in support of compliance with the Municipal Regional Stormwater Permit, provision C.8.e. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/stormwater/MRP/2012_AR/BASMAA/BASMAA_2011-12_MRP_AR_POC_APPENDIX_B4.pdf

Birgand, F., C. Faucheux, G. Gruau, B. Augeard, F. Moatar, and P. Bordenave. 2010. Uncertainties in assessing annual ntirate loads and concentration indicators: Part 1. Impact of sampling frequency and load estimation algorithms. Transactions of the Asabe **53**:437-446.

Bradford, G. R., A. C. Chang, A. L. Page, D. Balhtar, J. A. Frampton, and H. Wright. 1996. Background Concentrations of Trace and Major Elements in California Soils Kearney Foundation of Soil Science.

Bradley, P. M., D. A. Burns, K. Riva-Murray, M. E. Brigham, D. T. Button, L. C. Chasar, M. Marvin-DiPasquale, M. A. Lowery, and C. A. Journey. 2011. Spatial and Seasonal Variability of Dissolved Methylmercury in Two Stream Basins in the Eastern United States. Environmental Science & Technology 45:2048-2055.

Chalmers, A. T., D. P. Krabbenhoft, P. C. Van Metre, and M. A. Nilles. 2014. Effects of urbanization on mercury deposition and accumulation in New England. Environmental Pollution 192:104-112.

Chow, V. T. 1959. Open-Channel Hydraulics. McGraw-Hill, Inc. 680pp.

Conaway, C. H., E. B. Watson, J. R. Flanders, and A. R. Flegal. 2003. Mercury deposition in a tidal marsh of south San Francisco Bay downstream of the historic New Almaden mining district, California. Marine Chemistry 90:175-184.

David, N., Gluchowski, D.C, Leatherbarrow, J.E, Yee, D., and McKee, L.J, (in press). Evaluation of loads of mercury, selenium, PCBs, PAHs, PBDEs, dioxins, and organochlorine pesticides from the Sacramento-San Joaquin River Delta to San Francisco Bay. Water Environment Research, 87, xx-xx.

Davis, J.A., Hetzel, F., Oram, J.J., and McKee, L.J., 2007. Polychlorinated biphenyls (PCBs) in San Francisco Bay. Environmental Research 105, 67-86. http://www.sciencedirect.com/science/article/pii/S0013935107000400

Davis, J.A., Yee, D., Grenier, L., McKee, L.J., Greenfield, B.A., Looker, R., Austin, C., Marvin-DePasquale, M., Brodberg, R., and Blum, J., 2012. Reducing methylmercury accumulation in the food webs of San Francisco Bay and its local watersheds. Environmental Research 119, 3-26. http://www.sciencedirect.com/science/article/pii/S001393511200285

Dillon, P. J., and Kirchner, W.B. 1975. The effects of geology and land use on the export of phosphorus from watersheds. Water Research 9:135-148.

Domagalski, J. 2001. Mercury and methylmercury in water and sediment of the Sacramento River Basin, California. Applied Geochemistry 16:1677.

Eckley, C. S. and B. Branfireun. 2008. Mercury mobilization in urban stormwater runoff. Science of the Total Environment 403:164-177.

Ensminger, M.P., Budd, R., Kelley, K.C., and Goh, K.S., 2012. Pesticide occurrence and aquatic benchmark exceedances in urban surface waters and sediments in three urban areas of California, USA, 2008-

2011. Environmental Monitoring and Assessment http://www.springerlink.com/content/g11r274187122410/

Fitzgerald, W. F., D. R. Engstrom, R. P. Mason, and E. A. Nater. 1998. The case for atmospheric mercury contamination in remote areas. Environmental Science and Technology 32:1-7.

Foster GD, Lippa KA, Miller CV. Seasonal concentrations of organic contaminants at the fall line of the Susquehanna River Basin and estimated fluxes to Northern Chesapeake Bay, USA. Environ Toxicol Chem 2000a; 19(4):992-1001.

Foster GD, Roberts EC Jr., Gruessner B, Velinsky DJ. Hydrogeochemistry and transport of organic contaminants in an urban watershed of Chesapeake Bay USA. App Geochem 2000b; 15:901-15.

Gilbreath, A.N., Gluchowski, D.C., Wu, J., Hunt, J.A., and McKee, L.J., 2014. Pollutants of concern (POC) loads monitoring data progress report, water year (WYs) 2012 and 2013. 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). Contribution No. 708. San Francisco Estuary Institute, Richmond, California. http://www.sfei.org/sites/default/files/Final_WY%202013_POC%20loads%20progress%20report_24%20February%202014_web%20posted.pdf

Gilbreath, A., Yee, D., McKee, L.J., 2012. Concentrations and loads of trace contaminants in a small urban tributary, San Francisco Bay, California. A Technical Report of the Sources Pathways and Loading Work Group of the Regional Monitoring Program for Water Quality: Contribution No. 650. San Francisco Estuary Institute, Richmond, California. 40pp. http://www.sfei.org/sites/default/files/Z4LA_Final_2012May15.pdf

Gomez-Gutierrez, A. I., E. Jover, L. Bodineau, J. Albaiges, and J. M. Bayona. 2006. Organic contaminant loads into the Western Mediterranean Sea: Estimate of Ebro River inputs. Chemosphere 65:224-236.

Greenfield, B.K., and Allen, R.M., 2013. Polychlorinated biphenyl spatial patterns in San Francisco Bay forage fish. Chemosphere 90, 1693–1703.

Guan YF, Wang J, Ni HG, Luo XJ, Mai BX, Zeng EY., 2007. Riverine inputs of polybrominated diphenyl ethers from the Pearl River Delta (China) to the coastal ocean. Environ Sci Tech 41(17):6007-13.

Henjum, M. B., R. M. Hozalski, C. R. Wennen, W. A. Arnold, and P. J. Novak. 2010. Correlations between in situ sensor measurements and trace organic pollutants in urban streams. Journal of Environmental Monitoring 12:225-233.

Howell NL, Lakshmanan D, Rifai HS, Koenig L. PCB dry and wet weather concentration and load comparisons in Houston-area urban channels. Sci Tot Environ 2011; 409: 1867-1888.

Hudak, P. F. and K. E. Banks. 2006. Compositions of first flush and composite storm water runoff in small urban and rural watersheds, north-central Texas. Urban Water Journal 3:43-49.

Hunt, J., Gluchowski, D., Gilbreath, A., and McKee, L.J., 2012. Pollutant Monitoring in the North Richmond Pump Station: A Pilot Study for Potential Dry Flow and Seasonal First Flush Diversion for Wastewater Treatment. A report for the Contra Costa County Watershed Program. Funded by a grant from the US Environmental Protection Agency, administered by the San Francisco Estuary Project. San Francisco Estuary Institute, Richmond, CA. http://www.sfei.org/sites/default/files/NorthRichmondPumpStation_Final_19112012_ToCCCWP.pdf

Hurley, J. P., J. M. Benoit, C. L. Babiarz, M. M. Shafer, A. W. Andren, J. R. Sullivan, R. Hammond, and D. A. Webb. 1995. Influences of watershed characteristics on mercury levels in Wisconsin rivers. Environmental Science and Technology 29:1867-1875.

Hwang, H.-M. and G. D. Foster. 2008. Polychlorinated biphenyls in stormwater runoff entering the tidal Anacostia River, Washington, DC, through small urban catchments and combined sewer outfalls. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering 43:567-575.

Inman, D. L. and S. A. Jenkins. 1999. Climate change and the episodicity of sediment flux of small California River. The Journal of Geology 107:251-270.

Keppeler, E. T. 2012. Sediment production in a coastal watershed: legacy, land use, recovery, and rehabilitation. Pages 69-77 in R. B. Standiford, T. J. Weller, D. D. Piirto, and J. D. Stuart, editors. Proceedings of coast redwood forests in a changing California: A symposium for scientists and managers, Albany, CA.

Kocman, D., M. Horvat, N. Pirrone, and S. Cinnirella. 2013. Contribution of contaminated sites to the global mercury budget. Environmental Research 125:160-170.

Kronvang, B. and A. J. Bruhn. 1996. Choice of sampling strategy and estimation method for calculating nitrogen and phosphorus transport in small lowland streams. Hydrological Processes 10:1483-1501.

Lamborg, C. H., W. F. Fitzgerald, A. W. H. Damman, J. M. Benoit, P. H. Balcom, and D. R. Engstrom. 2002. Modern and historic atmospheric mercury fluxes in both hemispheres: Global and regional mercury cycling implications. Global Biogeochemical Cycles 16:1104.

Lawson, N. M., R. P. Mason, and J.-M. Laporte. 2001. The fate and transport of mercury, methylmercury, and other trace metals in chesapeake bay tributaries. Water Research 35:501.

LeBlanc, L. A. and K. M. Kuivila. 2008. Occurrence, distribution and transport of pesticides into the Salton Sea Basin, California, 2001-2002. Hydrobiologia 604:151-172.

Lent, M.A. and McKee, L.J., 2011. Development of regional suspended sediment and pollutant load estimates for San Francisco Bay Area tributaries using the regional watershed spreadsheet model (RWSM): Year 1 progress report. A technical report for the Regional Monitoring Program for Water Quality, Small Tributaries Loading Strategy (STLS). Contribution No. 666. San Francisco Estuary

Institute, Richmond, CA.
http://www.sfei.org/sites/default/files/RWSM_EMC_Year1_report_FINAL.pdf

Lent, M.A., Gilbreath, A.N., and McKee, L.J., 2012. Development of regional suspended sediment and pollutant load estimates for San Francisco Bay Area tributaries using the regional watershed spreadsheet model (RWSM): Year 2 progress report. A technical progress report prepared for the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP), Small Tributaries Loading Strategy (STLS). Contribution No. 667. San Francisco Estuary Institute, Richmond, California.
http://www.sfei.org/sites/default/files/RWSM_EMC_Year2_report_FINAL.pdf

Lewicki, M., and McKee, L.J, 2009. Watershed specific and regional scale suspended sediment loads for Bay Area small tributaries. A technical report for the Sources Pathways and Loading Workgroup of the Regional Monitoring Program for Water Quality: SFEI Contribution #566. San Francisco Estuary Institute, Oakland, CA. 28 pp + Appendices.
http://www.sfei.org/sites/default/files/566_RMP_RegionalSedimentLoads_final_web.pdf

Lewis, J. 1996. Turbidity-controlled suspended sediment sampling for runoff-event load estimation. Water Resources Research 32:2299-2310.

Lin, Y., R. Vogt, and T. Larssen. 2012. Environmental mercury in China: A review. Environmental Toxicology and Chemistry 31:2431-2444.

Line, D. E. 2013. Effect of development on water quality for seven streams in North Carolina. Environmental Monitoring and Assessment 185:6277-6289.

Ma, J.-S., J.-H. Kang, M. Kayhanian, and M. K. Stenstrom. 2009. Sampling Issues in Urban Runoff Monitoring Programs: Composite versus Grab. Journal of Environmental Engineering.

Marsalek, J. and H. Y. F. Ng. 1989. Evaluation of pollution loadings from urban nonpoint sources: methodology and applications. Journal Of Great Lakes Research 15:444-451.

Mason RP, Sullivan KA. Mercury and methylmercury transport through an urban watershed. Water Research 1998:32:321-30.

McKee, L.J., Gilbreath, A.N., Hunt, J.A., Wu, J., and Yee, D., 2015. Sources, Pathways and Loadings: Multi-Year Synthesis. 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). Contribution No. 773. San Francisco Estuary Institute, Richmond, California.
http://www.sfei.org/documents/sources-pathways-and-loadings-multi-year-synthesis-pcbs-and-hg

McKee, L.J., Gilbreath, A.N., Wu, J., Kunze, M.S., Hunt, J.A., 2014. Estimating Regional Pollutant Loads for San Francisco Bay Area Tributaries using the Regional Watershed Spreadsheet Model (RWSM): Year's 3 and 4 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). Contribution No. 737. San Francisco Estuary Institute, Richmond, California. http://www.sfei.org/sites/default/files/737%20RWSM%20Progress%20Report%20Y3_4%20for%20the%20WEB.pdf

McKee, L.J., Gilbreath, A.N., Gluchowski, D.C., Hunt, J.A., and Yee, D., 2013. Quality assurance methods for continuous rainfall, run-off, and turbidity data. A draft report prepared for Bay Area Stormwater Management Agencies Association (BASMAA), and the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP) Sources Pathways and Loadings Workgroup (SPLWG),Small Tributaries Loading Strategy (STLS). San Francisco Estuary Institute, Richmond, CA. http://www.sfei.org/sites/default/files/739.pdf

McKee, L.J., Gluchowski, D.C., Gilbreath, A.N., and Hunt, J.A., 2013. Pollutants of concern (POC) loads monitoring data progress report, water year (WY) 2012. 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). Contribution No. 690. San Francisco Estuary Institute, Richmond, California. See pages 574-633 of the report link below: http://www.swrcb.ca.gov/rwqcb2/water_issues/programs/stormwater/UC_Monitoring_Report_2012.pdf

McKee, L.J., Gilbreath, A.N., Hunt, J.A., and Greenfield, B.K., 2012. Pollutants of concern (POC) loads monitoring data, Water Year (WY) 2011. A technical report prepared for the Regional Monitoring Program for Water Quality in San Francisco Bay (RMP), Small Tributaries Loading Strategy (STLS). Contribution No. 680. San Francisco Estuary Institute, Richmond, California. http://www.sfei.org/sites/default/files/POC%20loads%20WY%202011%202013-03-03%20FINAL%20with%20Cover.pdf

McKee, L.J., Hunt, J., Greenfield, B.J., 2010. Concentration and loads of mercury species in the Guadalupe River, San Jose, California, Water Year 2010. A report prepared for the Santa Clara Valley Water District in Compliance with California Regional Water Quality Control Board San Francisco Bay Region Order Number 01- 036 as Amended by Order Number R2-2009-0044, Requirement D. October 29, 2010. San Francisco Estuary Institute. http://www.sfei.org/sites/default/files/SFEI_Guadalupe_final_report_12_23_10_0.pdf

McKee, L., Oram, J., Leatherbarrow, J., Bonnema, A., Heim, W., and Stephenson, M., 2006. Concentrations and loads of mercury, PCBs, and PBDEs in the lower Guadalupe River, San Jose, California: Water Years 2003, 2004, and 2005. A Technical Report of the Regional Watershed Program: SFEI Contribution 424. San Francisco Estuary Institute, Oakland, CA. 47pp + Appendix A and B. http://www.sfei.org/sites/default/files/424_Guadalupe_2005Report_Final_0.pdf

McKee, L., and Krottje, P.A., 2005 (Revised July 2008). Human influences on nitrogen and phosphorus concentrations in creek and river waters of the Napa and Sonoma watersheds, northern San Francisco Bay, California. A Technical Report of the Regional Watershed Program: SFEI Contribution #421. San Francisco Estuary Institute, Oakland, CA. 50pp. http://www.sfei.org/sites/default/files/McKeeandKrottje2005.pdf

McKee, L., Leatherbarrow, J., and Oram, J., 2005. Concentrations and loads of mercury, PCBs, and OC pesticides in the lower Guadalupe River, San Jose, California: Water Years 2003 and 2004. A Technical Report of the Regional Watershed Program: SFEI Contribution 409. San Francisco Estuary Institute, Oakland, CA. 72pp. http://www.sfei.org/sites/default/files/409_GuadalupeRiverLoadsYear2.pdf

McKee, L., Leatherbarrow, J., Eads, R., and Freeman, L., 2004. Concentrations and loads of PCBs, OC pesticides, and mercury associated with suspended sediments in the lower Guadalupe River, San Jose, California. A Technical Report of the Regional Watershed Program: SFEI Contribution #86. San Francisco Estuary Institute, Oakland, CA. 79pp. http://www.sfei.org/sites/default/files/GuadalupeYear1final.pdf

McKee, L., 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 66. San Francisco Estuary Institute, Oakland, Ca. http://www.sfei.org/sites/default/files/Urban_runoff_literature~000.pdf

Melwani, A., Lent, M., Greenfield, B., and McKee, L., 2010. Optimizing sampling methods for pollutant loads and trends in San Francisco Bay urban stormwater monitoring. A technical report for the Sources Pathways and Loading Workgroup of the Regional Monitoring Program for Water Quality. San Francisco Estuary Institute, Oakland, CA. Final Draft. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/stormwater/MRP/2011_AR/BASMAA/B2c_2010-11_MRP_AR.pdf

Moran, K.D. 2007. Urban use of the insecticide fipronil – water quality implications. Memorandum to the Urban Pesticide Committee. http://www.up3project.org/documents/Final_Fipronil_Memo_2007.pdf

Oram, J.J., McKee, L.J., Werme, C.E., Connor, M.S., Oros, D.R. 2008. A mass budget of PBDEs in San Francisco Bay, California, USA. Environment International 34, 1137-47. http://www.sciencedirect.com/science/article/pii/S0160412008000688

Owens, J., White, C., and Hecht, B., 2011. Mercury Sampling and Load Calculations at Upstream and Downstream Stations on the Guadalupe River, Santa Clara County, California, Water Year 2011. A report prepared for Santa Clara Valley Water District by Balance Hydrologics Inc.

Pearce, S., McKee, L.J., and Shonkoff, S., 2005. Pinole Creek watershed sediment source assessment. A Technical Report of the Regional Watershed Program prepared for the Contra Costa Resources Conservation District (CC RCD): SFEI Contribution #316. San Francisco Estuary Institute, Oakland, CA. 102pp + appendix. http://www.sfei.org/sites/default/files/PinoleCreekFinal.pdf

Phillips, B.M., Anderson, B.S., Voorhees, J.P., Hunt, J.W., Holmes, R.W., Mekebri, A., Connor, V., Tjeerdema,R.S., 2010b. The contribution of pyrethroid pesticides to sediment toxicity in four urban creeks in California, USA. Journal of Pesticide Science 35, 302-309.

Picado, F. and G. Bengtsson. 2012. Temporal and spatial distribution of waterborne mercury in a gold miner's river. Journal of Environmental Monitoring 14:2746-2754.

Quémerais, B., Cossa, D., Rondeau, B., Pham, T.T., Gagnon, P., Fottin, B., 1999. Sources and fluxes of mercury in the St. Lawrence River. Environmental Science and Technology 33, 840-49. http://pubs.acs.org/doi/abs/10.1021/es980400a?journalCode=esthag

Rasmussen, P.P., Gray, J.R., Glysson, G.D., and Ziegler, A.C., 2009, Guidelines and procedures for computing timeseries suspended-sediment concentrations and loads from in-stream turbidity-sensor and streamflow data: U.S. Geological Survey Techniques and Methods, book 3, chap. C4, 52 pp.

Rimondi, V., P. Costagliola, J. E. Gray, P. Lattanzi, M. Nannucci, M. Paolieri, and A. Salvadori. 2014. Mass loads of dissolved and particulate mercury and other trace elements in the Mt. Amiata mining district, Southern Tuscany (Italy). Environmental Science and Pollution Research 21:5575-5585.

Riscassi, A. L. and T. M. Scanlon. 2013. Particulate and dissolved mercury export in streamwater within three mid-Appalachian forested watersheds in the US. Journal of Hydrology 501:92-100.

Riverside County Flood Control and Water Conservation District (Riverside County) (2007). Santa Margarita Region Monitoring Annual Report Fiscal Year 2006-2007.

Rothenberg, S.E., McKee, L.J., Gilbreath, A., Yee, D., and Conner, M., and Fu, X., 2010a. Wet deposition of mercury within the vicinity of a cement plant before and during cement plant maintenance. Atmospheric Environment 44, 1255-1262.

Rothenberg, S.E., McKee, L.J., Gilbreath, A., Yee, D., and Conner, M., and Fu, X., 2010b. Evidence for short range transport of atmospheric mercury to a rural, inland site. Atmospheric Environment 44, 1263-1273.

Rowland, A. P., C. Neal, P. Scholefield, A. P. Halford, C. D. Vincent, and K. Hockenhull. 2010. Mercury in rivers in NW England: from rural headwaters to the heartlands of the historic industrial base. Journal of Environmental Monitoring 12:2299-2306.

Ruzycki, E. M., R. P. Axler, J. R. Henneck, N. R. Will, and G. E. Host. 2011. Estimating mercury concentrations and loads from four western Lake Superior watersheds using continuous in-stream turbidity monitoring. Aquatic Ecosystem Health & Management 14:422-432.

Sickman, J. O., M. J. Zanoli, and H. L. Mann. 2007. Effects of urbanization on organic carbon loads in the Sacramento River, California. Water Resources Research 43.

SFEI, 2009. RMP Small Tributaries Loading Strategy. A report prepared by the strategy team (L McKee, A Feng, C Sommers, R Looker) for the Regional Monitoring Program for Water Quality. SFEI Contribution #585. San Francisco Estuary Institute, Oakland, CA. http://www.sfei.org/rmp/stls

SFRWQCB, 2006. California Regional Water Quality Control Board San Francisco Bay Region Mercury in San Francisco Bay Proposed Basin Plan Amendment and Staff Report for Revised Total Maximum Daily Load (TMDL) and Proposed Mercury Water Quality Objectives. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/TMDLs/sfbaymercury/sr080906.pdf

SFRWQCB, 2008. California Regional Water Quality Control Board San Francisco Bay Region Total Maximum Daily Load for PCBs in San Francisco Bay Staff Report for Proposed Basin Plan Amendment. 134 pp. http://www.waterboards.ca.gov/sanfranciscobay/board_info/agendas/2008/february/tmdl/appc_pcbs_staffrept.pdf

SFRWQCB, 2009. California Regional Water Quality Control Board San Francisco Bay Region Municipal Regional Stormwater NPDES Permit, Order R2-2009-0074, NPDES Permit No. CAS612008. Adopted October 14, 2009. 279pp. http://www.waterboards.ca.gov/sanfranciscobay/water_issues/programs/stormwater/Municipal/index.shtml

Steding, D. J. and A. R. Flegal. 2002. Mercury concentrations in coastal California precipitation: Evidence of local and trans-Pacific fluxes of mercury to North America. Journal Of Geophysical Research-Atmospheres 107.

Stone et al., 2000. Flow-proportional, time-composited, and grab sample estimation of nitrogen export from an eastern Coastal Plain watershed.

Thomas, M. A., C. H. Conaway, D. J. Steding, M. Marvin-DiPasquale, K. E. Abu-Saba, and A. R. Flegal. 2002. Mercury contamination from historic mining in water and sediment, Guadalupe River and San Francisco Bay, California. Geochemistry: Exploration, Environment, Analysis 2:1-7.

Ullrich, S. M., M. A. Llyushchenko, G. A. Uskov, and T. W. Tanton. 2007. Mercury distribution and transport in a contaminated river system in Kazakhstan and associated impacts on aquatic biota. Applied Geochemistry 22:2706-2734.

United States (US) Census Bureau (2010). Population statistics for the United States of America based on the 2010 census. http://www.census.gov/#

Wall, G.R., Ingleston, H.H., Litten S., 2005. Calculating mercury loading to the tidal Hudson River, New York, using rating curve and surrogate methodologies. Water Air Soil Pollution 165, 233–48. http://pubs.er.usgs.gov/publication/70031430

Walling, D.E., Webb, B.W., 1985. Estimating the discharge of contaminants to coastal waters by rivers: some cautionary comments. Marine Pollution Bulletin 16, 488-92. http://www.sciencedirect.com/science/article/pii/0025326X85903820

Warrick, J. A., M. A. Madej, M. A. Goñi, and R. A. Wheatcroft. 2013. Trends in the suspended-sediment yields of coastal rivers of northern California, 1955–2010. Journal of Hydrology 489:108–123.

Warrick, J. A. and D. M. Rubin. 2007. Suspended-sediment rating curve response to urbanization and wildfire, Santa Ana River, California. Journal of Geophysical Research 112:15 pp.

Werner, I., Deanovic, L.A., Markewicz, D., Khamphanh, M., Reece, C.K., Stillway, M., Reece, C., 2010. Monitoring acute and chronic water column toxicity in the northern Sacramento-San Joaquin Estuary, California, USA, using the euryhaline amphipod, Hyalella azteca: 2006 to 2007. Environmental Toxicology and Chemistry 29, 2190-2199.

Weston Solutions (2006). Toxicity Identification Evaluation (TIE) of County of San Diego and Copermittees Chollas Creek Stormwater Sampling. September.

Weston, D.P., Holmes, R.W., You, J., Lydy, M.J., 2005. Aquatic toxicity due to residential use of pyrethroid insecticides. Environmental Science & Technology 39, 9778-9784.

Weston, D.P., Lydy, M.J., 2010a. Focused toxicity identification evaluations to rapidly identify the cause of toxicity in environmental samples. Chemosphere 78, 368-374.

Weston, D.P., Lydy, M.J., 2010b. Urban and Agricultural Sources of Pyrethroid Insecticides to the Sacramento-San Joaquin Delta of California. Environmental Science & Technology 44, 1833-1840.

Wood, M.L., Morris, P.W., Cooke, J., and Louie, S.L., 2010. Amendments to the Water Quality Control Plan for the Sacramento River and San Joaquin River Basins for the Control of Methylmercury and Total Mercury in the Sacramento-San Joaquin Delta Estuary. Staff Report prepared for the Regional Water Quality Control Board - Central Valley Region. April 2010. 511pp. http://www.waterboards.ca.gov/centralvalley/water_issues/tmdl/central_valley_projects/delta_hg/april_2010_hg_tmdl_hearing/apr2010_bpa_staffrpt_final.pdf

Zgheib, S., R. Moilleron, and G. Chebbo. 2011. Influence of the land use pattern on the concentrations and fluxes of priority pollutants in urban stormwater. Water Science and Technology 64:1450-1458.

Zgheib, S., R. Moilleron, and G. Chebbo. 2012. Priority pollutants in urban stormwater: Part 1-Case of separate storm sewers. Water Research 46:6683-6692.

Zheng, W., S. C. Kang, X. B. Feng, Q. G. Zhang, and C. L. Li. 2010. Mercury speciation and spatial distribution in surface waters of the Yarlung Zangbo River, Tibet. Chinese Sci. Bulletin 55:2697-2703.

8. Detailed information for each sampling location

8.1. Marsh Creek

8.1.1. *Marsh Creek flow*

The US geological survey has maintained a flow record on Marsh Creek (gauge number 11337600) since October 1, 2000 (13 WYs). Data collection at this site was discontinued after September 30, 2013 due to budget reductions. Flow for WY 2014 was based on a continuous stage record generated by the STLS sampling team combined with the flow rating curve provided by the USGS. Peak annual flows for the 14 years have ranged between 168 cfs (1/22/2009) and 1770 cfs (1/2/2006). For the same period, annual runoff has ranged between 3.03 Mm^3 (WY 2009) and 26.8 Mm^3 (WY 2006). In the Bay Area, at least 30 years of observations are needed at a particular site to get a reasonable understanding of climatic variability (McKee et al., 2003). Since, at this time, Marsh Creek has a relatively short history of gauging, flow record on Marsh Creek were compared with a reasonably long record at an adjacent monitoring station near San Ramon. Based on this comparison, WY 2006 may be considered representative of very rare wet conditions (upper 10th percentile) and WY 2009 is perhaps representative of moderately rare dry conditions (lower 20th percentile) based on records that began in WY 1953 at San Ramon Creek near San Ramon (USGS gauge number 11182500).

A number of relatively minor storms occurred during WYs 2012, 2013, and 2014 (4). In WY 2012, flow peaked at 174 cfs on 1/21/2012 at 1:30 am and then again 51 ½ hours later at 143 cfs on 1/23/2012 at 5:00 am. Total runoff during the whole of WY 2012 (October 1st to September 30th) was 1.87 Mm^3. During WY 2013, flow peaked at 1300 cfs at 10:00 am on 11/30/2012; total run-off for the water year was 6.26 Mm^3. During WY 2014, flow peaked at 441 cfs on 2/28/2014 at 6:20 am and total runoff was 1.31 Mm^3, the lowest of the 3 years of observations during the study and the lowest in the 14 year record for the site. Although the peak discharge for WY 2013 was the second highest since records began in WY 2001, total annual flow ranked eighth in the last 13 years. Thus, discharge of these magnitudes for all three water years are likely exceeded most years in this watershed. Rainfall data corroborates this assertion; rainfall during WYs 2012, 2013, and 2014 respectively were 70%, 71%, and 61% of mean annual precipitation (MAP) based on a long-term record at Concord Wastewater treatment plant (NOAA gauge number 041967) for the period Climate Year (CY) 1992-2014. Marsh Creek has a history of mercury mining in the upper part of the watershed. The Marsh Creek Reservoir is downstream from the historic mining area but upstream of the current gauging location. During WYs 2012 to 2014, discharge through the reservoir occurred on March, November, and December 2012. It is possible that in the future when larger releases occur, additional Hg loads may be transported down the Creek system but for these dry years, this was not a big component of the flow-source-transport process.

8.1.2. *Marsh Creek turbidity and suspended sediment concentration*

Turbidity generally responded to rainfall events in a similar manner to runoff. During WY 2012, turbidity peaked at 532 NTU during a late season storm on 4/13/12 at 7 pm. Relative to flow magnitude, turbidity

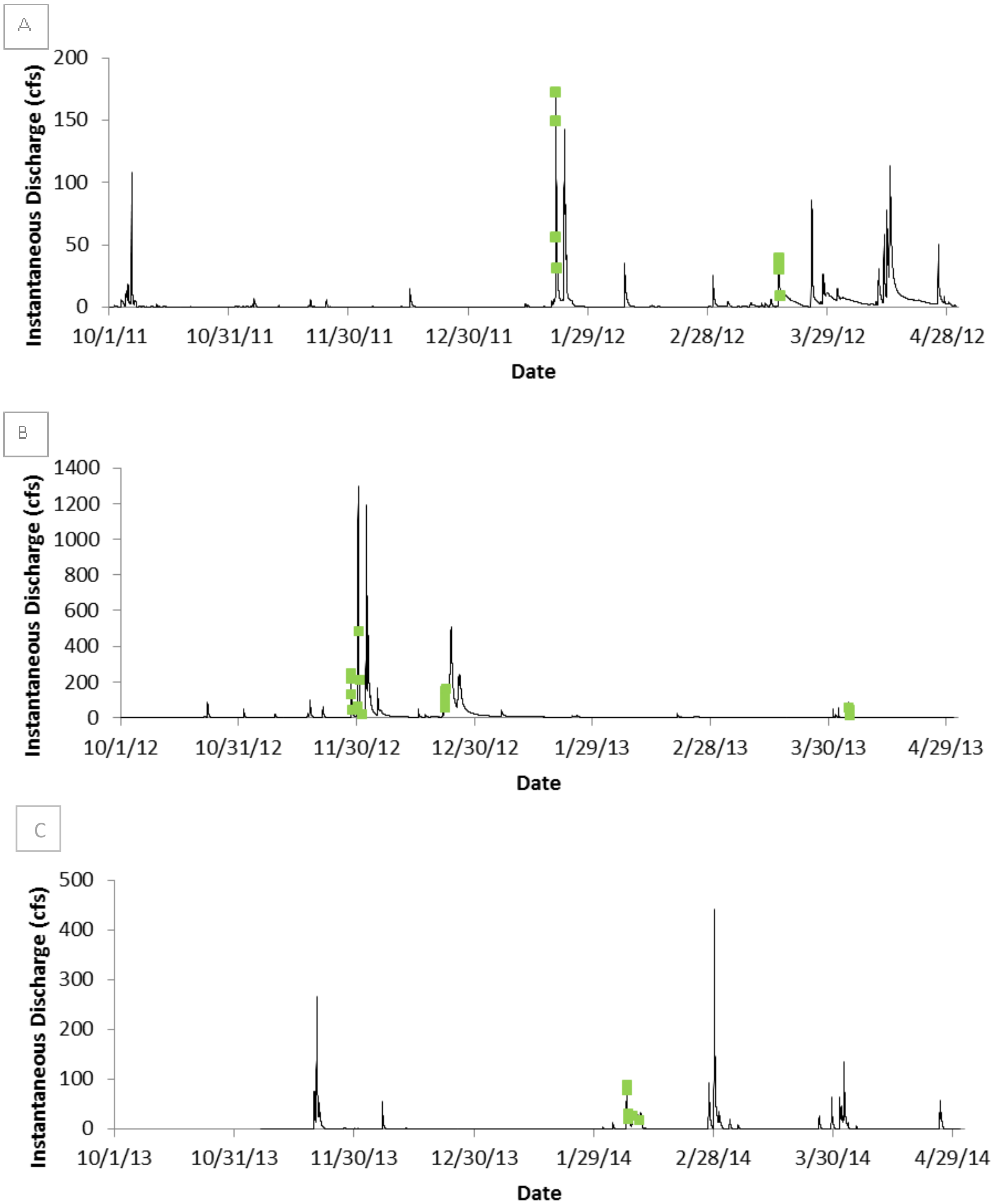

Figure 4. Flow characteristics in Marsh Creek during Water Year 2012 (A) and Water Year 2013 (B) based on published 15 minute data provided by the United States Geological Survey, gauge number 11337600) with sampling events plotted in green. Flow for WY 2014 (C) was based on stage measurements taken by the STLS study team combined with the USGS rating curve for the site.

remained elevated during all storms and was the greatest during the last storm despite lower flow. During WY 2013, turbidity peaked at 1384 NTU during the December storm series on 12/02/12 at 7:05pm. This occurred during a period when the Marsh Creek Reservoir was overflowing. During WY 2014, turbidity peaked at 458 NTU during the November storm on 11/20/2013 at 2:30 pm, very similar to the peak turbidity (432 NTU) observed later in the year during the storm that yielded the peak flow for the year. These observations, and observations made previously during the RMP reconnaissance study (maximum 3211 NTU; McKee et al., 2012), provide evidence that during larger storms and wetter years, the Marsh Creek watershed is capable of much greater sediment erosion and transport than occurred during observations in the three WYs reported here, resulting in greater turbidity and concentrations of suspended sediment. The OBS-500 instrument utilized at this sampling location with a range of 0-4000 NTU will likely be exceeded during larger storms if such storms are observed during some future sampling effort.

Suspended sediment concentration, since it was computed from the continuous turbidity data, follows the same patterns as turbidity in relation to discharge. Computed SSC peaked at 1312 mg/L during the 4/13/12 late season storm, at 1849 mg/L on 12/02/12, and at 682 mg/L on 11/20/2013 at 2:30 pm at the same times as the peaks in turbidity. During WY 2012, relative to flow magnitude, SSC remained elevated during all storms and was the greatest during the last storm despite lower flow. A similar pattern was also observed during WY 2013. Turbidity and computed SSC peaked during a smaller storm in December rather than the largest storm which occurred in late November. Turbidity remained relatively elevated from an even smaller storm that occurred on December 24th. This pattern was not observed in WY 2014 perhaps because storms were minor and few. Observations of increased sediment transport as the season progresses relative to flow in addition to the maximum SSC observed during the RMP reconnaissance study of 4139 mg/L (McKee et al., 2012), suggest that in wetter years, greater SSC can be expected.

8.1.3. Marsh Creek POC concentrations summary (summary statistics)

In relation to the other five monitoring locations, Marsh Creek is representative of a relatively rural watershed with lower urbanization but potentially impacted by mercury residues from historic mining upstream. Summary statistics (Table 11) were used to provide useful information to compare Marsh Creek water quality to other Bay Area streams. The comparison of summary statistics to knowledge from other watersheds and conceptual models of pollutant sources and transport processes provided a further check on data quality.

The maximum PCB concentration (4.32 ng/L) was similar to background concentrations normally found in relatively nonurban areas (Lent and McKee, 2011). For example, maximum concentrations in watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). Marsh Creek, at the sampling point, has the lowest percentage imperviousness (10%) of any Bay Area watershed measured to-date for PCBs and exhibits the lowest measured particle ratio of 5 pg/mg. If this is taken to be background for the Bay Area, any rural watershed with little urban land use that has suspended

Table 11. Summary of laboratory measured pollutant concentrations in Marsh Creek during WY 2012, 2013, and 2014.

		2012							2013							2014						
Analyte	Unit	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation
SSC	mg/L	27	96%	0	930	180	297	276	54	100%	3.3	1040	167	217	230	20	75%	0	161	12	41.9	57
ΣPCB	ng/L	7	100%	0.354	4.32	1.27	1.95	1.61	15	100%	0.24	3.46	0.676	0.927	0.856							
Total Hg	ng/L	8	100%	8.31	252	34.5	74.3	85.2	17	100%	1.9	120	19	32.5	33.9	6	100%	2.4	18	4.55	7.35	6.02
Total MeHg	ng/L	5	100%	0.085	0.406	0.185	0.218	0.12	14	93%	0	1.2	0.185	0.337	0.381	1	0%	0	0	0	0	
TOC	mg/L	8	100%	4.6	12.4	8.55	8.34	2.37	16	100%	4.3	9.5	6.55	6.52	1.6	6	100%	6	8.7	7.05	7.17	1.04
NO3	mg/L	8	100%	0.47	1.1	0.635	0.676	0.202	16	94%	0	1	0.525	0.531	0.222	4	100%	0.28	0.59	0.575	0.505	0.15
Total P	mg/L	8	100%	0.295	1.1	0.545	0.576	0.285	16	100%	0.14	0.95	0.34	0.395	0.21	6	100%	0.097	0.5	0.22	0.255	0.137
PO4	mg/L	8	100%	0.022	0.12	0.0563	0.0654	0.0298	16	100%	0.046	0.18	0.11	0.114	0.0365	6	100%	0.046	0.15	0.108	0.101	0.0415
Hardness	mg/L	2	100%	200	203	202	202	2.12								2	100%	120	180	150	150	42.4
Total Cu	ug/L	2	100%	13.8	27.5	20.6	20.6	9.7	4	100%	3.8	30	12.5	14.7	11	2	100%	4.5	4.7	4.6	4.6	0.141
Dissolved Cu	ug/L	2	100%	4.99	5.62	5.3	5.3	0.445	4	100%	1.3	2.4	1.45	1.65	0.52	2	100%	2.1	2.6	2.35	2.35	0.354
Total Se	ug/L	2	100%	0.647	0.784	0.716	0.716	0.0969	4	100%	0.525	1.4	0.67	0.816	0.395	2	100%	0.44	0.8	0.62	0.62	0.255
Dissolved Se	ug/L	2	100%	0.483	0.802	0.643	0.643	0.226	4	100%	0.51	1.2	0.585	0.72	0.323	2	100%	0.42	0.59	0.505	0.505	0.12
Carbaryl	ng/L	2	50%	0	16	8	8	11.3	4	25%	0	13	0	3.25	6.5	2	0%	0	0	0	0	0
Fipronil	ng/L	2	100%	7	18	12.5	12.5	7.78	4	100%	10	13	10.8	11.1	1.44	2	100%	13	15	14	14	1.41
ΣPAH	ng/L	1	100%	216	216	216	216		2	100%	85.7	222	154	154	96.4	1	100%	37.8	37.8	37.8	37.8	
ΣPBDE	ng/L	1	100%	20	20	20	20		2	100%	11.2	56.4	33.8	33.8	32	1	100%	20.3	20.3	20.3	20.3	
Delta/ Tralomethrin	ng/L	2	100%	0.954	5.52	3.23	3.23	3.23	4	75%	0	2.2	0.75	0.925	0.943	2	50%	0	1.8	0.9	0.9	1.27
Cypermethrin	ng/L	2	50%	0	68.5	34.2	34.2	48.4	4	100%	1.8	13	2.15	4.78	5.49	2	100%	0.6	5.3	2.95	2.95	3.32
Cyhalothrin lambda	ng/L	2	50%	0	2.92	1.46	1.46	2.06	4	100%	0.5	3.2	0.8	1.33	1.27	1	100%	0.4	0.4	0.4	0.4	
Permethrin	ng/L	2	100%	3.81	17.3	10.6	10.6	9.54	4	75%	0	12	6.55	6.28	6.11	2	50%	0	2.4	1.2	1.2	1.7
Bifenthrin	ng/L	2	100%	25.3	257	141	141	163	4	100%	27	150	45	66.8	56.2	2	100%	20	33	26.5	26.5	9.19

Analyzed but not detected: Fenpropathrin, Esfenvalerate/ Fenvalerate, Cyfluthrin, Allethrin, Prallethrin, Phenothrin, and Resmethrin
Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation at Marsh Creek was two.
All Hardness results in WY 2013 were censored.

sediment concentrations during flood periods exceeding 1000 mg/L could be expected to exhibit PCB concentrations exceeding 5 ng/L. Of the 23 Bay Area watersheds reviewed by McKee et al. (2003), rural dominated areas including Cull Creek above Cull Creek Reservoir, San Lorenzo Creek above Don Castro Reservoir, Wildcat Creek near the park entrance, and Crow Creek exhibited FWMC > 1000 mg/L and could, if measured, show similar PCB concentrations to those observed in Marsh Creek.

Maximum total mercury concentrations (252 ng/L) were similar to concentrations found in mixed land use watersheds with some urban related influence such as atmospheric burden (McKee et al., 2004; Lent and McKee, 2011). Given global Hg cycling has a large atmospheric component (Fitzgerald et al., 1998; Lamborg et al., 2002; Steding and Flegal, 2002) and background soil concentrations in California are typically on the order of 0.1 mg/kg (equivalent to ng/mg) (Bradford et al., 1996), concentrations of this magnitude in a watershed with higher sediment erosion and higher average suspended sediment concentrations can occur when associated with the transport of low concentration particles (McKee et al., 2012). Thus Bay Area watersheds that exhibit suspended sediment concentrations in excess of 2,000 mg/L during floods should exhibit total Hg concentrations during floods in excess of 200 ng/L, even when no urban or mining sources are present. The particle ratio of Hg in Marsh Creek averaged 0.21 mg/kg for the three years of study, only 3-fold background CA soils concentrations, and was the 5th lowest observed in Bay Area watersheds to-date.

Maximum MeHg concentrations (0.407 ng/L during WY 2012, 1.2 ng/L during WY 2013, and ND during WY 2014 for the single sample collected at low flow) were greater during the first two years of observations than the proposed implementation goal of 0.06 ng/l for methylmercury in ambient water for watersheds tributary to the Central Delta (Wood et al., 2010: Table 4.1, page 40), however concentrations of this magnitude or greater have been observed in a number of Bay Area watersheds (Guadalupe River: McKee et al., 2006; McKee et al., 2010; Zone 4 Line A: Gilbreath et al., 2012; Glen Echo Creek and Zone 5 Line M: McKee et al., 2012). Indeed, concentrations of methylmercury of this magnitude have commonly been observed in rural watersheds (Domagalski, 2001; Balogh et al., 2002) and production has been related to organic carbon transport, riparian processes and percentage of watershed with wetlands (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed that Hg sources are not a primary limiting factor in MeHg production.

Nutrient concentrations appear to be reasonably typical of other Bay Area rural watersheds (McKee and Krottje, 2005; Pearce et al., 2005) but perhaps a little greater for PO4 and TP than concentrations found in watersheds in grazing land use from other parts of the country and world (e.g. three rural dominated watersheds North Carolina: Line, 2013; comprehensive Australian literature review for concentrations bay land use class: Bartley et al., 2012). This appears typical in the Bay Area; phosphorus concentrations appear greater than elsewhere in the world under similar land use scenarios, an observation perhaps attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Organic carbon concentrations observed in Marsh Creek were lesser than observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012) but compared more closely to Belmont, Borel, Calabazas, San Tomas, and Walnut Creeks (McKee et al., 2012). Indeed, TOC concentrations of 4-12 mg/L have been observed elsewhere in California (Sacramento River: Sickman et al., 2007).

For pollutants sampled at a sufficient frequency for loads analysis (suspended sediments, PCBs, mercury, organic carbon, and nutrients), concentrations exhibited the typical pattern of median < mean with the exception of organic carbon. A similar style of first order quality assurance based on comparisons to observations in other studies is also possible for analytes measured at a lower frequency. Pollutants sampled at a lesser frequency using composite sampling design (see methods section) and appropriate for characterization only (copper, selenium, PAHs, carbaryl, fipronil, and PBDEs) were quite low and similar to concentrations found in watersheds with limited or no urban influences. Carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and California (fipronil: 70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012). The Carbaryl concentrations we observed were more similar to those observed in tributaries to Salton Sea, Southern CA (geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008). Pyrethroid concentrations of Delta/ Tralomethrin were similar to those observed in Zone 4 Line A, a small 100% urban tributary in Hayward, whereas concentrations of Permethrin and Cyhalothrin lambda were about 10-fold and 2-fold lower and concentrations of Bifenthrin were about 5-fold higher; cypermethrin was not detected in Z4LA (Gilbreath et al., 2012). In summary, the statistics indicate pollutant concentrations typical of a Bay Area non-urban stream and there is no reason to suspect data quality issues.

8.1.4. Marsh Creek toxicity

Composite water samples were collected at the Marsh Creek station during two storm events in WY 2012, four storm events in WY 2013 and two events in WY 2014. No significant reductions in the survival, reproduction and growth of three of four test species were observed during WY 2012 – WY 2014 except two occurrences of fathead minnow testing with 17% mortality rate (WY 2014 sample) and 42% mortality rate (WY 2013). Significant reductions in the survival of the amphipod Hyalella azteca was observed during both WY 2012 storm events while WY 2013 and 2014 had complete mortality of Hyalella Azteca between 5 and 10 days of exposure to storm water during all storm events.

8.1.5. Marsh Creek loading estimates

Site-specific methods were developed for computed loads (Table 12). Methylmercury data was flow-stratified for improved relationships between turbidity and the pollutant under different flow conditions. Preliminary loads estimates generated for WY 2012 and reported by McKee et al. (2013) have now been revised based on additional data collected in WY 2013 and 2014 and an improving understanding of pollutant transport processes for the site. Monthly loading estimates correlate well with monthly discharge (Table 13). There are no data available for October and November 2011 and October 2013 because monitoring equipment was not installed. Monthly discharge was greatest in December 2012 as were the monthly loads for each of the pollutants regardless of transport mode (dominantly particulate or dissolved). The discharge was relatively high for December given the rainfall, an indicator that the watershed was reasonably saturated by this time. The sediment loads are well-aligned with the total discharge and the very high December 2012 sediment load appears real; the watershed became saturated after late November rains such that early December and Christmas time storms transported a lot of sediment. Monthly loads of total Hg appear to correlate with discharge for all months; this would not be the case if there was variable release of mercury from historic mining sources upstream associated with climatic and reservoir discharge conditions. Importantly, if data were to be collected to capture periods when saturated and high rainfall conditions occur along with reservoir releases, new information may emerge about the influence, if any, of Hg pollution associated with historic mining. If these conditions were to result in significant Hg releases, then any estimate of long term average load might be elevated above what can be computed now. Given the very dry flow conditions of WYs 2012, 2013, and 2014 (see discussion on flow above), loads presented here are considered representative of dry conditions.

Table 12. Regression equations used for loads computations for Marsh Creek during water years 2012, 2013 and 2014.

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Suspended Sediment (mg/L/NTU)	Mainly urban	1.49		0.63	Regression with turbidity
Total PCBs (ng/L/NTU)	Mainly urban	0.00878		0.86	Regression with turbidity
Total Mercury (ng/L/NTU)	Mainly urban	0.3174		0.68	Regression with turbidity
Total Methylmercury (ng/L/NTU) - Storm Flows	Mainly urban	0.00136	0.0199	0.86	Regression with turbidity
Total Methylmercury (ng/L/NTU) - Low Flow[a]	Mainly urban	0.0067	0.039	0.94	Regression with turbidity
Total Organic Carbon (mg/L)	Mainly urban	6.9			Flow weighted mean concentration
Total Phosphorous (mg/L/NTU)	Mainly urban	0.00174	0.176	0.71	Regression with turbidity
Nitrate (mg/L)	Mainly urban	0.594			Flow weighted mean concentration
Phosphate (mg/L)	Mainly urban	0.111			Flow weighted mean concentration

[a] Includes small storms after extended dry periods.

Table 13. Monthly loads for Lower Marsh Creek during water years 2012 - 2014. Italicized loads are estimated based on monthly rainfall-load relationships.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2012	11-Oct	33	*0.153*	*9.59*	*1,057*	*0.056*	*1.73*	*0.0224*	*91.0*	*17.0*	*44.2*
	11-Nov	26	*0.0717*	*2.72*	*495*	*0.0159*	*0.50*	*0.0087*	*42.6*	*7.96*	*17.5*
	11-Dec	6	0.0252	0.819	174	0.00483	0.247	0.00466	14.8	2.77	5.38
	12-Jan	51	0.318	77.5	2,443	0.414	19.1	0.0687	190	33.1	158
	12-Feb	22	0.0780	4.56	538	0.0269	1.377	0.00704	46.0	8.58	19.0
	12-Mar	60	0.361	23.5	2,485	0.148	6.64	0.0321	213	38.8	93.8
	12-Apr	59	0.607	114	4,188	0.673	34.5	0.118	358	66.8	240
	Wet season total	257	1.61	233	11,380	1.34	64.0	0.262	956	175	578
2013	12-Oct	23	0.0875	7.98	603	0.0470	1.22	0.0393	51.6	9.62	24.7
	12-Nov	96	0.989	237	6,309	1.42	32.2	0.331	625	132	457
	12-Dec	75	4.00	2,435	27,474	14.4	372	2.32	2,363	444	3,573
	13-Jan	15	0.428	11.1	2,955	0.0655	1.69	0.0256	253	47.1	88.3
	13-Feb	6	0.142	1.39	981	0.00819	0.212	0.0118	83.9	15.6	26.7
	13-Mar	9	0.0721	1.57	497	0.00925	0.239	0.00987	42.5	7.93	14.5
	13-Apr	19	0.0978	8.75	680	0.0476	1.34	0.0412	54.8	10.5	28.0
	Wet season total	243	5.82	2,703	39,500	16.0	408	2.78	3,474	666	4,212
2014	13-Oct	1	*0.0252*	*0.48*	*174*	*0.00280*	*0.0885*	*0.00237*	*15.0*	*2.80*	*4.91*
	13-Nov	41	0.261	49.1	1,800	0.289	7.48	0.0504	154	28.7	103
	13-Dec	6	0.005	0.0185	36.5	0.000109	0.00282	0.000256	3.12	0.582	0.953
	14-Jan	4	0.032	1.39	224	0.00821	0.212	0.00225	19.1	3.56	7.33
	14-Feb	79	0.618	122	4308	0.729	18.5	0.126	363	69.1	259
	14-Mar	24	0.179	9.17	1232	0.0540	1.40	0.0128	105	19.6	42.1
	14-Apr	29	0.215	20.2	1483	0.119	3.07	0.0231	127	23.6	61.4
	Wet season total	184	1.34	202	9,257	1.20	30.7	0.217	786	148	479

[a] April 2012 monthly loads are reported for only the period April 01-26. In the 4 days missing from the record, <0.03 inches of rain fell in the lower watershed.

[b] October 2012 monthly loads are reported for only the period October 19-31. In the 18 days missing from the record, <0.05 inches of rain fell in the lower watershed.

[c] April 2013 monthly loads are reported for only the period April 01-18. In the 12 days missing from the record, no rain fell in the lower watershed.

[d] November 2013 are reported for only the period November 6-30. No rain fell during the missing period.

8.2. North Richmond Pump Station

8.2.1. North Richmond Pump Station flow

Richmond discharge estimates were calculated during periods of active pumping at the station during WYs 2013 and 2014 (Figure 5). Discharge estimates include all data collected when the pump rate was operating at greater than 330 RPM, the rate which marks the low end of the pump curve provided by the pump station. This rate is generally reached 30 seconds after pump ignition. For the purposes of this study, flows at less than 330 RPM were considered negligible due to limitations of the pump efficiency curve. This assumption may have resulted in slight underestimation of active flow from the station particularly during shorter duration pump outs but this under estimate was minor relative to storm and annual flows. The annual estimated discharge from the station was 0.74 Mm^3 for WY 2013 and 0.50 Mm^3 for WY 2014. A discharge estimate at the station for WY 2011 was 1.1 Mm^3 (Hunt et al., 2012). The rainfall to runoff ratios between the two studies was similar supporting the hypothesis that the flows and resulting load estimates from the previous study remain valid.

Precipitation in WY 2013 was 89% mean annual precipitation (MAP) based on a long-term record PRISM data record (modeled PRISM data) for the period Climate Year (CY) 1970-2000. Thus it appears WY 2013 was slightly drier than average. Of the total annual rainfall, 74% fell during a series of larger events in the period late November to December. Otherwise, WY 2013 had a number of very small events, three of which were sampled for water quality (Figure 5). The pumps at this pump station operate at a single speed, and therefor flow rates at this location are governed by the number of pumps operating at a given time. Most pump-outs during these storms had one operating pump except for a few storm events where two pumps were in operation. Flow "peaked" during one of these times when two pumps were in operation simultaneously. The peak rate was 210 cfs and occurred on December 2, 2013 after approximately 3.8 inches of rain fell over a 63 hour period.

WY 2014 was even drier than the previous year, with only 62% MAP (12.8 inches of rain). In total, five events were sampled for water quality, including the intense early season first flush on November 19 and 20, 2013, and multiple events in February 2014. Similar to WY 2013, a single pump operated for the majority of pump outs, with only a couple of occasions when two pumps were simultaneously operating. Flow peaked at 191 cfs on March 29th, 2014 after 0.84 inches fell in the previous three hours.

8.2.2. North Richmond Pump Station turbidity and suspended sediment concentration

Maximum turbidity during the study was measured at 772 NTU and which occurred during a dry flow pump out on January 24, 2013 following a low magnitude storm event of 0.22 inches on January 23rd. Maximum turbidity during other storm events ranged up to 428 NTU in WY 2013 and 466 NTU in WY 2014. Storms typically peaked in turbidity between 150 and 500 NTU. The pattern of turbidity variation over the wet season was remarkably similar to that observed during WY 2011 in the previous study (Hunt et al., 2012). The turbidity dataset collected by Hunt et al. (2012) was noisy and contained unexplainable turbidity spikes that were censored. The similarities between the WY 2011 and 2013 datasets suggest that the WY 2011 data set was not over-censored and therefore that pollutant loads based on both flow and turbidity computed by Hunt et al. (2012) remain valid. Suspended sediment

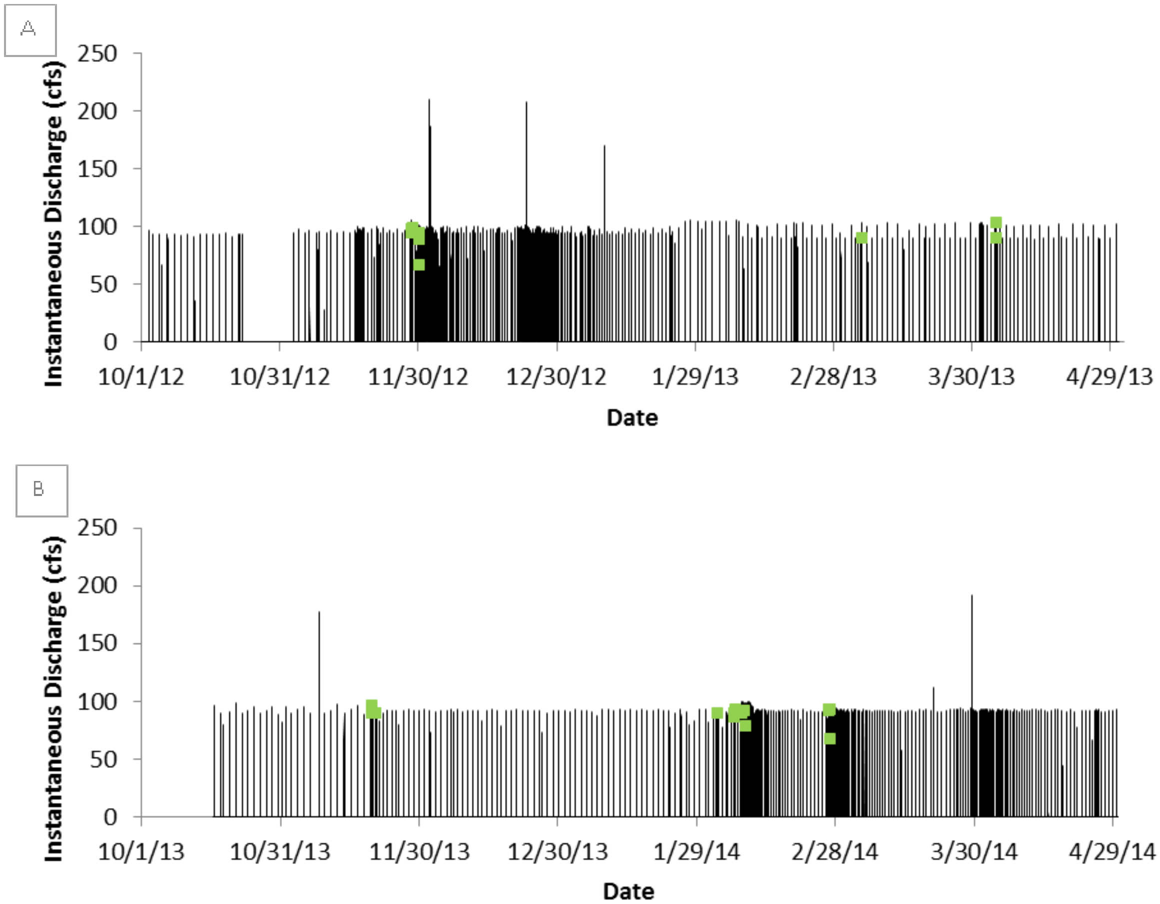

Figure 5. Flow characteristics at North Richmond Pump Station during Water Year 2013 and 2014 with sampling events plotted in green.

concentration was computed from the continuous turbidity data. Computed SSC peaked at 1010 mg/L during the 1/24/13 low flow pump out when turbidity also peaked. In WY 2014, the peak computed SSC was 579 mg/L during the 3/26/14 event; SSC in most storms peaked between 200 and 600 mg/L.

8.2.3. North Richmond Pump Station POC concentrations (summary statistics)

The North Richmond Pump Station is a 1.6 km watershed primarily comprised of industrial, transportation, and residential land uses. The watershed has a long history of industrial land use and is downwind from the Richmond Chevron Oil Refinery and the Port of Richmond. The land-use configuration results in a watershed that is approximately 62% covered by impervious surface and these land use and history factors help to contribute to potentially high concentrations loads of PCB and Hg. Summary statistics (Table 14) were used to provide useful information to compare Richmond pump station water quality to other Bay Area monitoring locations. The comparison of summary statistics to knowledge from other watersheds and conceptual models of pollutant sources and transport processes provided a further check on data quality.

Table 14. Summary of laboratory measured pollutant concentrations in North Richmond Pump Station during water year 2013 and 2014.

		2013							2014						
Analyte	**Unit**	**Samples Taken (n)**	**Proportion Detected (%)**	**Min**	**Max**	**Median**	**Mean**	**Standard Deviation**	**Samples Taken (n)**	**Proportion Detected (%)**	**Min**	**Max**	**Median**	**Mean**	**Standard Deviation**
SSC	mg/L	41	95%	0	213	26.5	45.7	54.3	67	99%	0	325	52	63.9	58.1
ΣPCB	ng/L	12	100%	4.85	31.6	10.1	12	7.09	20	100%	2.23	38.5	13.7	15	9.83
Total Hg	ng/L	12	100%	13	98	18.5	27.7	24.6	20	100%	11.5	230	28.5	46.7	51.8
Total MeHg	ng/L	6	100%	0.03	0.19	0.145	0.118	0.0705	10	100%	0.03	1.1	0.16	0.261	0.309
TOC	mg/L	12	100%	3.5	13.5	6.6	7.46	3.36	20	100%	5.2	60	9.85	13.4	12.4
NO3	mg/L	12	100%	0.21	3.1	0.855	1.13	0.848	20	100%	0.32	3.9	0.688	0.882	0.792
Total P	mg/L	12	100%	0.18	0.35	0.27	0.276	0.0449	20	100%	0.3	0.75	0.405	0.448	0.146
PO4	mg/L	11	100%	0.11	0.24	0.16	0.168	0.0424	20	100%	0.15	0.44	0.23	0.245	0.0809
Hardness	mg/L								5	100%	46	260	120	129	86.4
Total Cu	ug/L	3	100%	9.9	20	16	15.3	5.09	5	100%	11	46	30	26.8	14.4
Dissolved Cu	ug/L	3	100%	4.4	10	4.7	6.37	3.15	5	100%	4.7	15.5	7.3	9.7	4.75
Total Se	ug/L	3	100%	0.27	0.59	0.33	0.397	0.17	5	100%	0.24	0.74	0.4	0.416	0.206
Dissolved Se	ug/L	3	100%	0.26	0.56	0.27	0.363	0.17	5	100%	0.16	0.61	0.415	0.367	0.183
Carbaryl	ng/L	3	100%	12	40	19	23.7	14.6	5	80%	0	37	25.5	20.3	14.2
Fipronil	ng/L	3	33%	0	4	0	1.33	2.31	5	100%	5	14	7	9.3	4.35
ΣPAH	ng/L	2	100%	160	1350	754	754	840	2	100%	195	405	300	300	148
ΣPBDE	ng/L	2	100%	153	3360	1760	1760	2270	3	100%	18	241	170	143	114
Delta/ Tralomethrin	ng/L	3	100%	1	3.5	3.05	2.52	1.33	5	60%	0	6.2	0.3	2.16	2.9
Cypermethrin	ng/L	3	100%	2.1	4.35	3.1	3.18	1.13	5	100%	2.1	13	3.4	5.84	4.75
Cyhalothrin lambda	ng/L	3	100%	0.4	1.3	0.6	0.767	0.473	4	100%	0.5	1.9	1.5	1.35	0.619
Permethrin	ng/L	3	100%	6.4	16	13.5	12	4.98	5	100%	7.2	55	7.9	21.1	20.9
Bifenthrin	ng/L	3	100%	3.8	8.05	6.1	5.98	2.13	5	100%	3.4	8.6	5	5.82	2.57

Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation at the North Richmond Pump Station was two.

The maximum PCB concentration measured during the project study period was 38 ng/L. In WY2011, the maximum concentration measured was 82 ng/L (Hunt et al., 2012). PCB concentrations were in the range of other findings for urban locations (range 0.1-1120 ng/L) (Lent and McKee, 2011). Although highly impervious with an industrial history, the North Richmond Pump Station Watershed contains no known PCB sources of specific focus at this time; PCB transport in this watershed could be more generally representative of older mixed urban and industrial land use areas. In contrast, watersheds with known specific industrial sources appear to exhibit average concentrations in excess of about 100 ng/l (Marsalek and Ng, 1989; Hwang and Foster, 2008; Zgheib et al., 2011; Zgheib et al., 2012; McKee et al., 2012) and watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land, concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). The North Richmond Pump Station Watershed has an imperviousness of 62% and exhibits a PCB particle ratio of 267 pg/mg; the sixth highest observed so far in the Bay Area and well above the background of rural areas (indicated by Marsh Creek in the Bay Area).

Maximum total mercury concentrations (230 ng/L) during WYs 2013 and 2014 were of a similar magnitude with maximum observed concentrations during previous monitoring efforts (200 ng/L) (Hunt et al., 2012). This sample was collected during the February 26, 2014 storm event where approximately 1 inch of rain fell in the watershed. This event followed a 17 day dry period. Mercury concentrations were higher than in the range found in Zone 4 Line-A, another small urban impervious watershed (Gilbreath et al., 2012). Concentrations were also much greater than those observed in three urban Wisconsin watersheds (Hurley et al., 1995), urban influenced watersheds of the Chesapeake Bay region (Lawson et al., 2001), and two sub-watersheds of mostly urban land use in the Toronto area (Eckley and Branfirheun, 2008). Unlike, Marsh Creek, where the maximum Hg concentrations for the most part are attributed to the erosion of high masses of relatively low concentration soils, North Richmond Pump Station Watershed transports relatively low concentrations and mass of suspended sediment (maximum observed from grab samples was just 347 mg/L). Hg sources and transport in this watershed are more likely attributed to local atmospheric re-deposition from historical and ongoing oil refining and shipping and from within-watershed land use and sources. The source-release-transport processes are more likely similar to those of other urbanized and industrial watersheds (Barringer et al., 2010; Rowland et al., 2010; Lin et al., 2012) but not of very highly contaminated watersheds with direct local point source discharge (e.g. 1600-4300 ng/L: Ullrich et al., 2007; 100-5000 ng/L: Picado and Bengtsson, 2012; Kocman et al., 2012; 78-1500 ng/L: Rimondi et al., 2014).

The MeHg concentrations during the two-year study ranged from 0.03-1.1 ng/L compared with WY 2011 maximum concentrations of 0.6 ng/L (Hunt et al., 2012). Concentrations of this magnitude or greater have been observed in a number of Bay Area urban influenced watersheds (Guadalupe River: McKee et al., 2006; McKee et al., 2010; Zone 4 Line A: Gilbreath et al., 2012; Glen Echo Creek and Zone 5 Line M: McKee et al., 2012). However, concentrations of methylmercury of this magnitude have not been observed in urbanized watersheds (Mason and Sullivan, 1998; Naik and Hammerschmidt, 2011; Chalmers et al., 2014). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed, at least for agricultural and forested systems with lesser urban influences, that Hg sources are not a primary limiting factor in MeHg production (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011).

Nutrient concentrations in the North Richmond Pump Station appear to be reasonably typical of other Bay Area more rural watersheds (McKee and Krottje, 2005; Pearce et al., 2005) and compare closely to those observed in Guadalupe River during this study. North Richmond had the highest nitrate concentrations (equivalent to Guadalupe River) and orthophosphate concentrations of the six POC locations in this study. Concentrations also appear typical or slightly greater than for PO4 and TP of found in urban watersheds in other parts of the country and world (e.g. Hudak and Banks, 2006; comprehensive Australian literature review for concentrations by land use class: Bartley et al., 2012). Phosphorus concentrations appear greater here than elsewhere in the world under similar land use scenarios, an observation perhaps attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Organic carbon concentrations observed in North Richmond Pump Station were similar to those observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012) were similar to Belmont, Borel, Calabazas, and Walnut Creeks (McKee et al., 2012) and Guadalupe and East Sunnyvale Channel. They were much lower than observed in Pulgas Green Pump Station. Indeed, TOC concentrations of 4-12 mg/L have been observed elsewhere in California (Sacramento River: Sickman et al., 2007).

For pollutants sampled at a sufficient frequency for loads analysis (suspended sediments, PCBs, mercury, organic carbon, and nutrients), concentrations exhibited an unexpected pattern of median < mean except for PAH, PBDE, total copper, and hardness. This is perhaps indicative of some kind of point source for these pollutants in this watershed that is diluted during higher flows. Maximum PBDE concentrations at Richmond were 4200 ng/L which is 85-fold greater than the highest average observed in the five other locations of this current study and 50-fold greater than previously reported for Zone 4 Line A (Gilbreath et al., 2012). These are the highest PBDE concentrations measured in Bay Area stormwater to-date of any study. The North Richmond watershed currently contains an auto dismantling yard and a junk/wrecking yard; possible source areas. Only two peer reviewed articles have previously described PBDE concentrations in runoff, one for the Pearl River Delta, China (Guan et al., 2007), and the other for the San Francisco Bay (Oram et al., 2008) based, in part, on concentrations observed in Guadalupe River and Coyote Creek. Maximum total PBDE concentrations measured by Guan et al. (2007) were 68 ng/L, a somewhat surprising result given that the Pearl River Delta is a known global electronic-waste recycling hot spot. However, the Guan et al. study was based on monthly collection as opposed to storm-based sampling as was completed in a larger river system where dilution of point source may have occurred.

Copper, selenium, carbaryl, fipronil, and pyrethroids were sampled at a lesser frequency using a composite sampling design (see methods section) and were used to characterize pollutant concentrations to help support management questions possible causes of toxicity (in the case of the pesticides). Similar to the other sites, carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and

California (fipronil: 70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012; tributaries to Salton Sea, Southern CA geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008).

Pyrethroid concentrations of Delta/ Tralomethrin were similar to those observed in Zone 4 Line A, Cypermethryn was not detected in Z4LA, whereas concentrations of Permethrin and Bifenthrin were about 2-fold lower (Gilbreath et al., 2012). In summary, the statistics indicate pollutant concentrations typical of a Bay Area urban stream and there is no reason to suspect data quality issues.

8.2.4. North Richmond Pump Station toxicity

At North Richmond Pump Station, no significant effects were observed for the crustacean Ceriodaphnia dubia, the algae Selenastrum capricornutum, or fathead minnows during any tests for either year of monitoring. Two of three WY 2013 samples had a significant decrease in Hyalella Azteca survival. One sample showed an 88% survival rate compared to a 98% lab survival rate. The other sample showed a 12% survival rate compared to a 100% lab survival rate. In the five storm WY 2014 storm events, mortality of Hyalella azteca ranged from 8% to 80%.

8.2.5. North Richmond Pump Station loading estimates

The following methods were applied for calculating loading estimates (Table 15). Given that there were no flows out of the pump station when the pumps were not on, loads were only calculated for periods during active pumping conditions. Regression equations between turbidity and the particle-associated pollutants (SSC, PCBs, total mercury, methylmercury, total organic carbon and total phosphorous) were used to estimate loads (Table 16). Because there was no relation or trend in the concentrations of nitrate and phosphate in relation to flow or turbidity, flow weighted mean concentrations were applied. Monthly loading estimates correlate very well with monthly discharge (Table 16). Monthly discharge was greatest in December 2012 as were the monthly loads for suspended sediment and pollutants. Although there were slight climatic differences that have not been adjusted for, WY 2013 suspended sediment (35.7 t) and PCB (8.14 g) load estimates were comparable to the WY 2011 estimates (29 t and 8.0 g, respectively) even thought it was a wetter year (134% MAP) (Hunt., 2012) providing further support and confidence that the computed loads are reasonable. Due to lessons learned from the previous study, there is much higher confidence in the WY 2013 and 2014 loads estimates due to improvements in both the measurements of turbidity and flow rate using optical sensor equipment. Given the below average rainfall conditions experienced during WY 2013 and 2014, loads from the present study may be considered representative of somewhat dry conditions.

Table 15. Regression equations used for loads computations for North Richmond Pump Station during water year 2013 and 2014.

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Suspended Sediment (mg/L/NTU)	Mainly urban	1.31		0.58	Regression with turbidity
Total PCBs (ng/L/NTU)	Mainly urban	0.237	2.12	0.76	Regression with turbidity
Total Mercury (ng/L/NTU) WY 2013	Mainly urban	0.442		0.89	Regression with turbidity
Total Mercury (ng/L/NTU) WY 2014	Mainly urban	0.733		0.71	Regression with turbidity
Total Methylmercury (ng/L/NTU)	Mainly urban	0.0044	0.0542	0.47	Regression with turbidity
Total Organic Carbon (mg/L/NTU) WY 2013	Mainly urban	-0.0295	8.84	0.09	Regression with turbidity
Total Organic Carbon (mg/L/NTU) WY 2014	Mainly urban	0.0326	11.4	0.01	Regression with turbidity
Total Phosphorous (mg/L/NTU) WY 2013	Mainly urban	0.000754	0.241	0.34	Regression with turbidity
Total Phosphorous (mg/L/NTU) WY 2014	Mainly urban	0.00255	0.293	0.42	Regression with turbidity
Nitrate (mg/L)	Mainly urban	0.958			Flow weighted mean concentration
Phosphate (mg/L)	Mainly urban	0.206			Flow weighted mean concentration

Table 16. Monthly loads for North Richmond Pump Station. Italicized loads are estimated based on monthly rainfall-load relationships.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2013	12-Oct	33	*0.0590*	*2.36*	*604*	*0.525*	*1.28*	*0.0129*	*56*	*11.9*	*18.5*
	12-Nov	156	0.152	7.88	1167	1.75	3.48	0.0429	146	30.9	41.2
	12-Dec	232	0.374	20.8	2834	4.56	9.19	0.112	358	75.8	102
	13-Jan	18	0.0640	1.31	537	0.373	0.578	0.00923	61.4	13.0	16.2
	13-Feb	18	0.0438	1.28	358	0.324	0.564	0.00799	42.0	8.89	11.3
	13-Mar	19	0.0418	0.414	360	0.164	0.183	0.00408	40.0	8.48	10.3
	13-Apr	26	0.0602	1.72	493	0.440	0.761	0.0108	57.6	12.2	15.5
	Wet season total	502	0.795	35.7	6353	8.14	16.0	0.200	761	161	215
2014	13-Oct	0	0.0113	0.0184	129	0.0272	0.0142	0.000691	10.8	2.28	3.33
	13-Nov	36	0.0509	2.09	632	0.487	1.61	0.0119	48.7	10.3	19.0
	13-Dec	8	0.0271	0.393	319	0.129	0.304	0.00320	26.0	5.50	8.7
	14-Jan	1	0.0216	0.0739	248	0.0592	0.0571	0.00149	20.6	4.38	6.46
	14-Feb	176	0.224	9.87	2798	2.27	7.63	0.0556	214	45.4	84.8
	14-Mar	74	0.0967	5.64	1243	1.23	4.36	0.0301	92.6	19.6	39.3
	14-Apr	32	0.0676	2.31	829	0.563	1.79	0.0138	64.8	13.7	24.3
	Wet season total	326	0.499	20.4	6,197	4.76	15.77	0.1168	478	101	186

8.3. San Leandro Creek

8.3.1. San Leandro Creek flow

Rainfall at San Leandro Creek during the study was below average all three years. During WY 2012, total rainfall was 19.14 inches, or 75% of mean annual precipitation (MAP = 25.7 in) based on a long-term record at Upper San Leandro Filter (gauge number 049185) for the period 1971-2010 (WY). In WYs 2013 and 2014, rainfall totaled 17.2 and 13.3 inches, respectively, for MAPs of just 67% and 52% in each of those years. Since 1971, 2012-14 were the 14th, 11th, and 3rd driest years on record, respectively, and together had the second lowest 3-year cumulative rainfall, excepting the record dry 1975-1977 drought.

There is no historic flow record on San Leandro Creek. The challenges of developing a rating curve for this site have already been described (see “Continuous data quality assurance summary“). During WY 2012 monitoring, a preliminary rating curve was developed for stages up to 3.65 feet based on discharge sampling. This rating was augmented in WY 2014 with additional discharge measurement at wadeable stages, though gaps in the rating exist between 2 and 3.5 feet of stage as well for stages greater than

3.65 feet. As such, the estimated discharge at this site is of marginal quality. Additionally, the rainfall to runoff relationship during individual storms[4] between WY 2012 and WYs 2013-14 shifts down.

Total estimated runoff for the monitoring years was 7.3 Mm^3, 7.2 Mm^3, and 0.24 Mm^3 for WYs 2012, 2013 and 2014, respectively. This larger total annual discharge during WYs 2012 and 2013 was mostly a result of reservoir discharge from the upstream Lake Chabot, indicated by the square and sustained nature of the hydrographs during those water years, which may have been atypical[5]. Additionally, a series of relatively minor storms occurred throughout each WY (Figure 6). Flows peaked at 313 cfs in WY 2012, at 344 cfs in WY 2013, and at 152 cfs in WY 2014. San Lorenzo Creek to the south has been gauged by the USGS in the town of San Lorenzo (gauge number 11181040) from WY 1968-78 and again from WY 1988-present. Based on these records, annual peak flow has ranged between 300 cfs (1971) and 10300 cfs (1998). During WY 2012, flow peaked on San Lorenzo Creek at San Lorenzo at 2150 cfs on 1/20/2012 at 23:00; a flow that has been exceeded 54% of the years on record. During, WY 2013, flow in San significantly from 0.38 in WY 2012 to 0.22 in WY 2013 and to 0.12 in WY 2014. We cannot explain this shift, adding further uncertainty to discharge quality. Lorenzo peaked at 3080 cfs on 12/2/2012 at 11:15 am; a flow of this magnitude has been exceeded 38% of the years on record. And during WY 2014, flow peaked at 1320 cfs, a magnitude which has historically been exceeded 72% of the monitored years. Annual flow for San Lorenzo Creek at San Lorenzo (gauge number 11181040) for WY 2012 - 2014 respectively was 57%, 65% and 27% of normal. Based on this evidence alone, we suggest that storm driven flows in San Leandro Creek were likely much lower than average during this study.

8.3.2. San Leandro Creek turbidity and suspended sediment concentration

Turbidity generally responded to rainfall events in a similar manner to runoff. During the reservoir release period in the early part of WY 2012, turbidity remained relatively low indicating very little sediment was eroded from within San Leandro Creek at this magnitude and consistency of stream power. A similar phenomenon occurred in January of WY 2013 when again little rainfall occurred and relatively clean runoff devoid of sediment and pollutants was associated with the reservoir release.

Turbidity peaked at 929 NTU during a late season storm on 4/13/12 at 5:15 am. In contrast, during WY 2013, saturated watershed conditions began to occur in late November and sediment began to be released from the upper watershed much earlier in the season. A peak turbidity of 495 NTU occurred on 11/30/12 at 9:45 am. The post new year period was relatively dry and the latter season storm in April was relatively minor. Turbidity in WY 2014 was not well-characterized for a large portion of the season, but the late February through to early April period was measured with a peak of 347 NTU. These observations provide evidence that during larger storms and wetter years, the urbanized lower San

[4] Storms with flow that was augmented with reservoir release were removed from this analysis.

[5] Lake Chabot provides emergency water storage and recreation downstream of the East Bay Municipal Utility District's main Upper San Leandro Reservoir. Downstream releases are episodic and in WYs 2012 and 2013 included lake drawdowns for studies associated with preparation of the December 2013 Environmental Impact Report for planned seismic upgrades of Chabot Dam. http://www.ebmud.com/water-and-wastewater/project-updates/chabot-dam-upgrade

Leandro Creek watershed is likely capable of much greater sediment erosion and transport resulting in greater turbidity and concentrations of suspended sediment.

Suspended sediment concentration, since it was computed from the continuous turbidity data, follows the same patterns as turbidity. Suspended sediment concentration during WY 2012 peaked at 1059 mg/L during the late season storm on 4/13/12 at 5:15 am; a peak SSC of 898 mg/L occurred on 11/30/12

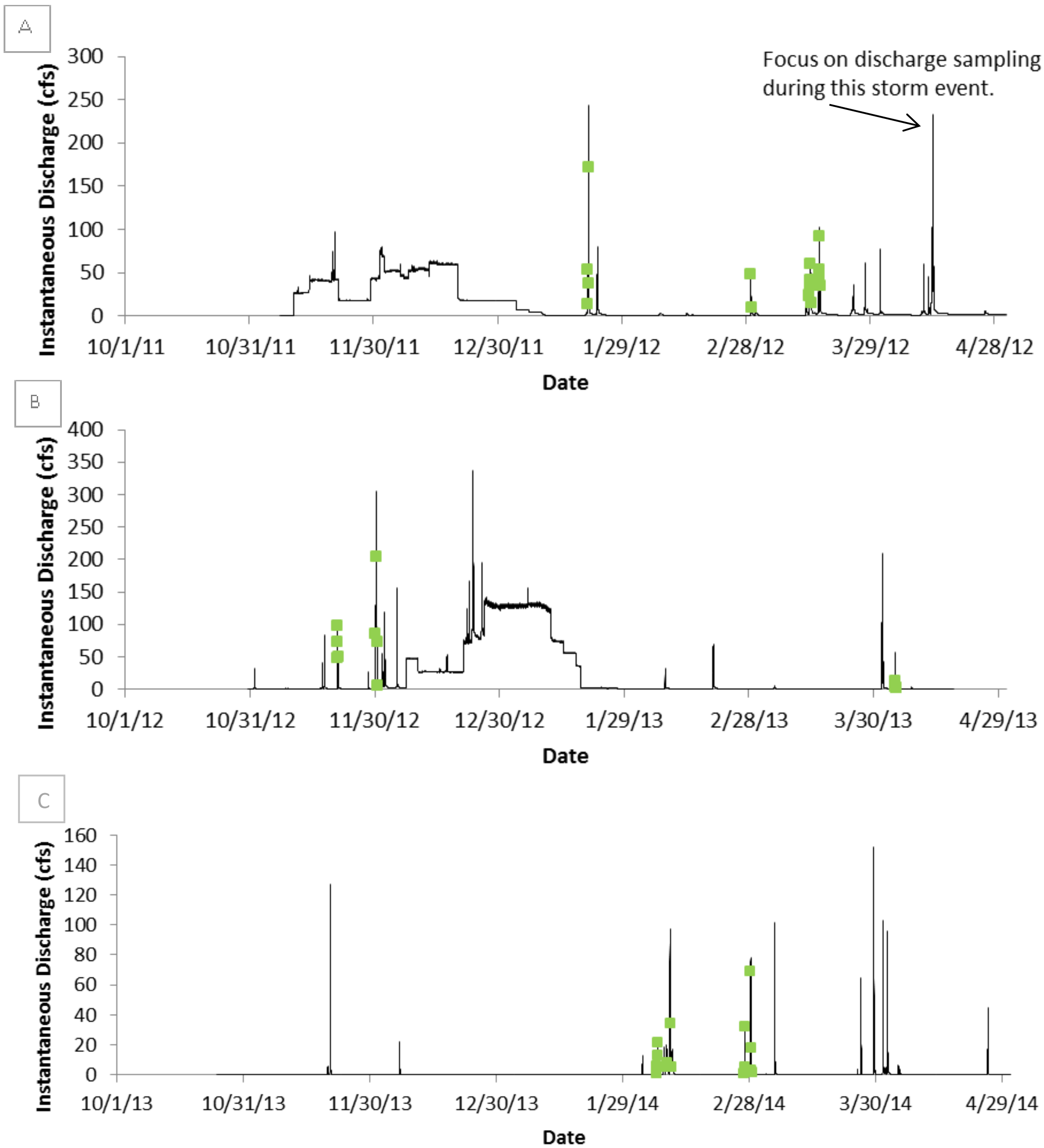

Figure 6. Flow characteristics in San Leandro Creek at San Leandro Boulevard during Water Year 2012 (A), WY 2013 (B) and WY 2014 (C) with sampling events plotted in green. Note, flow information could be updated in the future if additional discharge data are collected.

at 9:45 am for WY 2013; and a peak SSC of 337 mg/L was measured on 2/28/14 at 8:25. The maximum concentration observed during the RMP reconnaissance study (McKee et al., 2012) was 965 mg/L but at this time we have not evaluated the relative storm magnitude between WY 2011 and the current study to determine if the relative concentrations are logical.

8.3.1. San Leandro Creek POC concentrations (summary statistics)

Summary statistics of pollutant concentrations measured in San Leandro Creek during the project provide a basic understanding of general water quality and also allow a first order judgment of quality assurance (Table 17). For pollutants sampled at a sufficient frequency for loads analysis (suspended sediments, PCBs, mercury, organic carbon, and nutrients), concentrations followed the typical pattern of median < mean for most analytes.

The range of PCB concentrations (0.73-29.4 ng/L) were in the lower range of findings for urban locations (range 0.1-1120 ng/L) (Lent and McKee, 2011). PCB processes are complex in this watershed and appear to be greater in runoff derived from the urban landscape and lower in upper watershed runoff. In contrast, watersheds with known specific industrial sources appear to exhibit average concentrations in excess of about 100 ng/l (Marsalek and Ng, 1989; Hwang and Foster, 2008; Zgheib et al., 2011; Zgheib et al., 2012; McKee et al., 2012) and watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). The San Leandro Creek watershed has an average imperviousness of only 38% yet it may be an oversimplification to compare it to less urbanized watersheds since it has a very urban and impervious lower watershed. Indeed, it exhibits a particle ratio for PCBs of 101 pg/mg; the ninth highest observed so far in the Bay Area out of 24 locations and well above the background of rural areas (indicated by Marsh Creek in the Bay Area).

Maximum mercury concentrations (590 ng/L) were greater than observed in Zone 4 Line A in Hayward (Gilbreath et al., 2012) and of a similar magnitude to those observed in the San Pedro stormdrain draining an older urban residential area of San Jose (SFEI, unpublished). Concentrations were also much greater than those observed in three urban Wisconsin watersheds (Hurley et al., 1995), urban influenced watersheds of the Chesapeake Bay region (Lawson et al., 2001), and two sub-watersheds of mostly urban land use in the Toronto area (Eckley and Branfirheun, 2008). Unlike fully urban systems, San Leandro Creek appears to exhibit Hg transport processes in relation to both the erosion of soils and urban processes such as atmospheric deposition and within-watershed urban legacy Hg sources. The source-release-transport processes are not likely similar to those of very highly contaminated watersheds with direct local point source discharge (e.g. 1600-4300 ng/L: Ullrich et al., 2007; 100-5000 ng/L: Picado and Bengtsson, 2012; Kocman et al., 2012; 78-1500 ng/L: Rimondi et al., 2014).

The MeHg concentrations during the three-year study ranged from 0.1-1.48 ng/L. Concentrations of this magnitude or greater have been observed in a number of Bay Area urban influenced watersheds (Guadalupe River: McKee et al., 2006; McKee et al., 2010; Zone 4 Line A: Gilbreath et al., 2012; Glen

Table 17. Summary of laboratory measured pollutant concentrations in San Leandro Creek during water years 2012, 2013, and 2014.

		2012							2013							2014						
Analyte	Unit	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation
SSC	mg/L	53	98%	0	590	100	162	144	28	86%	0	904	48	114	202	36	97%	0	178	17.5	46.2	55.1
ΣPCB	ng/L	16	100%	2.91	29.4	10.5	12.3	8.74	12	100%	0.73	15.7	4.15	5.59	4.65	16	100%	1.6	26	2.73	5.48	6.8
Total Hg	ng/L	16	100%	11.9	577	89.4	184	203	12	100%	7.5	590	44	92.8	162	16	100%	4.9	170	17.5	37.4	44.4
Total MeHg	ng/L	9	100%	0.164	1.48	0.22	0.499	0.456	9	100%	0.15	1.4	0.2	0.377	0.397	12	100%	0.1	1	0.24	0.335	0.261
TOC	mg/L	16	100%	4.5	12.7	7.95	7.79	2.12	12	100%	4	14	5.65	6.25	2.55	16	100%	5.75	17	9.53	10.2	3.22
NO3	mg/L	16	100%	0.14	0.83	0.34	0.356	0.194	13	100%	0.13	2.8	0.235	0.546	0.758	16	100%	0.17	0.9	0.27	0.405	0.266
Total P	mg/L	16	100%	0.2	0.76	0.355	0.393	0.176	12	100%	0.0915	0.61	0.205	0.212	0.138	16	100%	0.11	0.495	0.21	0.241	0.094
PO4	mg/L	16	100%	0.057	0.16	0.0725	0.0866	0.0282	13	100%	0.069	0.13	0.0965	0.0962	0.0189	16	100%	0.073	0.17	0.115	0.117	0.0239
Hardness	mg/L	4	100%	33.8	72.5	56.5	54.8	18.5								4	100%	46	69	59	58.3	10.3
Total Cu	ug/L	4	100%	12.3	39.5	20.1	23	11.8	3	100%	5.9	28	11	15	11.6	4	100%	8	14	9.75	10.4	2.75
Dissolved Cu	ug/L	4	100%	6.04	10	8.34	8.18	1.99	3	100%	3.5	4.9	4.1	4.17	0.702	4	100%	3.8	7.2	4.8	5.15	1.47
Total Se	ug/L	4	100%	0.104	0.291	0.216	0.207	0.0885	3	100%	0.18	0.29	0.19	0.22	0.0608	4	100%	0.19	0.29	0.24	0.24	0.0476
Dissolved Se	ug/L	4	100%	0.068	0.195	0.131	0.131	0.0572	3	100%	0.16	0.19	0.17	0.173	0.0153	4	100%	0.16	0.26	0.18	0.195	0.0443
Carbaryl	ng/L	4	50%	0	14	5	6	7.12	3	0%	0	0	0	0	0	5	80%	0	18	11	10	7.44
Fipronil	ng/L	4	100%	6	10	8	8	1.63	3	67%	0	9	2	3.67	4.73	4	100%	15	19	17	17	1.83
ΣPAH	ng/L	2	100%	1530	2890	2210	2210	966	1	100%	1400	1400	1400	1400		2	100%	162	299	231	231	96.6
ΣPBDE	ng/L	2	100%	41	64.9	53	53	16.9	2	100%	1.61	29.7	15.7	15.7	19.9	1	100%	5.19	5.19	5.19	5.19	
Delta/ Tralomethrin	ng/L	3	100%	0.163	1.74	1.41	1.1	0.832	3	33%	0	0.6	0	0.2	0.346	4	0%	0	0	0	0	0
Cypermethrin	ng/L	4	0%	0	0	0	0	0	3	67%	0	0.8	0.7	0.5	0.436	4	100%	0.4	0.9	0.625	0.638	0.25
Cyhalothrin lambda	ng/L	3	33%	0	3.86	0	1.29	2.23	3	33%	0	0.3	0	0.1	0.173	3	100%	0.1	1.1	0.4	0.5	0.424
Permethrin	ng/L	4	100%	3.34	13.1	5.77	7	4.45	3	33%	0	6	0	2	3.46	4	25%	0	4.2	0.675	1.39	1.98
Bifenthrin	ng/L	4	75%	0	32.4	12.1	14.1	13.5	3	100%	2.8	7.1	5.5	5.13	2.17	4	100%	2.85	6.5	3.8	4.24	1.58

Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation at San Leandro Creek was two.

Echo Creek and Zone 5 Line M: McKee et al., 2012). However, concentrations of methylmercury of this magnitude have not been observed in urbanized watersheds from other parts of the world (Mason and Sullivan, 1998; Naik and Hammerschmidt, 2011; Chalmers et al., 2014). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed, at least for agricultural and forested systems with lesser urban influences, that Hg sources are not a primary limiting factor in MeHg production (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011).

Nutrient concentrations in the San Leandro Creek watershed appear to be reasonably typical of Bay Area more rural watersheds (McKee and Krottje, 2005; Pearce et al., 2005). Nitrate concentrations appear strikingly similar between San Leandro Creek, Lower Marsh Creek, East Sunnyvale Channel, and Pulgas Pump Station - South. In contrast, nitrate concentrations were about 2-fold greater in North Richmond and Guadalupe River. Orthophosphate concentrations were similar between San Leandro Creek and Lower Marsh Creek and 1-5-2-fold lower than the other locations in this study. Total P concentrations were similar across the six sites. Concentrations appear typical or slightly greater than for PO4 and TP of found in urban watersheds in other parts of the country and world (e.g. Hudak and Banks, 2006; comprehensive Australian literature review for concentrations by land use class: Bartley et al., 2012). Slightly higher phosphorus concentrations may perhaps be attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Organic carbon concentrations observed in San Leandro Creek (4-17 mg/L) were similar to those observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012) were similar to Belmont, Borel, Calabazas, and Walnut Creeks (McKee et al., 2012). They were much lower than observed in Pulgas Green Pump Station. TOC concentrations of 4-12 mg/L have been observed elsewhere in California (Sacramento River: Sickman et al., 2007).

A similar style of first order quality assurance is also possible for analytes measured at a lesser frequency using composite sampling design (see methods section) (copper, selenium, PAHs, carbaryl, fipronil, and PBDEs) and appropriate for water quality characterization only. The maximum concentration of PBDEs (65 ng/L) was considerably lower than the other sites with the exception of Lower Marsh Creek where observed maximum concentrations were similar. This is possibly due to differences in the randomness of the representativeness of sub-samples of the composites or due to dilution from cleaner water and sediment loads from upstream. Only two peer reviewed articles have previously described PBDE concentrations in runoff, one for the Pearl River Delta, China (Guan et al., 2007), and the other for the San Francisco Bay (Oram et al., 2008) based, in part, on concentration data from Guadalupe River and Coyote Creek. Maximum total PBDE concentrations measured by Guan et al. (2007) were 68 ng/L, a somewhat surprising result given that the Pearl River Delta is a known global electronic-waste recycling hot spot. However, the Guan et al. study was based on monthly interval collection as opposed to storm event-based sampling, and was conducted in a very large river system where dilution of point source was likely to have occurred.

Similar to the other sites, carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and California (fipronil:

70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012; tributaries to Salton Sea, Southern CA geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008). The total selenium concentrations in San Leandro Creek appear to be about half those observed in Z4LA (Gilbreath et al., 2012). Pyrethroid concentrations of Delta/ Tralomethrin and Bifenthrin were similar to those observed in Z4LA whereas concentrations of Cyhalothrin lambda and Permethrin were about 3x and 11x lower, respectively (Gilbreath et al., 2012). In summary, mercury concentrations in San Leandro are on the high end of typical Bay Area urban watersheds, whereas concentrations of other POCs are either within the range of or below those measured in other typical Bay Area urban watersheds and appear consistent with or explainable in relation to studies from elsewhere. There do not appear to be any data quality issues.

8.3.1. San Leandro Creek toxicity

Composite water samples were collected at the San Leandro Creek station during four storm events in WY 2012, three storm events during WY 2013, and four storm events during WY 2014. The survival of the freshwater fish species Pimephales promelas was significantly reduced during one of the four WY 2012 and one of the three WY 2013 events. Similar to the results for other POC monitoring stations, significant reductions in the survival of the amphipod Hyalella azteca were observed, in this case in three of the four WY 2012 storm events sampled. In WY 2014 Hyalella azteca had mortality rates ranging from 16% to 98%. No significant reductions in the survival, reproduction and growth of the crustacean Ceriodaphnia dubia or the algae Selenastrum capricornutum were observed during any of these storms.

8.3.2. San Leandro Creek loading estimates

Site specific methods were developed for computed loads (Table 18). This watershed is among the most complex in terms of data interpretation. There were challenges with missing turbidity data, a poorly defined discharge rating, a side channel coming in at the site, reservoir releases potentially including imported water, and complexities associated with urban runoff and non-urban runoff origins of runoff. Loads estimates generated for WYs 2012 and 2013 and reported by Gilbreath et al. (2014) have now been revised based on revisions to the discharge estimates, additional pollutant concentration data collected in WY 2014 and a changing understanding of pollutant transport processes for the site. Monthly loading estimates correlate well with monthly discharge (Table 19). There are no data available for October of each water year because monitoring equipment was not installed. Discharge and rainfall were not aligned due to reservoir release. Monthly discharge was greatest in January 2013 when large releases were occurring from the upstream reservoir following the large storm period of November and December 2012. The greatest monthly loads for each of the pollutants regardless of transport mode (dominantly particulate or dissolved) occurred in December 2012 when rainfall induced run-off caused high turbidity and elevated concentrations of suspended sediments and pollutants. The sediment and pollutant loads were less well correlated with the total discharge than for other sampling sites due to reservoir releases and complex sources. When discharge was dominated by upstream flows induced by rainfall, relatively high loads of mercury occurred; conversely, PCB loads were greater relative to rainfall during smaller rainfall events when less runoff occurred from the upper watershed. Given the very dry flow conditions of WY 2012, 2013, and 2014 (see discussion on flow above), loads presented here may be considered representative of dry conditions. Any future sampling should be focus on larger rain storms during wetter years and improving the discharge rating for the site.

Table 18. Regression equations used for loads computations for San Leandro Creek during water years 2012-14.

Analyte	Origin of runoff	Slope	Inter-cept	Correlation coefficient (r^2)	Notes
Suspended Sediment (mg/L/NTU)	Mainly urban	1.35		0.9	Regression with turbidity
Suspended Sediment (mg/L/NTU)	Mainly non-urban	1.14		0.82	Regression with turbidity
Suspended Sediment (mg/L)	Mainly baseflow	3.39			Avg of 4/4/13 storm (all collected at base flow) and low flow samples
Suspended Sediment (mg/L/CFS)	Mixed	2.66		0.76	Regression with Flow
Total PCBs (ng/L/NTU)	Mainly urban	0.0935	3.95	0.58	Regression with turbidity
Total PCBs (ng/L/NTU)	Mainly non-urban	0.0322	0.957	0.87	Regression with turbidity
Total PCBs (ng/L)	Mainly baseflow	1.32			Avg of 4/4/13 storm (all collected at base flow)
Total PCBs (ng/L/CFS)	Mixed	0.121	2.89	0.54	Regression with Flow
Total Mercury (ng/L/NTU)	Mixed	1.13		0.79	Regression with turbidity
Total Mercury (ng/L)	Mainly baseflow	8.9			Avg of 4/4/13 storm (all collected at base flow)
Total Mercury (ng/L/CFS)	Mainly urban	1.8		0.67	Regression with Flow
Total Mercury (ng/L/CFS)	Mainly non-urban	3.13	44.1	0.43	Regression with Flow
Total Methylmercury (ng/L/NTU)	Mixed	0.00257	0.147	0.81	Regression with turbidity
Total Methylmercury (ng/L)	Mainly baseflow	0.217			Avg of 4/4/13 storm (all collected at base flow) and low flow samples
Total Methylmercury (ng/L/CFS)	Mainly urban	0.00225	0.171	0.14	Regression with Flow
Total Methylmercury (ng/L/CFS)	Mainly non-urban	0.00988	0.27	0.81	Regression with Flow
Total Organic Carbon (mg/L)	Mixed	7.28			Flow weighted mean concentration
Total Organic Carbon (mg/L)	Mainly baseflow	5.3625			Avg of 4/4/13 storm (all collected at base flow)
Total Phosphorous (mg/L/NTU)	Mixed	0.00128	0.158	0.67	Regression with turbidity
Total Phosphorous (mg/L)	Mainly baseflow	0.105			Avg of 4/4/13 storm (all collected at base flow)
Total Phosphorous (mg/L/CFS)	Mixed	0.00252	0.188	0.45	Regression with Flow
Nitrate (mg/L)	Mixed	0.384			Flow weighted mean concentration
Nitrate (mg/L)	Mainly baseflow	0.26			Avg of 4/4/13 storm (all collected at base flow)
Phosphate (mg/L)	Mixed	0.0932			Flow weighted mean concentration
Phosphate (mg/L)	Mainly baseflow	0.0768			Avg of 4/4/13 storm (all collected at base flow)

Table 19. Monthly loads for San Leandro Creek for water years 2012-14. Italicized loads are estimated based on monthly rainfall-load relationships.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2012	11-Oct	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*	*0*
	11-Nov	*3*	*0.00067*	*0.028*	*5.0*	*0.0045*	*0.042*	*0.0045*	*0.24*	*0.061*	*0.15*
	11-Dec	0	4.67	39.6	25,026	6.16	37.6	0.771	1,213	358	780
	12-Jan	73	0.845	34.1	4,959	3.24	34.7	0.192	244	68.4	170
	12-Feb	22	0.101	2.69	621	0.271	2.56	0.0217	30.5	8.20	17.4
	12-Mar	151	0.734	58.7	4,393	2.59	54.5	0.233	213	59.4	182
	12-Apr	85	0.956	96.7	5,484	4.12	92	0.349	272	76.5	255
	Wet season total	334	7.30	232	40,488	16.4	221	1.57	1,974	571	1,405
2013	12-Oct	*25*	*0.035*	*2.34*	*244*	*0.20*	*2.5*	*0.053*	*13*	*3.2*	*8.5*
	12-Nov	121	0.198	38.5	1,263	1.59	29.7	0.105	110	20.6	57.9
	12-Dec	127	3.29	127	23,951	7.92	124	0.796	1,263	307	621
	13-Jan	7	3.63	54.7	26,430	4.95	51.9	0.652	1,394	338	632
	13-Feb	19	0.0290	1.33	211	0.109	1.26	0.00712	11.1	2.70	6.00
	13-Mar	11	0.00752	0.702	54.7	0.0758	0.666	0.00262	2.89	0.701	1.94
	13-Apr[a]	41	0.0505	5.69	364	0.346	5.41	0.0197	19.1	4.68	14.0
	Wet season total	351	7.24	230	52,517	15.2	215	1.64	2,813	677	1,342
2014	13-Oct	*16*	*0.015*	*0.91*	*107*	*0.088*	*1.1*	*0.031*	*5.5*	*1.4*	*3.6*
	13-Nov	24	0.0276	5.11	199	0.311	5.68	0.908	10.4	2.55	10.0
	13-Dec	8	0.00350	0.104	24.9	0.0146	0.203	0.0880	1.30	0	0.746
	14-Jan	1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	14-Feb	93	0.103	8.46	839	0.803	6.65	1.52	44.6	10.6	28.2
	14-Mar	78	0.0756	9.95	543	0.586	9.45	0.0326	28.6	6.98	22.6
	14-Apr	36	0.0332	3.33	234	0.212	3.41	0.340	12.2	3.03	9.02
	Wet season total	256	0.258	27.9	1,946	2.02	26.5	2.92	103	24.8	74.3

[a] April 2013 monthly loads are reported for only the period April 01-18. In the 12 days missing from the record, no rain fell in the San Leandro Creek watershed.

8.4. Guadalupe River

8.4.1. Guadalupe River flow

The US Geological Survey has maintained a flow record on lower Guadalupe River (gauge number 11169000; 11169025) since October 1, 1930 (83 WYs; note 1931 is missing). Peak annual flows for the period have ranged between 125 cfs (WY 1960) and 11000 cfs (WY 1995). Annual runoff from Guadalupe River has ranged between 0.422 (WY 1933) and 241 Mm^3 (WY 1983).

During WY 2012, a series of relatively minor storms[6] occurred (7). A storm that caused flow to escape the low flow channel and inundate the in-channel bars did not occur until 1/21/12, very late in the season compared to what has generally occurred over the past years of sampling and analysis for this system (McKee et al., 2004; McKee et al., 2005; McKee et al., 2006; McKee et al., 2010; Owens et al., 2011). The flow during this January storm was 1220 cfs; flows of this magnitude are common in most years. Flow peaked in WY 2012 at 1290 cfs on 4/13/2012 at 07:15 and total runoff during WY 2012 based on USGS data was 38.0 Mm^3; discharge of this magnitude is about 85% mean annual runoff (MAR) based on 83 years of record and 68% MAR if we consider the period WY1971-2010 (perhaps more representative of current climatic conditions given climate change). Rainfall data corroborates this assertion; rainfall during WY 2012 was 7.09 inches, or 49% of mean annual precipitation (MAP = 15.07 in) based on a long-term record at San Jose (NOAA gauge No: 047821) for the period 1971-2010 (CY). CY 2012 was the driest year in the past 42 years and the 7th driest for the 138 year record beginning 1875.

Water year 2013 was only slightly wetter, raining 9.43 inches at the San Jose gauge (65% MAP for the period 1971-2010 [CY]). Three moderate sized storms occurred in late November and December which led to three peak flows above 1500 cfs within a span of one month (Figure 7). Flow peaked on the third of these storms at 3160 cfs on 12/23/12 at 18:45, a peak flow which has been exceeded in half of all years monitored (83 years). Total runoff during WY 2013 based on USGS data was 45.8 Mm^3; discharge of this magnitude is about 82% MAR based on 83 years of record and equivalent to the MAR for the period WY1971-2010.

Water year 2014 was drier than the two previous, raining only 6.32 inches (43 % MAP for the period 1971-2010 [CY]). One moderately sized storm occurred in late February 2014, but otherwise only minor storms occurred during the year. Flow peaked on February 28th, 2014 at 07:30 at 2310 cfs, which has historically been exceeded in 59% of all monitored years. Total flow for the water year has not been published by the USGS[7]. However, when just comparing the October-April time period for each water year monitored in this study, WY 2014 was less than the previous two at only 16.7 Mm^3 compared with 25.8 and 35.5 Mm^3 for WYs 2012 and 2013, respectively.

---

[6] A storm was defined as rainfall that resulted in flow that exceeds bankfull, which, at this location, is 200 cfs, and is separated by non-storm flow for a minimum of two days.

[7] The USGS normally publishes finalized data for the permanent record in the spring following the end of each Water Year.

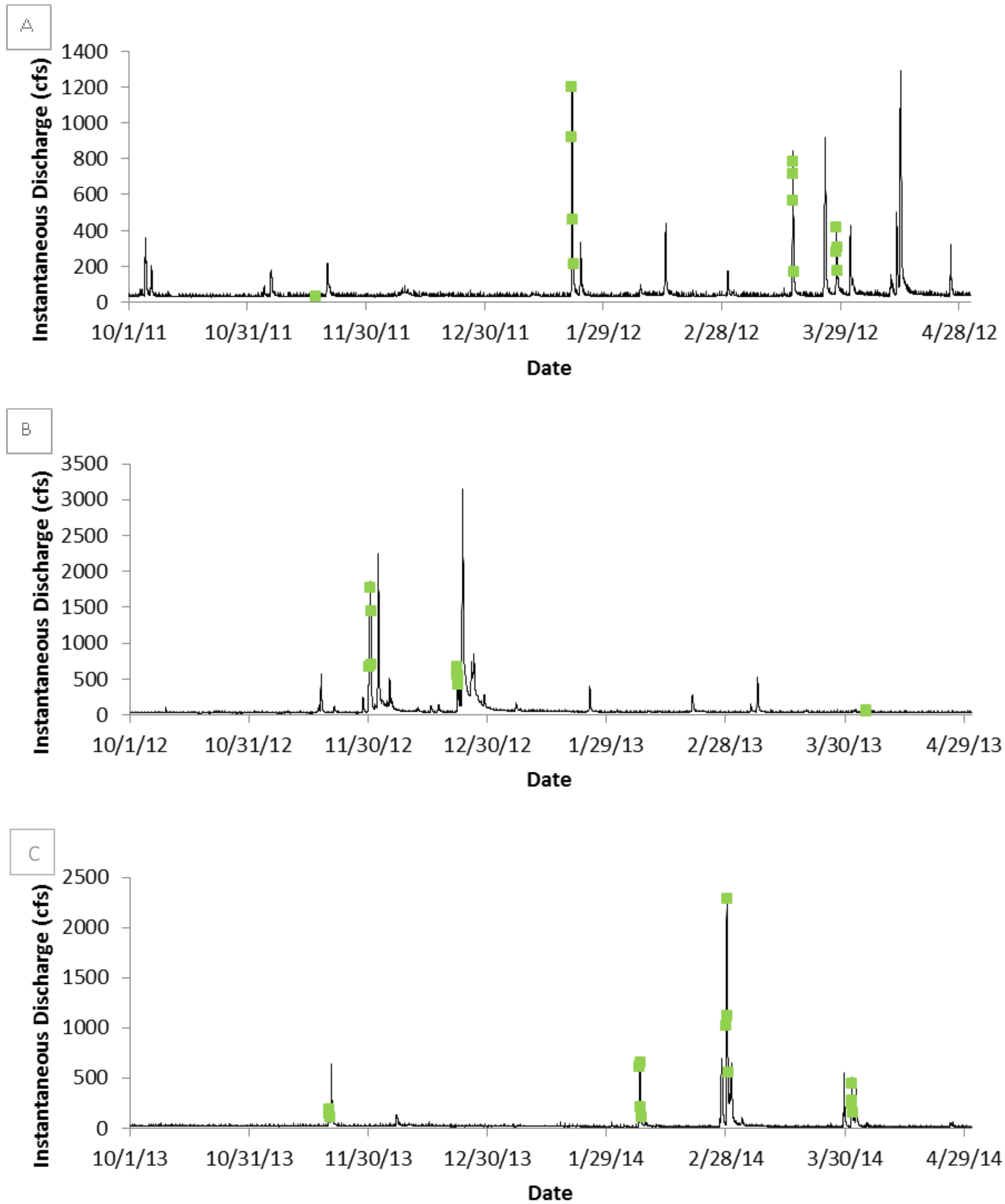

Figure 7. Flow characteristics in Guadalupe River during water year 2012 (A), 2013 (B) and 2014 (C) based on published 15-minute data provided by the USGS (gauge number 11169025), with sampling events plotted in green. The fuzzy nature of the low flow data are caused by baseflow discharge fluctuations likely caused by pump station discharges near the gauge.

8.4.2. Guadalupe River turbidity and suspended sediment concentration

The US Geological Survey also maintains the turbidity sensor at this location. Turbidity generally responded to rainfall events in a similar manner to runoff. Generally, peak turbidities fluctuated throughout the storm season between 150 – 600 FNU for each storm. Based on past years of record, turbidity can exceed 1000 FNU at the sampling location (e.g. McKee et al., 2004), so these monitored years produced turbidity conditions that were generally much lower than the system is capable of. In WY 2012, Guadalupe River exhibited a pronounced first flush during a very minor early season storm when, relative to flow, turbidity was elevated and reached 260 FNU. In contrast, the storm that produced the greatest flow for the season that occurred on 4/13/2012 had lower peak turbidity (185 FNU). A similar pattern occurred in WY 2013, except that the third large storm event on 12/23/12 raised turbidity to its peak for the season (551 FNU). Peak turbidity for WY 2012 was 388 FNU during a storm on 1/21/12 at 3:15 am. Despite higher peak flow in WY 2014 than 2012, turbidity peak only reached 273 FNU during in the intense first flush on 11/20/13 at 15:45.

Based on USGS sampling in Guadalupe River in past years, >90% of particles in this system are <62.5 µm in size (e.g. McKee et al., 2004). Because of these consistently fine particle sizes, turbidity correlates well with the concentrations of suspended sediments and hydrophobic pollutants (e.g. McKee et al., 2004). Suspended sediment concentration was computed by USGS from the continuous turbidity data. Daily and monthly loads are reported by USGS and were used in this report.

8.4.3. Guadalupe River POC concentrations (summary statistics)

A summary of concentrations is useful for providing comparisons to other systems and also for doing a first order quality assurance check. Concentrations measured in Guadalupe River during the project are summarized (Table 20). Guadalupe River is unique among the sampling location in that it has been sampled for POCs on and off since November 2002. The results from previous work (McKee et al., 2004; McKee et al., 2005; McKee et al., 2006; McKee et al., 2010; Owens et al., 2011) are not included in the summary statistics provided here. The interested reader will need to refer to those reports

The range of PCB concentrations are typical of mixed urban land use watersheds (Lent and McKee, 2011) and mean concentrations in this watershed were the 3rd highest measured of the six locations (Pulgas PS - South > Sunnyvale Channel > Guadalupe River = North Richmond PS > San Leandro Creek >Lower Marsh Creek). However, maximum concentrations measured in Guadalupe River in the past were ~2-fold greater (e.g. McKee et al., 2006). PCB processes are complex in this watershed and are known to be greater in runoff derived from the urban landscape and lower in runoff derived from the upper less urban watershed (McKee et al., 2006). Concentrations in Guadalupe River watershed at the Hwy 101 sampling location appear to be similar to watersheds with industrial sources where concentrations in excess of about 100 ng/L are common (Marsalek and Ng, 1989; Hwang and Foster, 2008; Zgheib et al., 2011; Zgheib et al., 2012; McKee et al., 2012). In contrast, watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). The Guadalupe River watershed has an imperviousness of 39% and exhibits a particle ratio of 84 pg/mg (based on all sampling to-date including previous studies); the 10th highest observed so far in the Bay Area out of 24 locations and well above the background of rural areas (indicated by Marsh Creek in the Bay Area).

Maximum mercury concentrations (1000 ng/L measured in WY 2012) are greater than observed in Z4LA (Gilbreath et al., 2012) and the San Pedro stormdrain (SFEI unpublished data), which drains an older urban residential area of San Jose. This maximum concentration was higher than the average mercury concentration (690 ng/L) but much less than the maximum concentration (~18,700 ng/L) observed over the period of record at this location (2002-2010) (McKee et al., 2010). Concentrations were orders of magnitude greater than those observed in three urban Wisconsin watersheds (Hurley et al., 1995), urban influenced watersheds of the Chesapeake Bay region (Lawson et al., 2001), and two sub-watersheds of mostly urban land use in the Toronto area (Eckley and Branfirheun, 2008). The concentrations in Guadalupe River are similar to those of very highly contaminated watersheds with direct local point source discharge or mining influences (e.g. 1600-4300 ng/L: Ullrich et al., 2007; 100-5000 ng/L: Picado and Bengtsson, 2012; Kocman et al., 2012; 78-1500 ng/L: Rimondi et al., 2014).

The MeHg concentrations during the three-year study ranged from 0.04-1.2 ng/L and were lower than maximum concentrations (2.51 ng/L) observed previously for this sampling location (McKee et al., 2010). Concentrations of this magnitude or greater have been observed in a number of Bay Area urban influenced watersheds (Zone 4 Line A: Gilbreath et al., 2012; Glen Echo Creek and Zone 5 Line M: McKee et al., 2012). However, concentrations of methylmercury of this magnitude have not been observed in urbanized watersheds from other parts of the world (Mason and Sullivan, 1998; Naik and Hammerschmidt, 2011; Chalmers et al., 2014). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed, at least for agricultural and forested systems with lesser urban influences, that Hg sources are not a primary limiting factor in MeHg production (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011). Based on previous sampling experience in the system (McKee et al., 2004; McKee et al., 2005; McKee et al., 2006; McKee et al., 2010; Owens et al., 2011) and these simple comparisons to other studies, there are no reasons to suspect any data quality issues.

Nutrient concentrations were in the same range as measured in in Z4LA (Gilbreath et al., 2012), and typical for the Bay Area, phosphorus concentrations appear greater than elsewhere in the world under similar land use scenarios, perhaps attributable to geological sources (McKee and Krottje, 2005). Nitrate concentrations were highest in Guadalupe River and North Richmond pump station during this study. Nitrate concentrations appear similar between San Leandro Creek, Lower Marsh Creek, East Sunnyvale Channel, and Pulgas Pump Station - South. In contrast, nitrate concentrations were about 2-fold greater in Guadalupe River and North Richmond Pump Station. Mean orthophosphate concentrations (0.15 mg/L) were slightly lower than observed in the Richmond Pump Station but 20-50% above the other four sample sites. The maximum total P concentration (1 mg/L) was very high in this study relative to the other watersheds; however, average total P concentrations were similar across the six sites. Concentrations appear typical or slightly greater than for PO4 and total P found in urban watersheds in other parts of the country and world (e.g. Hudak and Banks, 2006; comprehensive Australian literature review for concentrations by land use class: Bartley et al., 2012). These elevated phosphorus concentrations, especially the peak concentration observed in Guadalupe River, may perhaps be attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Table 20. Summary of laboratory measured pollutant concentrations in Guadalupe River for water years 2012, 2013, and 2014.

		2012							2013							2014						
Analyte	Unit	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation
SSC	mg/L	41	100%	8.6	730	82	198	205	41	100%	5.9	342	128	124	104	54	100%	5.8	358	110	150	102
ΣPCB	ng/L	11	100%	2.7	59.1	7.17	17.7	21.5	12	100%	2.04	47.4	6.29	10.6	12.7	16	100%	3.1	33.1	11.4	14.6	11.1
Total Hg	ng/L	12	100%	36.6	1000	125	268	324	12	100%	14.5	360	155	153	119	15	100%	45	740	130	215	193
Total MeHg	ng/L	10	100%	0.086	1.15	0.381	0.445	0.352	7	100%	0.04	0.94	0.49	0.428	0.34	10	100%	0.09	1.2	0.575	0.616	0.366
TOC	mg/L	12	100%	4.9	18	7.45	8.73	4.03	12	100%	5.3	11	6.05	6.36	1.55	16	100%	5.3	56	12.1	19.3	17.1
NO3	mg/L	12	100%	0.56	1.9	0.815	0.917	0.38	8	100%	0.45	2.3	1.43	1.38	0.905	16	100%	0.32	1.8	0.54	0.685	0.403
Total P	mg/L	12	100%	0.19	0.81	0.315	0.453	0.247	12	100%	0.098	0.61	0.355	0.31	0.159	16	100%	0.11	1	0.485	0.464	0.268
PO4	mg/L	12	100%	0.06	0.16	0.101	0.101	0.0321	12	100%	0.061	0.18	0.12	0.109	0.0339	16	100%	0.11	0.5	0.17	0.218	0.125
Hardness	mg/L	3	100%	133	157	140	143	12.3								4	100%	94	200	120	134	46
Total Cu	ug/L	3	100%	10.7	26.3	24.7	20.6	8.58	3	100%	5.9	28	23	19	11.6	4	100%	12	34	25.5	24.3	9.54
Dissolved Cu	ug/L	3	100%	5.07	7.91	5.51	6.16	1.53	3	100%	2.5	3.6	2.5	2.87	0.635	4	100%	2.9	12	4	5.72	4.24
Total Se	ug/L	3	100%	1.16	1.63	1.21	1.33	0.258	3	100%	0.7	3.3	0.78	1.59	1.48	4	100%	0.6	1.8	0.98	1.09	0.506
Dissolved Se	ug/L	3	100%	0.772	1.32	1.04	1.04	0.274	3	100%	0.4	3.2	0.54	1.38	1.58	4	100%	0.34	1.5	0.775	0.847	0.502
Carbaryl	ng/L	3	100%	13	57	54.3	41.4	24.7	3	67%	0	21	17	12.7	11.2	4	100%	12	64	28.5	33.3	21.9
Fipronil	ng/L	3	100%	6.5	20	11	12.5	6.87	3	100%	3	11	9	7.67	4.16	4	100%	8	15	14.5	13	3.37
ΣPAH	ng/L	1	100%	611	611	611	611		8	100%	40.7	736	174	251	245	2	100%	692	1260	978	978	405
ΣPBDE	ng/L	1	100%	23	23	23	23		2	100%	13.1	69.8	41.4	41.4	40.1	2	100%	96.7	101	99	99	3.18
Delta/ Tralomethrin	ng/L	3	100%	0.704	1.9	1.81	1.47	0.667	3	0%	0	0	0	0	0	4	50%	0	2.8	0.65	1.02	1.33
Cypermethrin	ng/L	3	0%	0	0	0	0		3	100%	0.5	3.3	1.7	1.83	1.4	4	100%	1.1	5	1.65	2.35	1.8
Cyhalothrin lambda	ng/L	3	33%	0	0.6	0	0.2	0.346	3	100%	0.3	1.5	0.5	0.767	0.643	4	75%	0	1.46	0.6	0.665	0.606
Permethrin	ng/L	3	100%	16.8	20.5	19.5	18.9	1.91	3	33%	0	5.4	0	1.8	3.12	4	100%	7.2	14	10.6	10.6	3
Bifenthrin	ng/L	3	67%	0	13.3	6.16	6.47	6.63	3	100%	0.9	7.6	5.9	4.8	3.48	4	100%	3.5	6.1	4.75	4.78	1.47

Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation at Guadalupe River was two.

Organic carbon concentrations observed in Guadalupe River during WYs 2012-2014 (4-56 mg/L) were higher than those observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012). They were greater than but more similar to maximum concentrations observed in East Sunnyvale Channel (30 mg/L) but less than Pulgas Pump Station - South (140 mg/L). Although we have not done an extensive literature review of TOC concentrations in the worlds river systems, our general knowledge of the literature would have us hypothesize that concentrations of these magnitudes are very high. These may be contributing to the apparent high methylation rates in the Bay Area.

A similar style of first order quality assurance is also possible for analytes measured at a lesser frequency using composite sampling design (see methods section) (copper, selenium, PAHs, carbaryl, fipronil, and PBDEs) and appropriate for water quality characterization only. The maximum concentration of PBDEs (101.2 ng/L) was similar to East Sunnyvale Channel, lesser by 15-fold than North Richmond Pump Station and greater by about 2-fold than the other locations. Only two peer reviewed articles describing PBDE concentrations in runoff have been located, one for the Pearl River Delta, China (Guan et al., 2007), and the other for the San Francisco Bay (Oram et al., 2008) based, in part, on concentration data from Guadalupe River and Coyote Creek taken during WYs 2003-2006. Maximum total PBDE concentrations measured by Guan et al. (2007) were 68 ng/L, a somewhat surprising result given that the Pearl River Delta is a known global electronic-waste recycling hot spot. However, the Guan et al. study was based on monthly interval collection as opposed to storm event-based sampling and was completed in a larger river system where dilution of point source may have occurred.

Copper, which was sampled at a lesser frequency for characterization only, was similar to concentrations previously observed (McKee et al., 2004; McKee et al., 2005; McKee et al., 2006) and similar to those observed in Z4LA (Gilbreath et al., 2012). Maximum selenium concentrations were generally 2-10 fold greater than the other five locations and were generally higher than Z4LA; elevated groundwater concentrations have been observed in Santa Clara County previously (Anderson, 1998). Similar to the other sites, carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and California (fipronil: 70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012; tributaries to Salton Sea, Southern CA geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008). Pyrethroid concentrations of Delta/ Tralomethrin and Bifenthrin were about 2.5-fold less than those observed in Z4LA whereas concentrations of Cyhalothrin lambda and Permethrin were about 8-fold lower (Gilbreath et al., 2012). In summary, mercury concentrations are elevated in the Guadalupe Giver relative to typical Bay Area and other urban watersheds and are more akin to concentrations observed in mining and point source contaminated systems. Concentrations of other POCs are either within the range of or below those measured in other typical Bay Area urban watersheds and appear consistent with or explainable in relation to studies from elsewhere. There do not appear to be any data quality issues.

8.4.4. Guadalupe River toxicity

Composite water samples were collected at the Guadalupe River station during three storm events in WY 2012, three storm events in WY 2013, and four storm events in WY 2014. Similar to the results for other POC monitoring stations, no significant reductions in the survival, reproduction and growth of three of four test species were observed during storms except for fathead minnow growth reductions in two WY 2014 samples and a reduction in fathead minnow survival in one WY 2014 sample. Significant reductions in the survival of the amphipod *Hyalella azteca* were observed during two of the three WY 2012 events sampled and three of the four WY 2014 samples.

8.4.5. Guadalupe River loading estimates

The following methods were applied to estimate loads for the Guadalupe River in WYs 2012, 2013, and 2014. Suspended sediment loads for WYs 2012-2014 were downloaded from USGS. Concentrations during storm flows were estimated using regression equations between the POCs and turbidity, except for nitrate and phosphate, in which a flow-surrogate regression was used (Table 21). As found during other drier periods (McKee et al., 2006), a separation of the data for PCBs to form regression relations based on origin of flow was possible. On the other hand, there was virtually no mining runoff during these very dry years and although a separation was made for Hg in addition to PCBs, very few data points populated the regression between Hg and turbidity for the upper watershed as the source of flow.

Monthly discharge was greatest in December 2012 as were loads of most pollutants. This single wet month transported approximately 50% of the PCB and mercury load of the two wet seasons combined. WY 2013 loads were approximately 3-fold higher than WY 2012 and 4-fold greater than WY 2014. However, compared to previous sampling years (McKee et al., 2004; McKee et al., 2005; McKee et al., 2006; McKee et al., 2010; Owens et al., 2011 [Hg only]), loads of total mercury and PCBs were lower than any previously observed years (Table 22). At this time, all loads estimates for WY 2014 should be considered preliminary. Once available, USGS official records for flow, turbidity, and SSC can be substituted for the preliminary data presented here. Overall, WY 2012, 2013, and 2014 loads may be considered representative of loads during dry conditions in this watershed.

Table 21. Regression equations used for loads computations for Guadalupe River during water year 2012, 2013, and 2014.

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Total PCBs (ng/L/NTU)	Mainly urban	0.236	1.42	0.71	Regression with turbidity
Total PCBs (ng/L/NTU)	Mainly non-urban & baseflow	0.081		0.81	Regression with turbidity
Total Mercury (ng/L/NTU)	Mixed	2.21		0.82	Regression with turbidity
Total Methylmercury (ng/L/NTU)	Mixed	0.00352	0.181	0.6	Regression with turbidity
Total Methylmercury (ng/L)	Baseflow	0.0994			Average
Total Organic Carbon (mg/L/NTU)	Mixed	0.0245	4.9715	0.49	Regression with turbidity
Total Phosphorous (mg/L/NTU)	Mixed	0.00213	0.153	0.72	Regression with turbidity

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Nitrate (mg/L/CFS)	Mainly urban	-0.00133	1.99	0.64	Regression with flow
Nitrate (mg/L/CFS)	Mainly non-urban & baseflow	-0.000161	0.732	0.17	Regression with flow
Phosphate (mg/L/CFS)	Mixed	0.0000336	0.0906	0.36	Regression with flow

Table 22. Monthly loads for Guadalupe River for water year 2012, 2013 and 2014.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2012	11-Oct	19	2.91	167	16,565	9.63	190	0.556	2449	270	628
	11-Nov	15	2.88	104	15,552	6.01	111	0.441	2300	266	548
	11-Dec	1	2.73	76.3	14,016	1.42	38.7	0.272	1984	251	455
	12-Jan	18	3.85	564	28,348	30.9	570	1.33	3077	396	1128
	12-Feb	14	3.15	305	18,361	10.4	243	0.613	2451	294	716
	12-Mar	50	5.08	403	30,542	35.1	433	1.50	4238	495	1314
	12-Apr	44	5.22	486	30,994	29.8	452	1.41	4381	527	1235
	Wet season total	161	25.8	2,106	154,379	123	2,039	6.13	20,879	2,498	6,023
2013	12-Oct	8	2.26	60.5	11,988	3.67	68.5	0.258	1810	207	411
	12-Nov	48	5.23	1092	38,487	53.1	999	2.68	4148	592	1862
	12-Dec	92	14.8	2768	117,823	230	4034	8.90	9301	1745	6174
	13-Jan	15	4.14	204	21,988	8.35	129	0.58	3237	385	756
	13-Feb	11	3.05	85.7	15,999	4.69	76.3	0.398	2355	282	539
	13-Mar	21	3.47	123	18,604	7.45	122	0.546	2837	325	648
	13-Apr	5	2.57	130	13,319	2.37	47.7	0.279	2087	235	439
	Wet season total	201	35.5	4,464	238,208	309	5,476	13.6	25,775	3,771	10,829
2014	13-Oct	0	1.72	81.5	8,902	1.22	33.2	0.171	1250	157	294
	13-Nov	21	2.25	191	17,545	16.2	169	0.510	2021	246	551
	13-Dec	4	1.96	79.1	10,106	2.23	32.2	0.225	1582	180	331
	14-Jan	3	1.53	69.4	7,837	0.748	20.4	0.152	1115	140	254
	14-Feb	64	4.55	927	34,750	57.6	1009	2.28	3076	538	1797
	14-Mar	35	3.07	217	17,982	13.7	188	0.673	2571	306	627
	14-Apr	17	1.67	103	9,020	5.51	66.2	0.274	1566	156	319
	Wet season total	144	16.7	1,668	106,141	97.2	1,519	4.29	13,182	1,723	4,172

8.5. East Sunnyvale Channel

8.5.1. East Sunnyvale Channel flow

Santa Clara Valley Water District (SCVWD) has maintained a flow gauge on East Sunnyvale Channel from WY 1983 to present. Unfortunately, the record is known to be of poor quality (pers. comm., Ken Stumpf, SCVWD), which was apparent when the record was regressed against rainfall (R^2 = 0.58) (Lent et al., 2012). The gauge is presently scheduled for improvement by SCVWD. Despite the poor historical flow record, velocity measurement conducted in WY 2013 confirmed the good quality of the SCVWD discharge-rating curve up to stages of 2.9 ft (corresponding to flows of 190 cfs) for this site. Consequently, flow could be calculated using that curve and the continuous stage record collected during this study.

All three monitored water years were relatively dry years and discharge was likely lower than average. Rainfall during WYs 2012-2014 was 8.82, 12.1 and 8.1 inches, respectively, at Palo Alto (NOAA gauge number 046646). Relative to mean annual precipitation (MAP = 15.5 in) based on a long-term record for the period 1971-2010 (CY), WY 2012 was only 57% MAP, WY 2013 was 78% MAP, and WY 2014 was 52% MAP.

A series of relatively minor storms occurred during WY 2012 (Figure 8). Flow peaked at 492 cfs overnight on 4/12/12- 4/13/12 at midnight. Total runoff during WY 2012 for the period 12/1/11 to 4/30/12 was 1.07 Mm^3 based on our stage record and the SCVWD rating curve. Total annual runoff WY 2013 for the period between 10/01/12 and 4/30/13 was 1.51 Mm^3 and likely below average based on below average rainfall. However, unlike WY 2012 in which the rainfall was spread over several smaller events, the majority of WY 2013 rainfall occurred during three large storm events in late November and December, each of which was of 1-2 year recurrence based on NOAA Atlas 14 partial duration series data for the area. Flow peaked during the third event of this series at 727 cfs on 12/23/12 at 15:15. Given that SCVWD maintains the channel to support a peak discharge of 800 cfs, the December 2012 storms resulted in significant flows for the system. Field observations during sampling of the early December storms corroborate this assertion; stages neared the top of bank and the banks of the channel for the observable reach at and upstream from the sampling location showed evidence of erosion. This is yet another vivid example of why peak discharge often correlates with total wet season load better than total wet season flow (Lewicki and McKee, 2009). The WY 2014 wet season was very similar to the WY 2012 season, both in terms of total annual flow (1.01 Mm3) as well as the relative size of the storms, peaking at 439 cfs on February 28th, 2014 at 3:45 am.

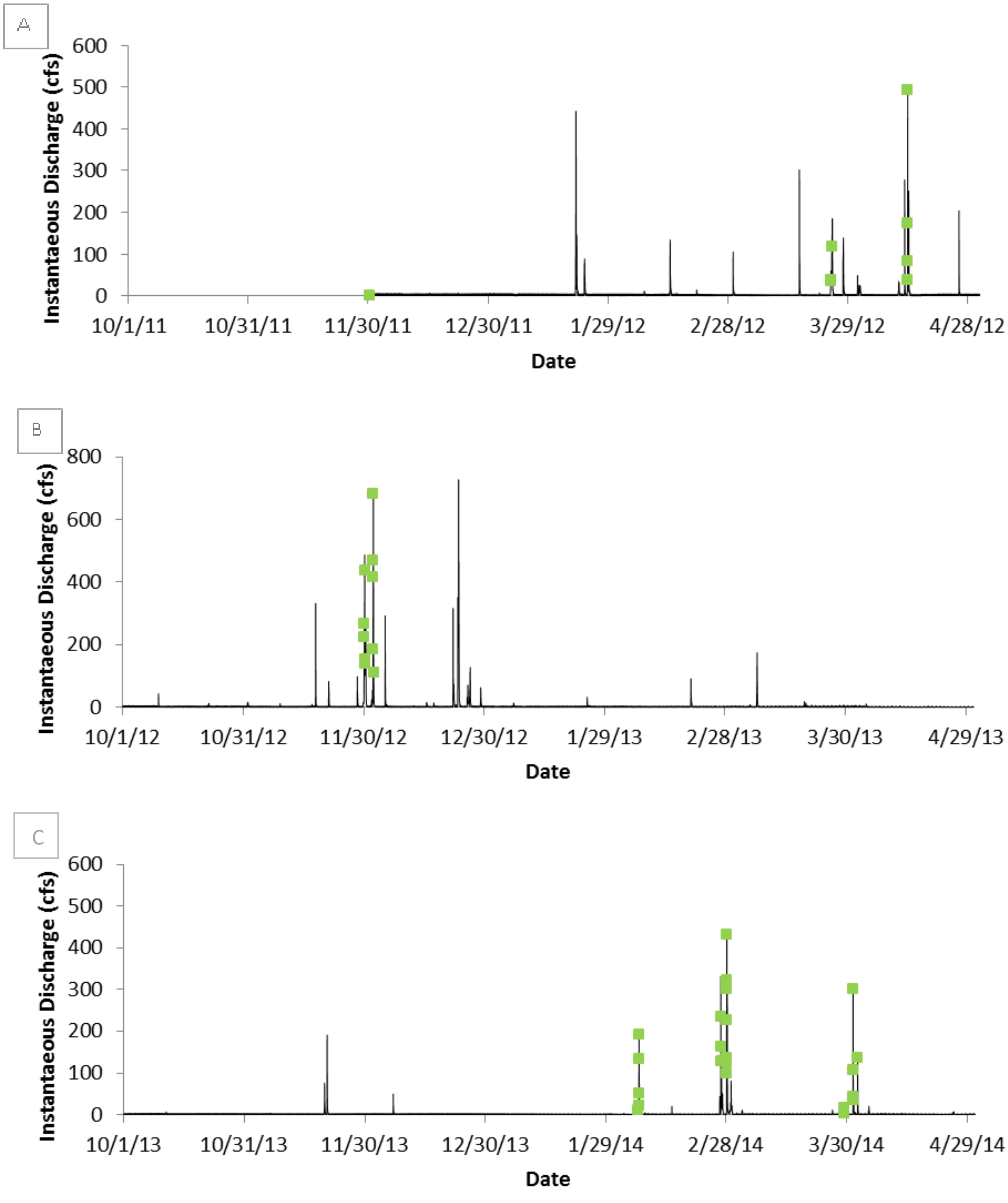

Figure 8. Flow characteristics in East Sunnyvale Channel at East Ahwanee Avenue during WY 2012 (A), WY 2013 (B) and WY 2014 (C) with sampling events marked in green. The flow record is based on the District rating curve for this station as verified by velocity sampling completed during the study in WY 2013.

8.5.2. East Sunnyvale Channel turbidity and suspended sediment concentration

The entire turbidity record for WY 2012 was censored due to problems believed to be with the installation design and the OBS-500 instrument reading the bottom of the channel. In WY 2013, the OBS-500 instrument was replaced with an FTS DTS-12 turbidity probe (0-1,600 NTU range). This instrument performed well through to the first large storm on 11/30/12 and then the turbidity record experienced numerous spikes through the rest of the season. Our observations during maintenance suggested that the three large storm events in late November and December uprooted and dislodged a lot of vegetation and some trash, which slowly passed through the system throughout the season and caught on the boom structure where turbidity was monitored. After field visits to download data and perform maintenance on site including removing the vegetation from the boom, the turbidity record cleared until the next elevated flow. Consequently, 8.3% of the turbidity record was censored due to fouling. In WY 2014, the FTS DTS-12 sensor was used again with more regular field maintenance. Vegetation continued to be a problem throughout the season, fouling the record at times. More regular maintenance and attempts at structural modifications to help deflect vegetation improved the completeness of the record from the previous year, this time with 7% of the record censored and corrected by interpolation.

Given the challenges with the turbidity sensor installation during the first year and vegetation disruptions in the subsequent years, multiple approaches were used for the estimation of SSC. For the portions of the record that were of good quality or deemed to be good quality after correction, turbidity surrogate regression could be used (R^2 = 0.99). For the entire WY 2012 and the portions of the WY 2013 record for which turbidity was not usable, SSC was alternatively computed as a function of flow (with much lower confidence due to the loss of hysteresis in the computational scheme). The relationship with flow was strong (R^2 = 0.98).

Turbidity in East Sunnyvale Channel in WY 2013 and 2014 remained low (<40 NTU) during base flows and increased to between 200 and 1000 NTU during storms. Interestingly, turbidity season peaks in both water years occurred during the seasonal first flush, which also happened to corresponded in both years with storms that were short-lived but relatively intense. Turbidity peaked at 1014 NTU early in the season on 10/9/12 in response to a small but intense rainfall in which 0.19 inches fell in 20 minutes. In WY 2014 and turbidity peaked for the season at 424 NTU on the 11/20/13 storm when 0.25 inches fell in one hour. Three large events in November and December 2012 resulted in turbidities in the 600-900 NTU range, and otherwise turbidity for most other events peaked between 200 and 400 NTU.

Computed suspended sediment concentration in WY 2012 peaked at 370 mg/L on 4/13/12, at 3120 mg/L on 12/2/12, and 845 mg/L on 11/20/13, all in response to the measured peak flow (in WY 2012) or peak turbidity (WY 2013 and 2014) for the given wet season. Two of these three SSC peaks occurred while staff were on site collecting samples. In WY 2014, the maximum SSC sample collected in the field was 514 mg/L (collected on 2/26/2014; the 11/20/13 estimated peak SSC also occurred during a non-sampled storm event).

8.5.3. East Sunnyvale Channel POC concentrations (summary statistics)

A summary of concentrations is useful for providing comparisons to other systems and also for doing a first order quality assurance check on the data generated; data that differs from that reported elsewhere may indicate errors or provide evidence for source characteristics. A wide range of pollutants were measured in East Sunnyvale Channel during the three-year project (Table 23). Concentrations for pollutants sampled at a sufficient frequency for loads analysis (suspended sediments, PCBs, mercury, organic carbon, and nutrients) exhibited the typical pattern of median < mean except for some cases where organic carbon, nitrate, phosphate, and PAH where the mean and median were similar.

The range of PCB concentrations were elevated relative to other mixed urban land use watersheds (range 0.1-1120 ng/L: Lent and McKee, 2011) with maximum concentrations observed at 980 ng/L. Highest PCB concentrations were measured during the February 28, 2014 storm event where an estimated 1.3 inches of rain fell in this watershed. This event followed a 0.9 inch rain event 2 days prior. These concentrations were amongst the highest PCB concentration measured to-date in the Bay Area with project site mean PCB concentrations ranking only behind Pulgas Pump Station - South and Santa Fe Channel. PCB concentrations remained elevated throughout other monitored storms during WY 2014 helping to support a hypothesis that there is a large PCB source in this watershed. Concentrations in the East Sunnyvale Channel watershed appear to be similar to watersheds with industrial sources where concentrations in excess of about 100 ng/L are common (Marsalek and Ng, 1989; Hwang and Foster, 2008; Zgheib et al., 2011; Zgheib et al., 2012; McKee et al., 2012). In contrast, watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). The East Sunnyvale Channel watershed has an imperviousness of 69% and exhibits a particle ratio of 869 pg/mg (based on all sampling to-date including WY 2011 data); the fourth highest observed so far in the Bay Area out of 24 locations and well above the background of rural areas (indicated by Marsh Creek in the Bay Area).

The range of mercury concentrations were comparable to those observed in Z4LA (Gilbreath et al., 2012) while the maximum total mercury concentration in East Sunnyvale Channel (220 ng/L) was greater than sampled in Z4LA (150 ng/L). Concentrations were also much greater than those observed in three urban Wisconsin watersheds (Hurley et al., 1995), urban influenced watersheds of the Chesapeake Bay region (Lawson et al., 2001), and two sub-watersheds of mostly urban land use in the Toronto area (Eckley and Branfirheun, 2008). Similar to Marsh Creek and San Leandro Creek, where the maximum Hg concentrations are somewhat attributed to the erosion of soils, East Sunnyvale Channel watershed also transports high concentrations of suspended sediment (maximum observed from grab samples was 3120 mg/L). Given the relatively low particle ratio (0.22 mg/kg) not greatly elevated about what might be considered background for CA soils (0.1 mg/kg equivalent to ng/mg: Bradford et al., 1996), Hg sources and transport in this watershed are more likely attributed to local atmospheric deposition or perhaps redeposition from historical and ongoing Lehigh Hanson Permanente Cement Plant (Rothenberg et al., 2010a; Rothenberg et al., 2010b). The source-release-transport processes for Hg in

Table 23. Summary of laboratory measured pollutant concentrations in East Sunnyvale Channel during water years 2012, 2013, and 2014.

		2012							2013							2014						
Analyte	Unit	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation
SSC	mg/L	28	96%	0	370	49.5	81.6	100	34	97%	0	3120	301	485	645	75	99%	0	514	125	173	134
ΣPCB	ng/L	8	100%	3.27	119	33.6	41.3	41.5	10	100%	9.16	176	31.3	59.3	64.3	22	100%	2.86	983	90.7	147	223
Total Hg	ng/L	8	100%	6.3	64.1	21.7	27.7	21.7	10	100%	13	220	55.5	72.9	65.2	22	100%	14	120	37	43.1	27
Total MeHg	ng/L	6	83%	0	0.558	0.226	0.25	0.22	6	100%	0.02	0.54	0.22	0.252	0.22	15	93%	0	0.7	0.33	0.332	0.173
TOC	mg/L	8	100%	4.91	8.6	5.94	6.41	1.4	10	100%	4.1	10	5.85	5.85	1.71	22	100%	4.5	30	10.5	13.4	7.94
NO3	mg/L	8	100%	0.2	0.56	0.28	0.309	0.119	10	100%	0.15	0.37	0.28	0.269	0.069	23	100%	0.13	2.6	0.28	0.618	0.714
Total P	mg/L	8	100%	0.19	0.5	0.25	0.277	0.0975	10	100%	0.23	1.7	0.385	0.522	0.434	23	100%	0.11	0.92	0.36	0.408	0.212
PO4	mg/L	8	100%	0.067	0.11	0.079	0.0847	0.0191	10	100%	0.094	0.13	0.12	0.115	0.0098	23	100%	0.006	0.285	0.13	0.148	0.069
Hardness	mg/L	2	100%	51.4	61.2	56.3	56.3	6.93								6	100%	92	340	100	146	97.5
Total Cu	ug/L	2	100%	10.8	19	14.9	14.9	5.79	2	100%	19	31	25	25	8.49	6	100%	11	21	18	16.5	4.09
Dissolved Cu	ug/L	2	100%	4.36	14.8	9.58	9.58	7.38	2	100%	3.1	4.9	4	4	1.27	6	100%	2.8	6.1	4.32	4.63	1.24
Total Se	ug/L	2	100%	0.327	0.494	0.41	0.41	0.118	2	100%	0.49	0.49	0.49	0.49	0	6	100%	0.33	1.9	0.545	0.71	0.593
Dissolved Se	ug/L	2	100%	0.308	0.325	0.317	0.317	0.012	2	100%	0.35	0.39	0.37	0.37	0.0283	6	100%	0.24	1.8	0.47	0.637	0.583
Carbaryl	ng/L	2	100%	11	21	16	16	7.07	2	50%	0	19	9.5	9.5	13.4	6	17%	0	14	0	2.33	5.72
Fipronil	ng/L	2	100%	6	12	9	9	4.24	2	50%	0	6	3	3	4.24	6	100%	3	11	6.5	6.83	2.86
ΣPAH	ng/L	1	100%	289	289	289	289		1	100%	1350	1350	1350	1350		4	100%	382	2770	1660	1620	1260
ΣPBDE	ng/L	1	100%	4.83	4.83	4.83	4.83		1	100%	34.9	34.9	34.9	34.9		4	100%	15.7	103	62	60.6	40.7
Delta/ Tralomethrin	ng/L	1	0%	0	0	0	0		2	100%	3.6	3.8	3.7	3.7	0.141	6	100%	0.6	3.25	1.13	1.42	0.947
Cypermethrin	ng/L	2	0%	0	0	0	0		2	100%	3.2	5.2	4.2	4.2	1.41	6	100%	2.6	6	4.13	4.08	1.16
Cyhalothrin lambda	ng/L	1	0%	0	0	0	0		2	100%	1.2	2.5	1.85	1.85	0.919	5	80%	0	0.6	0.3	0.31	0.213
Permethrin	ng/L	2	100%	5.7	20.9	13.3	13.3	10.8	2	100%	22	48	35	35	18.4	6	100%	11	29	18.8	20.2	6.45
Bifenthrin	ng/L	2	50%	0	8	4	4	5.66	2	100%	8.7	18	13.3	13.3	6.58	6	100%	2	18	5.3	7.56	5.94

Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation at East Sunnyvale Channel was two.

this watershed do not appear to be similar to those of very industrial watersheds with direct local point source discharge (e.g. 1600-4300 ng/L: Ullrich et al., 2007; 100-5000 ng/L: Picado and Bengtsson, 2012; Kocman et al., 2012; 78-1500 ng/L: Rimondi et al., 2014).

The MeHg concentrations during the three-year study ranged from DL-0.7 ng/L. Concentrations of this magnitude or greater have been observed in a number of Bay Area urban influenced watersheds (Zone 4 Line A: Gilbreath et al., 2012; Glen Echo Creek Santa Fe Channel, San Leandro Creek, and Zone 5 Line M: McKee et al., 2012). However, concentrations of methylmercury of this magnitude have not been observed in urbanized watersheds from other parts of the world (Mason and Sullivan, 1998; Naik and Hammerschmidt, 2011; Chalmers et al., 2014). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed, at least for agricultural and forested systems with lesser urban influences, that Hg sources are not a primary limiting factor in MeHg production (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011). Based on plenty of previous sampling experience in numerous Bay Area watershed systems there are no reasons to suspect any data quality issues. Bay Area methylmercury concentrations appear to be elevated, perhaps associated with arid climate seasonal wetting and drying and high vegetation productivity in riparian areas of channels systems with abundant supply of organic carbon each fall and winter.

Nutrient concentrations were also in the same range as measured in Z4LA (Gilbreath et al., 2012) and like the other watersheds reported from the current study, phosphorus concentrations appear to be greater than elsewhere in the world under similar land use scenarios perhaps attributable to geological sources (McKee and Krottje, 2005). Nitrate concentrations appear strikingly similar between East Sunnyvale Channel and San Leandro Creek, Lower Marsh Creek, and Pulgas Pump Station - South. In contrast, nitrate concentrations were about 2-fold greater in Guadalupe River and North Richmond Pump Station. Mean orthophosphate concentrations (0.128 mg/L) were similar to Pulgas Pump Station - South but much lower than observed in the Richmond Pump Station and about 30% elevated above Lower Marsh and San Leandro Creeks. The maximum total P concentration (1.7 mg/L) should be considered very high for an urban watershed however average total P concentrations were similar across the six sites. Concentrations appear typical or slightly greater than for PO4 and TP of found in urban watersheds in other parts of the country and world (e.g. Hudak and Banks, 2006; comprehensive Australian literature review for concentrations by land use class: Bartley et al., 2012). Higher phosphorus concentrations especially the peak concentration observed in East Sunnyvale Channel may perhaps be attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Organic carbon concentrations observed in East Sunnyvale Channel during WYs 2012-2014 (4.1-30 mg/L) were higher than those observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012). It turned out that these were the 3rd greatest observed in the Bay Area to-date. They were greater than but more similar to maximum concentrations observed in Guadalupe River (56 mg/L) but less than Pulgas Green Pump Station (140 mg/L). Although we have not done an extensive literature review of TOC concentrations in the worlds river systems, our general knowledge of the literature would have us hypothesize that concentrations of these magnitudes are very high. These may be contributing to the apparently high methylation rates in the Bay Area.

Of the pollutants sampled at a lesser frequency using a composite sampling design (see methods section) appropriate for characterization only, copper and selenium were similar to concentrations observed in Z4LA (Gilbreath et al., 2012) while PAHs and PBDEs were on the lower end of the range observed in Z4LA.

The maximum concentration of PBDEs (102.7 ng/L) was similar to Guadalupe River during this study (note greater concentrations have been observed in Guadalupe River at Hwy 101 previously: McKee et al., 2006) but lesser by 15-fold than North Richmond Pump Station and greater by about 2-fold than the other locations. Only two peer reviewed articles have previously described PBDE concentrations in runoff, one for the Pearl River Delta, China (Guan et al., 2007), and the other for the San Francisco Bay (Oram et al., 2008) based, in part, on concentration data from Guadalupe River and Coyote Creek taken during WYs 2003-2006. Maximum total PBDE concentrations measured by Guan et al. (2007) were 68 ng/L, a somewhat surprising result given that the Pearl River Delta is a known global electronic-waste recycling hot spot. However, the Guan et al. study was based on monthly interval collection as opposed to storm event-based sampling as was completed in a larger river system where dilution of point source may have occurred.

Similar to the other sites, carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and California (fipronil: 70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012; tributaries to Salton Sea, Southern CA geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008). Project mean Permethrin concentrations at East Sunnyvale Channel were amongst the highest measured to-date ranking only behind Zone 4 Line A. Concentrations of Delta/ Tralomethrin were similar to observed in Lower Marsh Creek and Richmond Pump Station. Bifenthrin were similar to all the other locations except Lower Marsh Creek where they were about 10-fold greater. Concentrations of Cyhalothrin lambda were similar in across San Leandro Creek, Guadalupe River, East Sunnyvale Channel, and Pulgas Pump Station - South and about 2-fold greater in Marsh Creek and Richmond Pump Station. In general, the mix of pyrethriods used in each watershed appears to differ remarkably and is perhaps associated with local applicator and commercially available product preferences in home garden stores.

In summary, PCB concentrations are elevated in the East Sunnyvale Channel relative to typical Bay Area and other urban watersheds, Hg appears to be relatively low, whereas concentrations of other POCs are either within the range of or below those measured in other typical Bay Area urban watersheds and appear consistent with or explainable in relation to studies from elsewhere. Based on these first order comparisons, we see no quality issues with the data.

8.5.4. East Sunnyvale Channel toxicity

Composite water samples were collected in the East Sunnyvale Channel during two storm events in WY 2012, two storm events in WY 2013, and six storm events in WY 2014. No significant reductions in the survival, reproduction and growth of three of four test species were observed during storms. Significant reductions in the survival of the amphipod Hyalella azteca were observed during all WY 2012, WY 2013, and WY 2014 storm events.

8.5.5. East Sunnyvale Channel loading estimates

Given that the turbidity record in WY 2012 was unreliable due to optical interference from bottom substrate (a problem rectified in 2013), and gaps that existed in the WY 2013 record due to vegetation interference throughout the season, continuous SSC was estimated from the discharge record using a linear relation in WY 2012 and power relation for WY 2013 for the period of record in which turbidity was censored, and otherwise using the power relation with turbidity during the period in which the turbidity record was acceptable (Table 24). Concentrations of other POCs were estimated using regression equations between the pollutant and either flow or estimated SSC, whichever relation was stronger. Total organic carbon and the dissolved nutrients did not have a strong relation with either suspended sediment or flow and therefore a flow weighted mean concentration was applied to estimate the loads reported in Table 25. This table highlights how monthly loads can be dominated by a few large storm events. Relative to discharge, suspended sediment load showed quite high variability relative to some of the other sampling locations in the study. Although just one month (December 2012) discharged 16% of the total volume for WYs 2012, 2013, and 2014 combined, 57% of the suspended sediment load was transported during this month as well as approximately 20% of the PCB and 42% of the mercury loads. Given the context that WYs 2012, 2013, and 2014 were relatively dry years, we may be likely to see an even broader range of rainfall-runoff-pollutant transport processes in East Sunnyvale Channel if wetter seasons are sampled in the future – this could be something to consider also if this station were to be chosen as a trend indicator station.

Table 24. Regression equations used for loads computations for East Sunnyvale Channel during water year 2012-2014. Note that regression equations will be reformulated upon future wet season storm sampling.

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Suspended Sediment (WY2012) (mg/L/CFS)	Mainly urban	0.97		0.98	Regression with flow
Suspended Sediment (WY2013) (mg/L/CFS)	Mainly urban	1.08548 (log-log transformed)		0.98	Regression with flow; Duan's BCF = 1.0099
Suspended Sediment (WY2013&14) (mg/L/NTU)	Mainly urban	1.1057 (log-log transformed)		0.99	Regression with turbidity; Duan's BCF = 1.0507
Total PCBs (ng/mg) prior to Feb 28, 2014	Mainly urban	0.0704	34.4079	0.413	Regression with estimated SSC
Total PCBs (ng/mg) post Feb 28, 2014	Mainly urban	1.05	12.91	0.23	Regression with estimated SSC
Total PCBs (ng/L) Fows < 40 CFS	Mainly urban	15.6			Average Low Flow Concentration
Total Mercury (ng/mg)	Mainly urban	0.145	13.1	0.91	Regression with estimated SSC
Total Methylmercury (ng/mg)	Mainly urban	0.000899	0.157	0.68	Regression with estimated SSC
Total Organic Carbon (WYs 2012-13) (mg/L)	Mainly urban	5.7917568			Flow weighted mean concentration
Total Organic Carbon (WY 2014) (mg/L)	Mainly urban	11.870684			Flow weighted mean concentration
Total Phosphorous (mg/mg)	Mainly urban	0.000526	0.272	0.75	Regression with estimated SSC
Nitrate (WYs 2012-13) (mg/L)	Mainly urban	0.245			Flow weighted mean concentration
Nitrate (WY 2014) (mg/L)	Mainly urban	0.323			Flow weighted mean concentration
Phosphate (WYs 2012-13) (mg/L)	Mainly urban	0.104			Flow weighted mean concentration
Phosphate (WY 2014) (mg/L)	Mainly urban	0.133			Flow weighted mean concentration

Table 25. Monthly loads for East Sunnyvale Channel during water years 2012, 2013 and 2014. Italicized loads are estimated based on monthly rainfall-load relationships.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2012	11-Oct	14	*0.128*	*3.22*	*1053*	*5.17*	*2.40*	*0.102*	*37.4*	*15.1*	*37.9*
	11-Nov	9	*0.110*	*2.07*	*939*	*4.16*	*1.83*	*0.0893*	*33.6*	*13.4*	*31.0*
	11-Dec	2	0.148	0.383	855	5.11	1.98	0.0228	36.2	15.4	40.3
	12-Jan	37	0.254	18.2	1473	10.0	5.96	0.0546	62.3	26.5	78.7
	12-Feb	22	0.151	1.85	875	5.33	2.24	0.0246	37.0	15.7	42.0
	12-Mar	69	0.260	11.25	1528	9.14	4.85	0.0494	65.1	26.9	75.3
	12-Apr	39	0.260	18.0	1503	11.7	6.63	0.0614	63.2	26.2	80.8
	Wet season total	192	1.31	55.0	8,227	50.6	25.9	0.404	335	139	386
2013	12-Oct	13	0.122	5.02	709	4.57	2.33	0.355	30.0	12.7	35.9
	12-Nov	61	0.357	84.9	2020	20.0	15.2	0.500	92	38.5	144
	12-Dec	101	0.610	326	3541	42.0	57.0	1.30	144	63.9	328
	13-Jan	8	0.114	2.23	660	4.08	1.81	0.134	27.9	11.9	32.1
	13-Feb	10	0.100	4.58	582	3.78	1.97	0.119	24.6	10.4	29.7
	13-Mar	20	0.138	6.25	799	5.19	2.71	0.193	33.8	14.3	40.7
	13-Apr	6	0.065	0.310	376	2.25	0.892	0.040	15.9	6.75	17.8
	Wet season total	219	1.51	430	8,685	81.9	81.9	2.64	369	159	628
2014	13-Oct	0	0.115	1.15	1374	4.05	1.67	0.109	37.3	15.4	31.9
	13-Nov	14	0.141	18.6	1683	6.17	4.54	0.235	45.7	18.8	48.2
	13-Dec	4	0.096	1.91	1140	3.43	1.53	0.104	31.0	12.7	27.0
	14-Jan	2	0.072	0.609	861	2.53	1.03	0.0971	23.4	9.62	20.0
	14-Feb	65	0.315	51.4	3771	45.2	12.6	0.320	90.9	42.9	141
	14-Mar	38	0.164	13.9	1942	11.0	4.29	0.149	55.7	22.4	49.2
	14-Apr	12	0.107	2.91	1269	4.38	1.82	0.113	52.0	13.5	29.5
	Wet season total	136	1.01	90.4	12,040	76.8	27.5	1.13	336	135	347

8.6. Pulgas Pump Station - South

8.6.1. Pulgas Pump Station - South flow

Flow from the southern catchment of the Pulgas Pump Station - South was monitored for two wet seasons. An ISCO area velocity flow meter situated in the incoming pipe (draining to the catch basin prior to entering the pump station) was used to measure stage and flow in WY 2013 and 2014. A monthly (or partial monthly for December 2012 and March 2013) rainfall to runoff regression (R^2 = 0.97) was applied to estimate total discharge during the missing period of the record. Based on this regression estimator method, coarse estimates of total runoff during WYs 2013 and 2014 were 0.22 Mm^3 and 0.08 Mm^3, respectively.

Runoff from the Pulgas Pump Station - South watershed is highly correlated with rainfall due to its small drainage area and high imperviousness. Mean Annual Precipitation (MAP) for the nearby Redwood City NCDC meteorologic gauge (gauge number 047339-4) was 78% and 35% of normal in WYs 2013 and 2014, respectively. Total runoff for both years at Pulgas Pump Station - South was also likely below normal, and probably more so than the rainfall since total annual discharge generally varies more widely than total annual rainfall. Indeed, the total annual discharge in the nearby USGS-gauged Saratoga Creek was 48% and 9% of normal in WYs 2013 and 2014, respectively.

During the two years of recorded data at Pulgas Pump Station - South, the largest storm series, and subsequently the largest discharge period, occurred in December 2012. Flow peaked during this storm at 50 cfs, while the peak flow in WY 2014 was 33 cfs and occurred during a short but relatively intense storm on 11/20/2013 (Figure 9). December 2012 was only partially monitored (record began on Dec 17, 2012), though by estimating total monthly discharge based on the rainfall-runoff regression, estimated discharge for December 2012 was higher than the entire WY 2014 season's estimated discharge. San Francisquito Creek to the south has been gauged by the USGS at the campus of Stanford University (gauge number 11164500) from WY 1930-41 and again from 1950-present. Annual peak flows in San Francisquito over the long term record have ranged between 12 cfs (WY 1961) and 7200 cfs (WY1998). During WY 2013, flow at San Francisquito Creek peaked at 5400 cfs on 12/23/12 at 18:45, a flow that has been exceeded in only two previous years on record. On the other extreme, during WY 2014 flow peaked at 100 cfs on 4/1/2014 at 22:00. Flow peaks at San Francisquito Creek during these two water years show the contrast in precipitation events between the two years monitored at this site. It is noted, however, that the December 23, 2012 event at Pulgas Pump Station - South was likely not equivalent in magnitude as that which occurred at San Francisquito since the smaller, highly impervious Pulgas Pump Station - South watershed would be less affected by antecedent saturation conditions than San Francisquito Creek and more by hourly and sub-hourly rainfall intensities. The maximum 1-hour rainfall intensity at Pulgas Pump Station - South in WY 2013 was 0.43 inches per hour on 12/23/12 and 0.28 inches per hour on 11/20/2013 in WY 2014, both concurrent with the peak flow for the respective year. Relative to the Redwood City NCDC meteorological gauge and based on the partial duration series, the maximum WY 2013 1-hour rainfall intensity at Pulgas has approximately a 1-year recurrence interval, and therefore much less than a 1-year recurrence for the most intense WY 2014 storm. Based on this rainfall intensity recurrence, we suggest peak flows in Pulgas Pump Station - South watershed were approximately average for WY 2013 and below average in WY 2014.

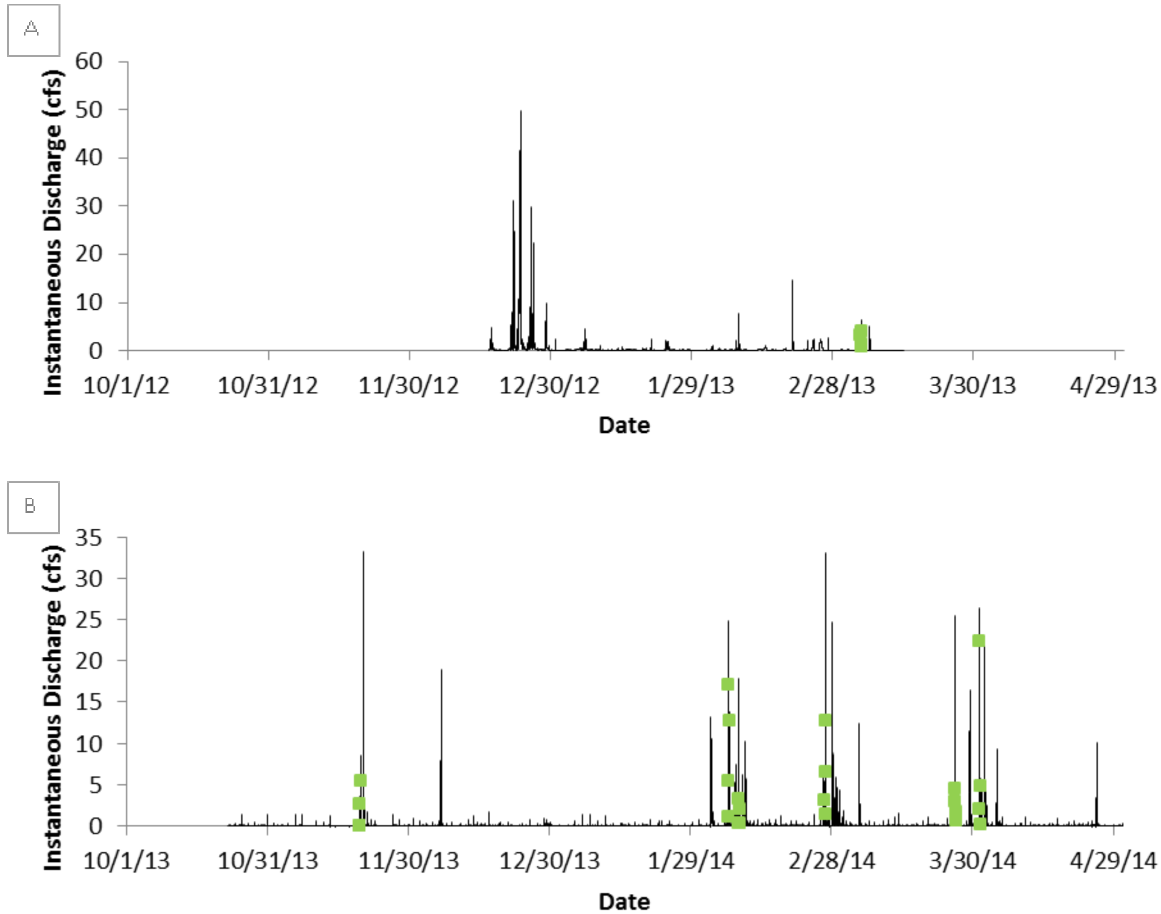

Figure 9. Flow characteristics at Pulgas Pump Station - South during Water Year 2013 and 2014 with sampling events plotted in green.

8.6.2. Pulgas Pump Station - South turbidity and suspended sediment concentration

Turbidity in Pulgas Pump Station - South watershed generally responded to rainfall events in a similar manner to runoff. During non-storm periods, turbidity generally fluctuated between 2 and 20 NTU, whereas during storms, maximum turbidity for each event reached between 100 and 600 NTU. Near midnight on 12/30/12, during flow conditions slightly elevated above base flows but not associated with rainfall, turbidity spiked above the sensor maximum[8] and did not return to readings below 20 NTU for 18 hours. After the first year of sampling, we noted that during all storm events after the 12/30/12 spike, storm maximum turbidities were all greater than maximum turbidities in the large storm series around 12/23/12. We proposed two hypotheses to explain these observations: a) during larger storm events such as the 12/23/12 storm, turbidity becomes diluted, or b) that the signal of particles released into the watershed and measured on 12/30/12 continued to present at lower magnitudes through the

[8] Note the reported DTS-12 turbidity sensor maximum is 1600 NTU. Maximum sensor reading during this spike was 2440 NTU. Given this is beyond the accurate range of the sensor, we do not suggest this reading is accurate but rather reflects that a significant spike in turbidity occurred in the system at this time.

remainder of the season. It remains challenging to tease out which of these hypotheses is more likely correct; turbidity in WY 2014 ranged up to 596 NTU and did peak in most storms higher that the large event on 12/23/12. This would suggest that these turbidities are typical in this watershed. However, WY 2014 was also a very dry year and so it remains possible that the particles released into the watershed and measured on 12/30/12 were still flushing through the system throughout WY 2014.

Turbidity measurements during storms were very spiky, possibly due to the combined factors of the location of the sensor in the catch basin vault and the cyclical pump out from the adjacent pump station. The turbidity record could not be used in regression with manually collected SSC to estimate SSC continuously and therefore it is not possible to estimate the peak SSC during the monitoring period. The highest manually collected SSC was 333mg/L and sampled on 11/19/13 at 16:12. This occurred during a sampled storm in which the continuous turbidity sensor was malfunctioning.

8.6.3. Pulgas Pump Station - South POC concentrations (summary statistics)

A summary of concentrations is useful for providing comparisons to other systems and also for doing a first order quality assurance check. Summary statistics of pollutant concentrations measured in Pulgas Pump Station - South in WY 2013 and 2014 are presented in Table 26. Samples were collected during one storm event in WY 2013 and 6 storm events in WY 2014 (except for dry weather methylmercury sample collection).

The range of WY 2013 PCB concentrations measured during one storm event were generally typical of mixed urban land use watersheds previously monitored in the San Francisco Bay Area (i.e. Guadalupe River, Zone 4 Line A, Coyote Creek, summarized by Lent and McKee, 2011). However, concentrations in WY 2014 were indicative of PCB watershed sources and were the highest concentrations measured in Bay Area stormwater. Maximum concentrations were measured during the storm event on 11/19/2013 and were quantified at 6669 ng/L. Approximately 0.5 inches of rain fell during this storm event and it was one of the earliest events of the WY 2014 season. The previous highest concentration measured (Santa Fe Channel in WY 2011 at 470 ng/L: McKee et al., 2012) was one order of magnitude lower. For the three-year project, mean PCB concentrations were highest at Pulgas Pump Station - South (Pulgas PS - South > East Sunnyvale Channel > Guadalupe River = Richmond Pump Station > San Leandro Creek > Lower Marsh Creek). Concentrations in the Pulgas Pump Station - South watershed appear to be similar to watersheds with industrial sources where concentrations in excess of about 100 ng/L are common (Marsalek and Ng, 1989; Hwang and Foster, 2008; Zgheib et al., 2011; Zgheib et al., 2012; McKee et al., 2012) and in fact are amongst the highest reported in peer-reviewed literature for urban systems. In contrast, watersheds with little to no urbanization dominated by agriculture and open space exhibit average concentrations <5 ng/l (David et al., in press; Foster et al., 2000a; Howell et al. 2011; McKee et al., 2012). In instances where urbanization and industrial sources are highly diluted by >75% developed agricultural land concentrations averaging 8.9 ng/L can be observed (Gómez-Gutiérrez et al., 2006). The Pulgas Pump Station - South watershed has an imperviousness of 87% and exhibits a particle ratio of 1079 pg/mg, the second highest observed so far in the Bay Area out of 24 locations (only Pulgas Pump Station - North is higher) and well above the background of rural areas (indicated by Marsh Creek in the Bay Area).

Table 26. Summary of laboratory measured pollutant concentrations in Pulgas Pump Station - South during water year 2013 and 2014.

		2013							2014						
Analyte	Unit	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation	Samples Taken (n)	Proportion Detected (%)	Min	Max	Median	Mean	Standard Deviation
SSC	mg/L	15	100%	4.3	110	24	33.3	33.1	81	99%	0	333	37	60.8	64.5
ΣPCB	ng/L	4	100%	15.1	62.7	30.5	34.7	20.1	25	100%	16.9	6670	69.5	581	1500
Total Hg	ng/L	6	100%	4.2	23	7.45	10.5	6.9	25	100%	4.2	69	16	20	13.9
Total MeHg	ng/L	6	100%	0.04	0.28	0.215	0.178	0.1	14	100%	0.02	0.66	0.155	0.193	0.167
TOC	mg/L	4	100%	7.3	17	8.35	10.3	4.53	24	100%	4.1	140	11	22.2	31.4
NO3	mg/L	4	100%	0.24	0.49	0.35	0.357	0.102	24	100%	0.1	2.3	0.3	0.484	0.491
Total P	mg/L	4	100%	0.1	0.25	0.125	0.15	0.0707	24	100%	0.067	1.2	0.23	0.313	0.261
PO4	mg/L	4	100%	0.0505	0.0935	0.059	0.0655	0.0195	24	100%	0.056	0.47	0.092	0.133	0.105
Hardness	mg/L								6	100%	40	110	63.5	69.8	29.4
Total Cu	ug/L	1	100%	30	30	30	30		6	100%	22.5	99	36.5	46.3	28.5
Dissolved Cu	ug/L	1	100%	20	20	20	20		6	100%	12	41	13.5	18.3	11.3
Total Se	ug/L	1	100%	0.18	0.18	0.18	0.18		6	100%	0.14	0.6	0.242	0.311	0.175
Dissolved Se	ug/L	1	100%	0.17	0.17	0.17	0.17		6	100%	0.1	0.48	0.19	0.257	0.148
Carbaryl	ng/L	1	100%	204	204	204	204		6	100%	41	189	65.5	88.5	59.4
Fipronil	ng/L	1	0%	0	0	0	0		6	100%	3	6	3.5	3.83	1.17
ΣPAH	ng/L	4	100%	211	1140	552	614	389	2	100%	552	6970	3760	3760	4540
ΣPBDE	ng/L	4	100%	5.18	89.8	32.5	40	39.7	2	100%	52.1	61.4	56.7	56.7	6.59
Delta/ Tralomethrin	ng/L	1	0%	0	0	0	0		6	50%	0	1.2	0.2	0.45	0.565
Cypermethrin	ng/L	1	100%	0.9	0.9	0.9	0.9		6	100%	0.8	5.65	2.7	2.68	1.78
Cyhalothrin lambda	ng/L	1	0%	0	0	0	0		5	100%	0.2	0.8	0.3	0.42	0.268
Permethrin	ng/L	1	100%	2.9	2.9	2.9	2.9		6	83%	0	20	14.3	12	7.94
Bifenthrin	ng/L	1	100%	1.3	1.3	1.3	1.3		6	100%	1.4	15	4.7	5.78	4.92

Zeroes were used in the place of non-detects when calculating means, medians, and standard deviations.
The minimum number of samples used to calculate standard deviation Pulgas Pump Station - South was four.

The range of total mercury concentrations (4-69 ng/L; mean = 15 ng/L) were lower than observed in any of the other watersheds in this study and on the very low end of concentrations sampled in Z4LA (Gilbreath et al., 2012). Pulgas Pump Station - South watershed also exhibits relatively low SSC compared to the other six locations. Of the six POC loads stations monitored during this study, total Hg concentrations in Pulgas Pump Station - South were most similar to those observed in three urban Wisconsin watersheds (Hurley et al., 1995), urban influenced watersheds of the Chesapeake Bay region (Lawson et al., 2001), and two sub-watersheds of mostly urban land use in the Toronto area (Eckley and Branfirheun, 2008). Unlike Marsh Creek, San Leandro Creek, or East Sunnyvale Channel where the maximum Hg concentrations could be either mostly or somewhat attributed to the erosion of upper watershed soils, Pulgas Pump Station - South Watershed transports relatively low Hg concentrations that are most likely attributable to local atmospheric deposition and minor within-watershed sources areas associated with industrial and commercial land uses. Despite low Hg concentrations in water, the particle ratio for total Hg relative to suspended sediment in this watershed (0.8 mg/kg) is the same as observed in Richmond Pump Station watershed and the 3rd highest behind San Leandro Creek and Ettie St. Pump Station watersheds (discounting Guadalupe River and its mining impacted tributaries which all rank higher still). The source-release-transport processes are likely similar to those of other urbanized and industrial watersheds (Barringer et al., 2010; Rowland et al., 2010; Lin et al., 2012) but not likely similar to very highly contaminated watersheds with direct local point source discharge (e.g. 1600-4300 ng/L: Ullrich et al., 2007; 100-5000 ng/L: Picado and Bengtsson, 2012; Kocman et al., 2012; 78-1500 ng/L: Rimondi et al., 2014).

The MeHg concentrations during the two-year study ranged from 0.04-0.66 ng/L. Concentrations of this magnitude or greater have been observed in a number of Bay Area urban influenced watersheds (Zone 4 Line A: Gilbreath et al., 2012; Glen Echo Creek Santa Fe Channel, San Leandro Creek, Zone 5 Line M, Borel Creek, and Pulgas Pump Station - North: McKee et al., 2012). However, concentrations of methylmercury of this magnitude have not been observed in urbanized watersheds from other parts of the world (Mason and Sullivan, 1998; Naik and Hammerschmidt, 2011; Chalmers et al., 2014). Although local Hg sources can be a factor in helping to elevate MeHg production and food-web impacts, it is generally agreed, at least for agricultural and forested systems with lesser urban influences, that Hg sources are not a primary limiting factor in MeHg production (Balogh et al., 2002; Balogh et al., 2004; Barringer et al., 2010; Zheng et al., 2010; Bradely et al., 2011). Based on plenty of previous sampling experience in numerous Bay Area watershed systems, there are no reasons to suspect any data quality issues. Bay Area methylmercury concentrations appear to be elevated perhaps associated with arid climate seasonal wetting and drying and high vegetation productivity in riparian areas of channels systems with abundant supply of organic carbon each fall and winter. Although there is no riparian corridor in the Pulgas Pump Station - South catchment, the pipes nearly always contain water-logged sediment that is deep enough in some areas to create anoxic conditions.

Nutrient concentrations in Pulgas Pump Station - South watershed were also generally in the same range as measured in Z4LA (Gilbreath et al., 2012) and like the other watersheds reported from the current study, phosphorus concentrations appear to be greater than elsewhere in the world under similar land use scenarios perhaps attributable to geological sources (McKee and Krottje, 2005). Nitrate concentrations appear lower in Pulgas Pump Station - South compared to Guadalupe River and Richmond Pump Station but similar East Sunnyvale Channel, San Leandro Creek, and Lower Marsh Creek. Mean orthophosphate concentrations (0.124 mg/L) were similar to East Sunnyvale Channel but much lower than observed in the Richmond Pump Station and about 30% elevated above Lower Marsh and San Leandro Creeks. The maximum total P concentration (1.2 mg/L) should be considered very high for an urban watershed, however average total P concentrations were similar across the six sites. Concentrations of PO4 and TP appear typical or slightly greater than observations in urban watersheds in other parts of the country and world (e.g. Hudak and Banks, 2006; comprehensive Australian literature review for concentrations by land use class: Bartley et al., 2012). Higher phosphorus concentrations, especially the peak concentration observed in Pulgas Pump Station - South may perhaps be attributable to geological sources (Dillon and Kirchner, 1975; McKee and Krottje, 2005; Pearce et al., 2005).

Organic carbon concentrations observed in Pulgas Pump Station - South during WYs 2013-2014 (4.1-140 mg/L) were much greater than those observed in Z4LA (max = 23 mg/L; FWMC = 12 mg/L: Gilbreath et al., 2012). It turned out that these were the greatest concentrations observed in the Bay Area to-date. They were greater than but more similar to maximum concentrations observed in Guadalupe River and East Sunnyvale Channel (56 and 30 mg/L respectively). Although we have not done an extensive literature review of TOC concentrations in the worlds river systems, our general knowledge of the literature would have us hypothesize that concentrations of these magnitudes are very high. High organic carbon concentrations may be contributing to the apparent high methylation rates in Bay Area urban storm drains, creeks, and rivers.

Pollutants sampled at a lesser frequency using a composite sampling design (see methods section) and appropriate for water quality characterization only (copper, selenium, PAHs, carbaryl, fipronil, and PBDEs) were similar to concentrations observed in Z4LA (Gilbreath et al., 2012). PAH concentrations at Pulgas Pump Station - South were almost 2 times higher than the next highest concentration (San Leandro Creek) and were more similar to the previous highest PAH concentration measured (Santa Fe Channel) (McKee et al., 2012). The maximum PBDE concentration (89.9 ng/L) was lower than the other 5 locations in this study with the exception of Lower Marsh Creek. It is possible that low sample numbers and very dry conditions (38% MAP in WY 2014) for this watershed biased the concentrations low; only a future sampling effort would verify the relatively low concentration in comparison to the other highly urban and impervious watersheds in this study. Only two peer reviewed articles have previously described PBDE concentrations in runoff, one for the Pearl River Delta, China (Guan et al., 2007), and the other for the San Francisco Bay (Oram et al., 2008) based, in part, on concentration data from Guadalupe River and Coyote Creek taken during WYs 2003-2006. Maximum total PBDE concentrations measured by Guan et al. (2007) were 68 ng/L, a somewhat surprising result given that the Pearl River Delta is a known global electronic-waste recycling hot spot. However, the Guan et al. study was based on monthly interval collection as opposed to storm event-based sampling as was completed in a larger river system where dilution of point source may have occurred.

Similar to the other sites, carbaryl and fipronil (not measured previously by RMP studies) were on the lower side of the range of peak concentrations reported in studies across the US and California (fipronil:

70 – 1300 ng/L: Moran, 2007) (Carbaryl: DL - 700 ng/L: Ensiminger et al., 2012; tributaries to Salton Sea, Southern CA geometric mean ~2-10 ng/L: LeBlanc and Kuivila, 2008). However, carbaryl concentrations at Pulgas Pump Station - South, although still very low, were 5 to 15 times higher than other POC sites. Concentrations of Cypermethrin were similar to those observed in Z4LA whereas concentrations of Permethrin and Bifenthrin were about 5x and 2x lower, respectively (Gilbreath et al., 2012). In general, the mix of pyrethroids used in each watershed appears to differ remarkably and is perhaps associated with local applicator and commercially available product preferences in home garden stores. For example, concentrations of Cyhalothrin lambda were similar across the Pulgas Pump Station - South, San Leandro Creek, Guadalupe River, and East Sunnyvale Channel sampling sites and about 2-fold greater in Marsh Creek and Richmond Pump Station. Bifenthrin was similar across all six sites with the exception of Lower Marsh Creek where concentrations were observed to be 10-fold greater.

In summary, PCB concentrations are extremely elevated in the Pulgas Pump Station - South relative to other Bay Area watersheds and urban watersheds in other parts of the world. Mercury appears to be relatively low when considering water concentrations alone but elevated in relation to the amount of sediment transported. Whereas concentrations of other POCs are either within range or below those measured in other typical Bay Area urban watersheds and appear consistent with or explainable in relation to studies from elsewhere. Based on these first order comparisons, we see no quality issues with the data.

8.6.4. Pulgas Pump Station - South toxicity

The Pulgas Pump Station - South site was sampled over one storm event in WY 2013 and six discrete storm events in WY 2014. There was no observed toxicity in the WY 2013 event. In WY 2014, Hyalella azteca had reduced survival in all the events sampled. The reductions ranged from 6% to 88%. Additionally the first storm sampled in WY 2014, on November 19, 2013, had a significant reduction in the growth of both S. capricornutum and the fathead minnow by 96% and 45%, respectively. The second WY 2014 storm sampled on February 2, 2014 had a reduction in growth of the fathead minnow by 18% while S. capricornutum was unaffected. No other significant reductions in survival or growth were reported in any of the species for any other samples.

8.6.5. Pulgas Pump Station - South loading estimates

Continuous concentrations of suspended sediment, PCBs, total mercury and methylmercury, and total phosphorous were computed using a simple FWMC estimator (Table 27). This method differs from the previous report (Gilbreath et al., 2014) when a regression estimator method was used. This occurred because more information revealed complex patterns that could not be explained using regression. If the dataset for this site were to improve in the future, these estimates could be recalculated and improved. With these caveats, preliminary monthly loading estimates are dominated by the three wet months (November and December, 2012 and February 2014) during which time 62% of the total discharge volume and load passed through the system. Pulgas Creek exhibited the highest concentrations and unit area normalized loads of the six loading stations for PCBs (Table 28).

Table 27. Regression equations used for loads computations for Pulgas Pump Station - South during water years 2013-2014.

Analyte	Origin of runoff	Slope	Intercept	Correlation coefficient (r^2)	Notes
Suspended Sediment (mg/L)	Mainly urban	66.1			Flow weighted mean concentration
Total PCBs (ng/L)	Mainly urban	132			Flow weighted mean concentration
Total Mercury (ng/L)	Mainly urban	18.6			Flow weighted mean concentration
Total Methylmercury (ng/L)	Mainly urban	0.176			Flow weighted mean concentration
Total Organic Carbon (mg/L)	Mainly urban	9.32			Flow weighted mean concentration
Total Phosphorous (mg/L)	Mainly urban	0.2			Flow weighted mean concentration
Nitrate (mg/L)	Mainly urban	0.249			Flow weighted mean concentration
Phosphate (mg/L)	Mainly urban	0.0776			Flow weighted mean concentration

Table 28. Monthly loads estimated for Pulgas Pump Station - South during water year 2013-2014.

Water Year	Month	Rainfall (mm)	Discharge (Mm^3)	SS (t)	TOC (kg)	PCBs (g)	HgT (g)	MeHgT (g)	NO3 (kg)	PO4 (kg)	Total P (kg)
2013	12-Oct	25	0.0100	0.659	92.9	1.32	0.185	0.00176	2.48	0.774	1.99
	12-Nov	121	0.0515	3.41	480	6.80	0.959	0.00908	12.8	4.00	10.3
	12-Dec	183	0.0829	5.48	773	10.94	1.54	0.0146	20.6	6.43	16.6
	13-Jan	8	0.0034	0.227	32.0	0.453	0.0639	0.000605	0.855	0.266	0.687
	13-Feb	10	0.0039	0.256	36.1	0.512	0.0721	0.000683	0.965	0.301	0.775
	13-Mar	20	0.0073	0.480	67.7	0.959	0.135	0.00128	1.81	0.564	1.45
	13-Apr	18	0.0062	0.407	57.5	0.814	0.115	0.00109	1.53	0.478	1.23
	Wet season total	386	0.165	10.9	1539	21.8	3.07	0.0291	41.1	12.8	33.0
2014	13-Oct	0	0.0004	0.0283	4.00	0.0566	0.00798	0.0000756	0.107	0.0333	0.0858
	13-Nov	24	0.0085	0.611	108	2.69	0.164	0.00160	2.55	0.770	1.96
	13-Dec	8	0.0047	0.309	43.6	0.617	0.0870	0.000824	1.16	0.363	0.935
	14-Jan	0	0.0008	0.0541	7.63	0.108	0.0152	0.000144	0.204	0.0635	0.164
	14-Feb	90	0.0400	2.61	364	5.09	0.745	0.00701	9.79	3.10	8.10
	14-Mar	41	0.0160	1.09	152	2.00	0.290	0.00283	4.06	1.26	3.03
	14-Apr	21	0.0092	0.605	85.3	1.21	0.170	0.00161	2.28	0.711	1.83
	Wet season total	185	0.0796	5.31	764	11.8	1.48	0.0141	20.1	6.31	16.1

Attachment 1. Quality Assurance information

Table A1: Summary of QA data at all sites. This table includes the top eight PAHs found commonly at all sites , the PBDE congeners that account for 75% of the sum of all PBDE congeners, the top nine PCB congeners found at all sites, and the pyrethroids that were detected at any site.

Analyte	Unit	Average Lab Blank	Detection Limit (MDL) (range; mean)	Average Reporting Limit (RL)	RSD of Lab Duplicates (% range; % mean)	RSD of Field Duplicates (% range; % mean)	Percent Recovery of CRM (% range; % mean)	Percent Recovery of Matrix Spike (% range; % mean)
Carbaryl	ng/L	0	9.9-10; 10	20	75.71-75.71; 75.71	1.39-83.55; 42.47	NA	66.64-120.25; 94.99
Fipronil	ng/L	0	0.5-5; 0.945	4.34	NA	0.00-141.42; 28.84	NA	51.52-150.00; 86.24
NO3	mg/L	0	0.002-0.05; 0.0113	0.0488	0.00-0.00; 0.00	0.00-42.43; 2.51	NA	90.00-105.00; 98.98
PO4	mg/L	0	0.0035-0.06; 0.00599	0.0112	0.00-1.61; 0.90	0.00-5.29; 1.51	NA	83.50-126.06; 97.94
Total P	mg/L	0	0.007-0.1; 0.016	0.01	0.00-2.40; 0.79	0.00-33.17; 3.90	NA	86.00-113.00; 97.30
SSC	mg/L	0	0.23-6.8; 2.28	3	NA	0.00-85.48; 12.61	80.99-114.49; 100.72	NA
Benz(a)anthracenes/Chrysenes, C1-	ng/L	0.245	0.0364-75.5; 2.64	NA	NA	NA	NA	NA
Benz(a)anthracenes/Chrysenes, C2-	ng/L	0.177	0.046-43.1; 1.98	NA	NA	NA	NA	NA
Fluoranthene	ng/L	0.152	0.0382-2.58;	NA	NA	NA	NA	NA

Analyte	Unit	Average Lab Blank	Detection Limit (MDL) (range; mean)	Average Reporting Limit (RL)	RSD of Lab Duplicates (% range; % mean)	RSD of Field Duplicates (% range; % mean)	Percent Recovery of CRM (% range; % mean)	Percent Recovery of Matrix Spike (% range; % mean)
			0.446					
Fluoranthene/Pyrenes, C1-	ng/L	0.531	0.103-25.4; 2.08	NA	NA	NA	NA	NA
Fluorenes, C3-	ng/L	1.42	0.0451-29.4; 1.47	NA	NA	NA	NA	NA
Naphthalenes, C4-	ng/L	1.86	0.0461-3.54; 0.751	NA	NA	NA	NA	NA
Phenanthrene/Anthracene, C4-	ng/L	1.44	0.0891-27.1; 2.72	NA	NA	NA	NA	NA
Pyrene	ng/L	0.133	0.0376-5.96; 0.562	NA	NA	NA	NA	NA
PBDE 047	ng/L	0.0363	0.000368-0.000872; 0.000414	NA	NA	NA	NA	NA
PBDE 099	ng/L	0.0379	0.000472-0.0124; 0.00366	NA	NA	NA	NA	NA
PBDE 209	ng/L	0.101	0.0127-0.24; 0.0771	NA	NA	NA	NA	NA
PCB 087	ng/L	0.00147	0.000184-0.0337; 0.00142	NA	NA	NA	NA	NA
PCB 095	ng/L	0.0013	0.000184-0.0372; 0.0016	NA	NA	NA	NA	NA

Analyte	Unit	Average Lab Blank	Detection Limit (MDL) (range; mean)	Average Reporting Limit (RL)	RSD of Lab Duplicates (% range; % mean)	RSD of Field Duplicates (% range; % mean)	Percent Recovery of CRM (% range; % mean)	Percent Recovery of Matrix Spike (% range; % mean)
PCB 110	ng/L	0.00184	0.000184-0.029; 0.00122	NA	NA	NA	NA	NA
PCB 138	ng/L	0.0018	0.000214-0.149; 0.00441	NA	NA	NA	NA	NA
PCB 149	ng/L	0.00101	0.00022-0.151; 0.00469	NA	NA	NA	NA	NA
PCB 151	ng/L	0.000445	0.000184-0.0195; 0.00115	NA	NA	NA	NA	NA
PCB 153	ng/L	0.00178	0.000209-0.132; 0.00392	NA	NA	NA	NA	NA
PCB 174	ng/L	0.0000338	0.000184-0.0118; 0.00106	NA	NA	NA	NA	NA
PCB 180	ng/L	0.000603	0.000184-0.00952; 0.000908	NA	NA	NA	NA	NA
Bifenthrin	ng/L	0.0457	0.05-5.52; 0.761	1.53	NA	NA	NA	NA
Cypermethrin	ng/L	0	0.1-5.29; 0.815	1.53	NA	NA	NA	NA
Delta/Tralomethrin	ng/L	0.155	0.1-1; 0.258	3.05	NA	NA	NA	NA
Total Cu	ug/L	0	0.042-	0.527	0.20-2.68;	0.00-3.72;	100.66-	80.00-

Analyte	Unit	Average Lab Blank	Detection Limit (MDL) (range; mean)	Average Reporting Limit (RL)	RSD of Lab Duplicates (% range; % mean)	RSD of Field Duplicates (% range; % mean)	Percent Recovery of CRM (% range; % mean)	Percent Recovery of Matrix Spike (% range; % mean)
			0.421; 0.114		0.88	1.06	106.15; 102.50	200.00; 97.76
Dissolved Cu	ug/L	0	0.042-0.421; 0.096	0.5	NA	0.00-12.65; 3.92	NA	85.50-98.00; 92.24
Total Hg	ng/L	0	0.2-2; 0.234	0.526	2.12-2.12; 2.12	0.00-63.15; 13.84	91.93-106.84; 99.17	92.99-119.87; 104.34
Total MeHg	ng/L	0.00354	0.01-0.02; 0.0177	0.0401	0.97-5.87; 3.35	0.00-37.52; 8.84	NA	58.99-137.27; 95.64
Total Se	ug/L	0.0094	0.024-0.06; 0.0503	0.0925	0.29-26.96; 5.76	0.00-33.12; 6.97	92.56-103.84; 100.00	80.78-121.22; 95.67
Dissolved Se	ug/L	0	0.024-0.06; 0.0523	0.124	6.18-6.18; 6.18	0.00-6.18; 3.03	NA	87.20-96.22; 91.35
TOC	mg/L	0.0197	0.035-0.3; 0.249	0.481	NA	0.00-15.71; 3.49	NA	0.03-123.00; 96.59

Table A2: Field blank data from all sites.

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
Carbaryl	ng/L	10	20	ND	ND	ND
Fipronil	ng/L	0.714	3.14	ND	ND	ND
NO3	mg/L	0.0123	0.047	ND	0.039	0.00279

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
PO4	mg/L	0.00583	0.01	ND	0.008	0.001
Total P	mg/L	0.00719	0.01	ND	0.057	0.00519
SSC	mg/L	2	3	ND	ND	ND
Acenaphthene	ng/L	0.31	-	ND	ND	ND
Acenaphthylene	ng/L	0.0803	-	ND	0.0663	0.0133
Anthracene	ng/L	0.143	-	ND	ND	ND
Benz(a)anthracene	ng/L	0.0394	-	ND	0.0406	0.00812
Benz(a)anthracenes/Chrysenes, C1-	ng/L	0.0293	-	ND	0.173	0.0814
Benz(a)anthracenes/Chrysenes, C2-	ng/L	0.0515	-	ND	0.393	0.186
Benz(a)anthracenes/Chrysenes, C3-	ng/L	0.0457	-	ND	0.389	0.174
Benz(a)anthracenes/Chrysenes, C4-	ng/L	0.0478	-	ND	1.03	0.329
Benzo(a)pyrene	ng/L	0.111	-	ND	ND	ND
Benzo(b)fluoranthene	ng/L	0.0509	-	ND	0.121	0.0407
Benzo(e)pyrene	ng/L	0.102	-	ND	0.0695	0.0139
Benzo(g,h,i)perylene	ng/L	0.0671	-	ND	ND	ND
Benzo(k)fluoranthene	ng/L	0.11	-	ND	ND	ND
Chrysene	ng/L	0.0407	-	ND	0.151	0.0704
Dibenz(a,h)anthracene	ng/L	0.0693	-	ND	ND	ND
Dibenzothiophene	ng/L	0.0688	-	ND	0.289	0.0974
Dibenzothiophenes, C1-	ng/L	0.089	-	ND	ND	ND
Dibenzothiophenes, C2-	ng/L	0.052	-	0.266	0.71	0.486
Dibenzothiophenes, C3-	ng/L	0.0524	-	0.484	0.782	0.637
Dimethylnaphthalene, 2,6-	ng/L	0.247	-	ND	0.854	0.327
Fluoranthene	ng/L	0.0333	-	0.104	0.343	0.238
Fluoranthene/Pyrenes, C1-	ng/L	0.113	-	0.0828	0.716	0.387
Fluorene	ng/L	0.103	-	ND	0.229	0.098
Fluorenes, C2-	ng/L	0.122	-	1.39	3.5	2.37
Fluorenes, C3-	ng/L	0.133	-	2.95	4.13	3.58

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
Indeno(1,2,3-c,d)pyrene	ng/L	0.0417	-	ND	ND	ND
Methylnaphthalene, 2-	ng/L	0.233	-	ND	5.56	1.7
Methylphenanthrene, 1-	ng/L	0.119	-	ND	0.12	0.0419
Naphthalene	ng/L	0.145	-	1.7	22.4	10.5
Naphthalenes, C1-	ng/L	0.093	-	ND	8.71	2.69
Naphthalenes, C3-	ng/L	0.167	-	0.601	3.94	2.15
Perylene	ng/L	0.116	-	ND	ND	ND
Phenanthrene	ng/L	0.0885	-	0.436	0.717	0.543
Phenanthrene/Anthracene, C1-	ng/L	0.119	-	ND	0.533	0.256
Phenanthrene/Anthracene, C2-	ng/L	0.068	-	0.0581	0.843	0.485
Pyrene	ng/L	0.0323	-	0.0763	0.229	0.164
Trimethylnaphthalene, 2,3,5-	ng/L	0.11	-	ND	0.385	0.176
PBDE 007	ng/L	0.000474	-	ND	0.00164	0.000328
PBDE 008	ng/L	0.000434	-	ND	0.0013	0.00026
PBDE 010	ng/L	0.000561	-	ND	ND	ND
PBDE 011	ng/L	-	-	-	-	-
PBDE 012	ng/L	0.000417	-	ND	0.000793	0.000159
PBDE 013	ng/L	-	-	-	-	-
PBDE 015	ng/L	0.000401	-	ND	0.00416	0.000832
PBDE 017	ng/L	0.000483	-	ND	0.0236	0.00503
PBDE 025	ng/L	-	-	-	-	-
PBDE 028	ng/L	0.000772	-	ND	0.029	0.00609
PBDE 030	ng/L	0.000457	-	ND	ND	ND
PBDE 032	ng/L	0.00042	-	ND	ND	ND
PBDE 033	ng/L	-	-	-	-	-
PBDE 035	ng/L	0.000939	-	ND	ND	ND
PBDE 047	ng/L	0.000478	-	0.0156	1.04	0.266
PBDE 049	ng/L	0.0009	-	ND	0.0863	0.0187

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
PBDE 051	ng/L	0.000521	-	ND	0.00865	0.00173
PBDE 066	ng/L	0.00136	-	ND	0.0494	0.00988
PBDE 071	ng/L	0.000579	-	ND	0.0143	0.00286
PBDE 075	ng/L	0.00102	-	ND	ND	ND
PBDE 077	ng/L	0.000537	-	ND	ND	ND
PBDE 079	ng/L	0.000484	-	ND	ND	ND
PBDE 085	ng/L	0.00151	-	ND	0.0578	0.0137
PBDE 099	ng/L	0.000743	-	0.0295	1.2	0.308
PBDE 100	ng/L	0.000564	-	0.00597	0.281	0.0726
PBDE 105	ng/L	0.0012	-	ND	ND	ND
PBDE 116	ng/L	0.00189	-	ND	0.0113	0.00226
PBDE 119	ng/L	0.00109	-	ND	0.00686	0.00149
PBDE 120	ng/L	-	-	-	-	-
PBDE 126	ng/L	0.000751	-	ND	0.00121	0.000242
PBDE 128	ng/L	0.00495	-	ND	ND	ND
PBDE 140	ng/L	0.000817	-	ND	0.00677	0.00154
PBDE 153	ng/L	0.000892	-	0.00334	0.135	0.0316
PBDE 155	ng/L	0.000608	-	ND	0.00943	0.00207
PBDE 166	ng/L	-	-	-	-	-
PBDE 181	ng/L	0.00218	-	ND	ND	ND
PBDE 183	ng/L	0.00253	-	ND	0.0437	0.00874
PBDE 190	ng/L	0.00454	-	ND	ND	ND
PBDE 197	ng/L	0.00387	-	0.00236	0.0973	0.0498
PBDE 203	ng/L	0.00308	-	ND	0.123	0.0266
PBDE 204	ng/L	-	-	-	-	-
PBDE 205	ng/L	0.00563	-	ND	ND	ND
PBDE 206	ng/L	0.0222	-	ND	1.4	0.287
PBDE 207	ng/L	0.0177	-	ND	2.33	0.488

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
PBDE 208	ng/L	0.0265	-	ND	1.69	0.338
PBDE 209	ng/L	0.0512	-	ND	22.9	4.99
PCB 008	ng/L	0.00134	-	ND	0.0204	0.00303
PCB 018	ng/L	0.000722	-	ND	0.109	0.0112
PCB 020	ng/L	-	-	-	-	-
PCB 021	ng/L	-	-	-	-	-
PCB 028	ng/L	0.000465	-	0.00121	0.065	0.00967
PCB 030	ng/L	-	-	-	-	-
PCB 031	ng/L	0.000515	-	ND	0.0477	0.00667
PCB 033	ng/L	0.000523	-	ND	0.0115	0.00202
PCB 044	ng/L	0.000904	-	ND	0.0494	0.00645
PCB 047	ng/L	-	-	-	-	-
PCB 049	ng/L	0.00102	-	ND	0.0245	0.00277
PCB 052	ng/L	0.000668	-	ND	0.0431	0.0062
PCB 056	ng/L	0.00056	-	ND	0.00776	0.00112
PCB 060	ng/L	0.000608	-	ND	0.0013	0.000306
PCB 061	ng/L	-	-	-	-	-
PCB 065	ng/L	-	-	-	-	-
PCB 066	ng/L	0.000699	-	ND	0.00817	0.00176
PCB 069	ng/L	-	-	-	-	-
PCB 070	ng/L	0.000534	-	0.00121	0.02	0.00467
PCB 074	ng/L	-	-	-	-	-
PCB 076	ng/L	-	-	-	-	-
PCB 083	ng/L	-	-	-	-	-
PCB 086	ng/L	-	-	-	-	-
PCB 087	ng/L	0.000815	-	ND	0.00809	0.00283
PCB 090	ng/L	-	-	-	-	-
PCB 093	ng/L	-	-	-	-	-

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
PCB 095	ng/L	0.000997	-	ND	0.0115	0.00335
PCB 097	ng/L	-	-	-	-	-
PCB 098	ng/L	-	-	-	-	-
PCB 099	ng/L	0.000777	-	ND	0.00753	0.00189
PCB 100	ng/L	-	-	-	-	-
PCB 101	ng/L	0.00155	-	ND	0.00392	0.00246
PCB 102	ng/L	-	-	-	-	-
PCB 105	ng/L	0.000877	-	ND	0.0033	0.000927
PCB 108	ng/L	-	-	-	-	-
PCB 110	ng/L	0.00099	-	ND	0.0113	0.00416
PCB 113	ng/L	-	-	-	-	-
PCB 115	ng/L	-	-	-	-	-
PCB 118	ng/L	0.000824	-	ND	0.00796	0.00237
PCB 119	ng/L	-	-	-	-	-
PCB 125	ng/L	-	-	-	-	-
PCB 128	ng/L	0.000753	-	ND	0.00127	0.000397
PCB 129	ng/L	-	-	-	-	-
PCB 132	ng/L	0.00104	-	ND	0.00272	0.00113
PCB 135	ng/L	-	-	-	-	-
PCB 138	ng/L	0.00124	-	ND	0.012	0.00353
PCB 141	ng/L	0.000792	-	ND	0.00096	0.000246
PCB 147	ng/L	-	-	-	-	-
PCB 149	ng/L	0.00126	-	ND	0.00828	0.00237
PCB 151	ng/L	0.000754	-	ND	0.00463	0.00103
PCB 153	ng/L	0.00193	-	ND	0.00341	0.00154
PCB 154	ng/L	-	-	-	-	-
PCB 156	ng/L	0.000731	-	ND	0.000581	0.000132
PCB 157	ng/L	-	-	-	-	-

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
PCB 158	ng/L	0.000607	-	ND	0.000602	0.000117
PCB 160	ng/L	-	-	-	-	-
PCB 163	ng/L	-	-	-	-	-
PCB 166	ng/L	-	-	-	-	-
PCB 168	ng/L	-	-	-	-	-
PCB 170	ng/L	0.000802	-	ND	0.00131	0.000401
PCB 174	ng/L	0.000818	-	ND	0.00139	0.000347
PCB 177	ng/L	0.000731	-	ND	0.000988	0.000278
PCB 180	ng/L	0.00137	-	ND	0.00274	0.000713
PCB 183	ng/L	0.000725	-	ND	0.00208	0.000442
PCB 185	ng/L	-	-	-	-	-
PCB 187	ng/L	0.00096	-	ND	0.00509	0.000853
PCB 193	ng/L	-	-	-	-	-
PCB 194	ng/L	0.000832	-	ND	0.000731	0.0000522
PCB 195	ng/L	0.000803	-	ND	0.000261	0.0000186
PCB 201	ng/L	0.000633	-	ND	ND	ND
PCB 203	ng/L	0.000903	-	ND	ND	ND
Allethrin	ng/L	0.465	1.5	ND	ND	ND
Bifenthrin	ng/L	0.202	1.5	ND	ND	ND
Cyfluthrin, total	ng/L	1.14	1.5	ND	ND	ND
Cyhalothrin,lambda, total	ng/L	0.24	1.5	ND	0.11	0.0157
Cypermethrin, total	ng/L	0.276	1.5	ND	ND	ND
Delta/Tralomethrin	ng/L	0.21	3	ND	ND	ND
Esfenvalerate/Fenvalerate, total	ng/L	0.254	3	ND	ND	ND
Fenpropathrin	ng/L	0.386	1.5	ND	ND	ND
Permethrin, total	ng/L	1.37	15	ND	ND	ND
Phenothrin	ng/L	0.525	-	ND	ND	ND
Prallethrin	ng/L	7.02	-	ND	ND	ND

Analyte	Unit	Average MDL	RL	Minimum Field Blank	Maximum Field Blank	Average Field Blank
Resmethrin	ng/L	0.653	-	ND	ND	ND
Total Cu	ug/L	0.066	0.444	ND	1.4	0.45
Dissolved Cu	ug/L	0.066	0.444	ND	1.4	0.297
Total Hg	ng/L	0.199	0.482	ND	4.4	0.271
Total MeHg	ng/L	0.0192	0.04	ND	0.021	0.00162
Dissolved Se	ug/L	0.0549	0.096	ND	ND	ND
Total Se	ug/L	0.0549	0.096	ND	ND	ND
Total Hardness (calc)	mg/L	1.46	4.3	ND	ND	ND
TOC	mg/L	0.3	0.5	ND	ND	ND

Table A3: Average RSD of field and lab duplicates at each site.

	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
Analyte	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
Carbaryl	-	-	-	-	-	-	83.50%	75.70%	-	-	1.40%	-
Fipronil	53.00%	-	31.40%	-	9.20%	-	10.90%	-	10.90%	-	-	-
NO3	0.00%	0.00%	9.90%	0.00%	0.50%	-	0.00%	0.00%	1.80%	-	0.40%	-
PO4	0.50%	0.80%	1.90%	0.90%	0.30%	-	1.40%	1.10%	1.50%	-	3.70%	-
Total P	3.60%	0.00%	0.90%	0.00%	3.00%	2.40%	12.40%	0.00%	1.70%	-	2.70%	-

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
SSC	11.00%	-	6.20%	-	11.90%	-	36.20%	-	12.40%	-	10.00%	-
Acenaphthene	20.10%	-	6.30%	3.70%	-	-	10.00%	0.40%	2.10%	1.50%	-	-
Acenaphthylene	10.70%	-	8.50%	5.00%	-	-	31.80%	18.10%	5.70%	5.50%	-	-
Anthracene	14.20%	-	14.10%	5.00%	43.40%	-	39.10%	23.40%	5.60%	4.10%	-	-
Benz(a)anthracene	15.30%	-	18.70%	11.40%	-	-	-	-	-	-	-	-
Benz(a)anthracenes/Chrysenes, C1-	5.70%	-	6.70%	2.30%	2.90%	-	17.30%	6.80%	1.30%	1.30%	-	-
Benz(a)anthracenes/Chrysenes, C2-	4.30%	-	7.80%	7.70%	6.00%	-	19.00%	16.40%	2.80%	1.70%	-	-
Benz(a)anthracenes/Chrysenes, C3-	23.60%	-	15.80%	13.50%	11.10%	-	40.20%	8.90%	2.50%	3.40%	-	-
Benz(a)anthracenes/Chrysenes, C4-	5.90%	-	23.90%	26.40%	10.60%	-	16.70%	7.00%	4.00%	0.40%	-	-
Benzo(a)pyrene	16.70%	-	11.80%	5.10%	20.80%	-	23.60%	6.50%	3.60%	4.80%	-	-
Benzo(b)fluoranthene	9.30%	-	9.70%	6.70%	26.60%	-	17.50%	5.20%	4.60%	4.70%	-	-

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
Benzo(e)pyrene	13.50%	-	7.50%	7.20%	9.90%	-	28.40%	5.90%	2.00%	1.00%	-	-
Benzo(g,h,i)perylene	16.60%	-	5.50%	0.60%	4.60%	-	14.20%	5.30%	3.50%	3.20%	-	-
Benzo(k)fluoranthene	36.40%	-	20.60%	1.80%	-	-	33.00%	2.80%	-	-	-	-
Chrysene	8.40%	-	8.90%	3.50%	9.50%	-	19.00%	7.50%	4.00%	5.00%	-	-
Dibenz(a,h)anthracene	39.90%	-	25.20%	10.90%	-	-	-	-	2.00%	1.20%	-	-
Dibenzothiophene	-	-	7.20%	5.20%	-	-	15.90%	13.00%	-	-	-	-
Dibenzothiophenes, C1-	8.90%	-	5.90%	3.90%	5.10%	-	24.60%	2.90%	7.00%	2.60%	-	-
Dibenzothiophenes, C2-	4.50%	-	7.20%	5.70%	10.20%	-	12.20%	2.90%	4.40%	4.90%	-	-
Dibenzothiophenes, C3-	4.80%	-	8.90%	2.30%	8.00%	-	14.70%	0.80%	3.70%	3.80%	-	-
Dimethylnaphthalene, 2,6-	22.20%	-	5.10%	3.70%	0.40%	-	12.20%	13.80%	4.20%	3.90%	-	-
Fluoranthene	16.00%	-	10.60%	3.30%	33.20%	-	17.20%	16.00%	5.50%	3.50%	-	-
Fluoranthene/Pyrenes, C1-	16.30%	-	9.90%	2.80%	8.70%	-	17.40%	2.90%	2.00%	2.30%	-	-
Fluorene	15.30%	-	15.00%	4.00%	-	-	15.80%	9.10%	2.70%	2.90%	-	-
Fluorenes, C2-	14.00%	-	7.30%	8.90%	0.80%	-	9.40%	1.20%	3.30%	4.30%	-	-

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
Fluorenes, C3-	7.00%	-	11.30%	2.80%	9.00%	-	12.30%	0.10%	2.00%	2.50%	-	-
Indeno(1,2,3-c,d)pyrene	21.90%	-	8.80%	2.30%	14.90%	-	18.10%	5.30%	6.70%	6.70%	-	-
Methylnaphthalene, 2-	9.30%	-	4.10%	2.60%	2.10%	-	10.60%	6.30%	2.40%	1.90%	-	-
Methylphenanthrene, 1-	16.70%	-	14.40%	9.50%	11.60%	-	14.60%	10.70%	0.80%	0.80%	-	-
Naphthalene	10.30%	-	5.20%	1.90%	3.20%	-	2.10%	3.80%	2.40%	0.50%	-	-
Naphthalenes, C1-	14.50%	-	6.40%	3.70%	0.50%	-	7.50%	5.70%	2.30%	1.70%	-	-
Naphthalenes, C3-	17.20%	-	7.80%	7.90%	0.60%	-	8.90%	11.20%	5.30%	5.80%	-	-
Perylene	17.60%	-	13.70%	5.80%	5.00%	-	25.60%	8.60%	3.50%	4.30%	-	-
Phenanthrene	5.80%	-	20.20%	5.30%	29.00%	-	21.30%	26.50%	2.50%	2.10%	-	-
Phenanthrene/Anthracene, C1-	28.70%	-	10.30%	3.00%	13.70%	-	13.00%	0.20%	2.60%	2.00%	-	-
Phenanthrene/Anthracene, C2-	15.60%	-	9.10%	7.30%	7.10%	-	12.90%	8.10%	2.80%	2.80%	-	-
Pyrene	16.70%	-	9.00%	3.00%	19.50%	-	19.20%	14.40%	4.60%	3.90%	-	-
Trimethylnaphthalene, 2,3,5-	22.10%	-	7.80%	3.40%	2.30%	-	17.60%	9.00%	3.30%	4.50%	-	-
PBDE 007	-	-	-	-	-	-	-	11.20%	15.40%	15.60%	2.00%	2.00%

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PBDE 008	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 010	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 012	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 015	11.20%	9.50%	0.70%	-	-	-	3.20%	4.30%	12.30%	15.40%	7.50%	7.50%
PBDE 017	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 028	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 030	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 032	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 035	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 047	8.30%	1.20%	4.40%	-	-	-	13.80%	18.20%	6.40%	0.70%	4.60%	4.60%
PBDE 049	4.10%	0.70%	1.50%	-	-	-	10.20%	8.60%	5.40%	3.20%	12.40%	12.40%
PBDE 051	5.70%	5.70%	0.70%	-	-	-	-	-	10.50%	6.70%	15.30%	15.30%
PBDE 066	2.00%	0.50%	1.10%	-	-	-	13.80%	14.10%	6.30%	2.80%	8.40%	8.40%
PBDE 071	1.90%	1.90%	2.30%	-	-	-	-	-	18.20%	19.60%	32.70%	32.70%

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PBDE 075	0.70%	0.70%	9.80%	-	-	-	-	-	0.80%	0.60%	22.00%	22.00%
PBDE 077	15.80%	15.80%	-	-	-	-	-	-	-	-	-	-
PBDE 079	16.40%	16.40%	-	-	-	-	-	-	21.80%	15.60%	-	-
PBDE 085	12.50%	5.20%	5.00%	-	-	-	4.60%	5.70%	12.40%	3.70%	2.90%	2.90%
PBDE 099	8.90%	3.90%	3.30%	-	-	-	8.10%	9.90%	9.30%	2.40%	4.80%	4.80%
PBDE 100	5.20%	0.30%	3.80%	-	-	-	9.20%	11.70%	8.90%	1.10%	6.00%	6.00%
PBDE 105	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 116	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 119	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 126	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 128	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 140	-	-	30.00%	-	-	-	12.10%	12.50%	15.70%	2.70%	9.80%	9.80%
PBDE 153	11.20%	6.60%	9.90%	-	-	-	6.20%	7.10%	9.50%	3.80%	3.50%	3.50%
PBDE 155	9.20%	12.50%	-	-	-	-	6.40%	7.80%	17.60%	3.70%	6.00%	6.00%

	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
Analyte	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PBDE 181	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 183	16.40%	1.50%	18.50%	-	-	-	27.40%	32.60%	15.40%	6.10%	11.00%	11.00%
PBDE 190	-	-	-	-	-	-	-	-	-	-	1.70%	1.70%
PBDE 197	34.70%	12.30%	15.80%	-	-	-	-	-	-	-	-	-
PBDE 203	25.10%	17.60%	14.80%	-	-	-	-	3.30%	22.40%	12.70%	4.60%	4.60%
PBDE 205	-	-	-	-	-	-	-	-	-	-	-	-
PBDE 206	18.40%	23.90%	10.60%	-	-	-	6.10%	7.60%	21.90%	10.50%	37.30%	37.30%
PBDE 207	24.20%	25.50%	8.30%	-	-	-	2.00%	2.10%	24.70%	14.30%	28.20%	28.20%
PBDE 208	23.50%	23.70%	11.30%	-	-	-	3.50%	4.10%	24.60%	14.50%	30.50%	30.50%
PBDE 209	21.60%	19.40%	1.60%	-	-	-	2.10%	2.20%	19.90%	5.10%	42.30%	42.30%
PCB 008	14.40%	10.40%	13.70%	13.60%	20.00%	-	5.00%	0.30%	23.50%	9.70%	6.90%	11.90%
PCB 018	-	-	-	-	-	-	-	-	26.60%	5.20%	4.70%	-
PCB 028	-	-	-	-	-	-	-	-	20.30%	3.60%	5.10%	-
PCB 031	10.80%	9.10%	8.80%	7.50%	8.50%	-	4.70%	0.70%	17.10%	2.60%	4.90%	0.80%

	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
Analyte	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PCB 033	-	-	-	-	-	-	-	-	24.40%	7.00%	6.50%	-
PCB 044	-	-	-	-	-	-	-	-	13.10%	8.60%	-	-
PCB 049	-	-	-	-	-	-	-	-	15.50%	12.80%	-	-
PCB 052	8.90%	13.80%	12.30%	10.40%	9.90%	-	7.00%	14.40%	18.60%	15.60%	11.40%	6.60%
PCB 056	6.20%	5.10%	13.90%	7.30%	2.20%	-	5.50%	12.00%	13.40%	1.70%	16.20%	3.80%
PCB 060	5.60%	4.30%	14.50%	7.80%	2.00%	-	6.10%	13.60%	11.30%	1.70%	14.60%	3.20%
PCB 066	7.00%	8.00%	11.40%	8.90%	1.50%	-	8.20%	15.00%	11.20%	2.80%	16.00%	1.60%
PCB 070	-	-	-	-	-	-	-	-	6.00%	9.90%	-	-
PCB 087	-	-	-	-	-	-	-	-	18.40%	22.40%	9.30%	-
PCB 095	-	-	-	-	-	-	-	-	21.10%	29.80%	16.10%	-
PCB 099	-	-	-	-	-	-	-	-	20.60%	24.70%	22.30%	-
PCB 101	-	-	-	-	-	-	-	-	17.10%	23.90%	20.10%	-
PCB 105	7.40%	7.90%	19.30%	11.00%	13.40%	-	7.70%	19.20%	14.90%	11.40%	17.30%	22.50%
PCB 110	-	-	-	-	-	-	-	-	16.60%	20.90%	11.00%	-

	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
Analyte	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PCB 118	7.70%	8.60%	21.00%	8.70%	15.00%	-	8.10%	20.80%	15.20%	13.60%	16.30%	27.90%
PCB 128	19.80%	19.80%	-	-	-	-	-	-	7.20%	15.00%	3.30%	-
PCB 132	9.70%	9.20%	20.00%	4.70%	18.50%	-	11.80%	25.80%	13.20%	18.40%	5.30%	11.40%
PCB 138	-	-	-	-	-	-	-	-	6.60%	10.80%	1.40%	-
PCB 141	9.40%	10.30%	19.40%	3.50%	14.80%	-	14.00%	22.90%	15.50%	15.60%	7.70%	15.90%
PCB 149	-	-	-	-	-	-	-	-	4.80%	10.40%	3.90%	-
PCB 151	-	-	-	-	-	-	-	-	3.00%	5.90%	3.50%	-
PCB 153	-	-	-	-	-	-	-	-	6.40%	7.60%	2.70%	-
PCB 156	-	-	-	-	-	-	-	-	8.00%	18.60%	-	-
PCB 158	8.90%	11.00%	18.50%	3.80%	16.70%	-	11.10%	24.80%	15.60%	16.00%	9.40%	16.70%
PCB 170	7.30%	4.70%	15.90%	1.40%	11.30%	-	13.20%	24.70%	20.80%	7.90%	5.30%	7.70%
PCB 174	5.60%	1.70%	14.20%	2.20%	11.50%	-	21.80%	36.30%	13.80%	1.50%	6.30%	7.20%
PCB 177	6.00%	3.70%	13.30%	3.40%	18.90%	-	20.10%	-	16.60%	4.30%	4.90%	6.00%
PCB 180	-	-	-	-	-	-	23.70%	29.50%	15.00%	4.40%	-	-

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
PCB 183	-	-	-	-	-	-	33.10%	31.60%	13.40%	5.50%	-	-
PCB 187	5.20%	3.80%	11.00%	3.90%	6.40%	-	23.80%	34.90%	15.00%	5.00%	8.60%	10.50%
PCB 194	7.40%	3.30%	19.00%	5.60%	14.40%	-	16.10%	38.70%	22.70%	12.20%	5.90%	8.20%
PCB 195	5.50%	2.00%	18.10%	3.40%	29.70%	-	15.30%	26.90%	24.80%	12.70%	4.30%	3.80%
PCB 201	8.80%	2.40%	13.20%	1.10%	10.10%	-	23.30%	-	13.20%	6.80%	8.00%	8.20%
PCB 203	7.70%	6.70%	15.50%	5.40%	14.30%	-	18.20%	44.10%	17.80%	17.10%	9.60%	12.90%
Allethrin	-	-	-	-	-	-	-	-	-	-	-	-
Bifenthrin	18.70%	-	11.10%	-	8.50%	-	4.80%	-	9.70%	-	0.00%	-
Cyfluthrin, total	14.60%	-	17.90%	-	-	-	-	-	4.30%	-	6.60%	-
Cyhalothrin,lambda, total	-	-	-	-	-	-	-	-	-	-	0.00%	-
Cypermethrin, total	-	-	30.40%	-	27.60%	-	-	-	1.60%	-	1.30%	-
Delta/Tralomethrin	-	-	39.50%	-	32.40%	-	23.00%	-	58.00%	-	12.90%	-
Esfenvalerate/Fenvalerate, total	-	-	10.10%	-	-	-	-	-	24.40%	-	-	-

Analyte	San Leandro Creek		East Sunnyvale Channel		Lower Marsh Creek		Guadalupe River		Richmond Pump Station		Pulgas Creek	
	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD	Avg Field RSD	Avg Lab RSD
Fenpropathrin	-	-	-	-	-	-	-	-	-	-	-	-
Permethrin, total	12.90%	-	10.90%	-	10.60%	-	2.10%	-	5.20%	-	4.00%	-
Phenothrin	-	-	-	-	-	-	-	-	-	-	-	-
Prallethrin	-	-	-	-	-	-	-	-	-	-	-	-
Resmethrin	-	-	-	-	-	-	-	-	-	-	-	-
Total Cu	0.90%	1.10%	0.10%	0.20%	0.40%	0.80%	-	-	0.00%	-	3.40%	-
Dissolved Cu	6.30%	-	1.60%	-	-	-	-	-	3.80%	-	-	-
Total Hg	18.70%	2.10%	11.80%	-	4.50%	-	12.30%	-	9.70%	-	16.90%	-
Total MeHg	10.00%	4.10%	11.90%	-	2.70%	-	10.60%	2.60%	10.70%	-	1.40%	-
Dissolved Se	3.10%	6.20%	1.60%	-	-	-	-	-	5.20%	-	-	-
Total Se	11.60%	10.10%	0.00%	-	4.10%	1.50%	1.40%	1.40%	0.00%	-	6.40%	-
Total Hardness (calc)	1.20%	-	8.30%	-	-	-	-	-	0.00%	-	6.30%	-
TOC	1.50%	-	3.00%	-	3.80%	-	6.10%	-	6.40%	-	1.50%	-

Attachment 2. Intercomparison Studies

Due to the change in analytical labs for 2013 and 2014 from those used previously in loading studies, a limited number of split samples were analyzed for intercomparison with results from laboratories contracted in previous years.

In general, the intra-lab variation from replicate analyses performed on these samples for both the current and previous contract labs, was much smaller than the inter-lab variation. This is to be expected; analytical biases (e.g., from mis-calibration, incomplete extraction, matrix interferences, etc.) will tend to recur and be more consistent within a lab than among labs. Even if both labs perform within typical acceptance limits for CRMs or other performance tests, the net difference between labs can sometimes be exacerbated by biases in opposite directions, or interferences present in specific field matrices but not reference materials, and in limited studies, it may be possible only to estimate a typical difference, not establish which lab's results are more accurate. Differences in results between years and between sites analyzed by different labs that are smaller than or similar to the inter-lab measurement differences may not be real or significant and may only reflect measurement differences between labs.

Even in larger intercomparison exercises with multiple labs, there is no assurance provided that the certified or consensus value is absolutely accurate, only a weight of evidence that more or most labs get a similar result. Such a consensus may in part reflect a common bias among labs encountering a similar interference or bias of choosing a particular extraction or analytical method.

The following section will discuss results on split samples analyzed for this project in 2013 and 2014 for various analytes. In most cases the differences among labs were within common precision acceptance limits (e.g., 25% RPD for intra-lab replicates for trace metals in RMP or SWAMP) or within the expected combined (propagated) error for separate measurements of recovery (e.g. within 25% of target values for 2 independent labs for reference materials or matrix spikes for trace elements; propagated error = square root $((25\%)^{\wedge 2} + (25\%)^{\wedge 2})$ = ~35%). In cases where the results between labs show a consistent bias, it may be possible to adjust for the bias in evaluating interannual or inter-site differences, but in cases where the inter-lab differences appear more randomly distributed, smaller interannual or inter-site differences may not be distinguishable from measurement uncertainty.

In cases where more random or less systematic differences were found between the labs' results, it is often difficult to diagnose the cause without extensive investigation. Causes of the discrepancies may be particular to specific samples, or sporadic and hard to reproduce. However, because the data in this study are compiled to develop overall pictures of concentrations and loads from the various watersheds, the impact of measurement errors or variations in any individual samples is lessened; random errors will partially offset and the aggregate statistics will reasonably allow estimation of the central tendency of the data. For many of the analytes, the results were often in good agreement (near a 1:1 line) for all but 1 of the split sample pairs, so the data can, in many cases, be compared with acknowledgement of measurement uncertainty but without requiring adjustment, which is suitable only for cases of systematic bias. Results for specific analytes are discussed below.

Trace Elements

Copper

Copper was measured by Caltest (the “Target Lab”) in 2013 and 2014, and by Brooks Rand (the “IC Lab”) in previous years. Three samples each of dissolved and total copper were split and analyzed by both labs in the course of the study. For both labs, the within lab RPDs were within 5% or better for these split samples, suggesting that individually, neither of the labs had noticeable issues with subsampling the provided samples uniformly for replicate analysis. In general, the IC lab reported concentrations higher than the target lab for any given sample (Figure 10). For dissolved copper, the average difference in slope (fitting a linear regression through the origin, vs. an “ideal” 1:1 line) was 28%, and for total copper, the average difference in slope was 15%. For individual result pairs, the target lab result was always lower, ranging 65% to 89% (average 74%) of the IC lab result for dissolved samples and 83% to 95% (average 87%) for total samples; average RPD was 31% for dissolved copper, and 14% for total copper. These data hint at a systematic bias, but because of the small number of samples in the comparison and differences between labs within or nearly within common acceptance limits for within lab variation (e.g., 25% RPD for metals) more evidence of a systematic bias would be recommended before attempting to develop an adjustment factor between labs.

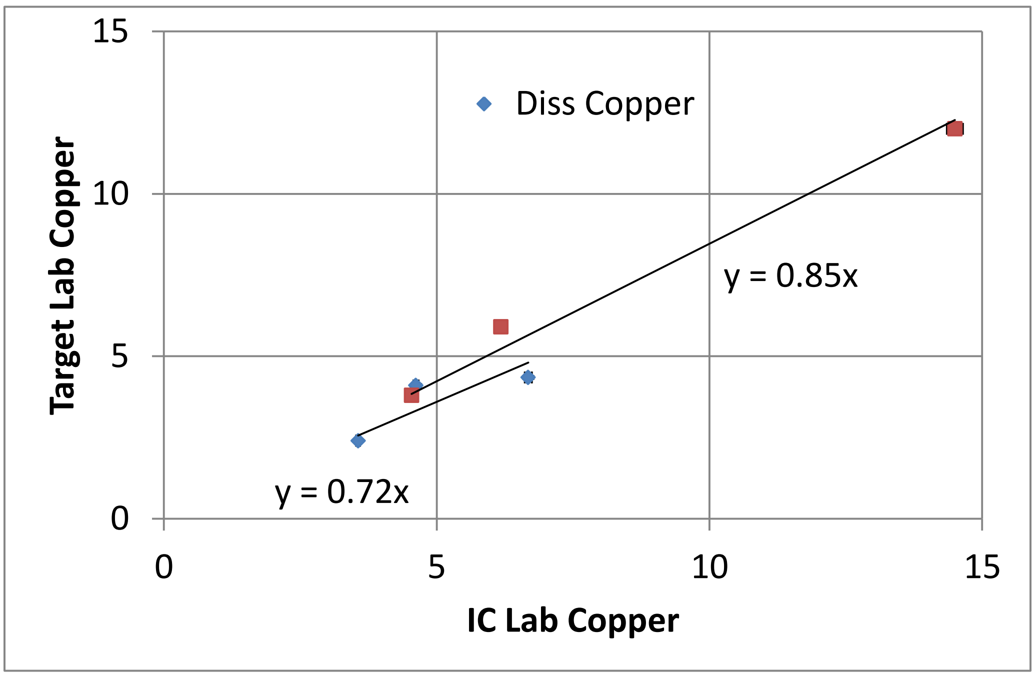

Figure 10. Target versus IC lab dissolved and total copper in split water samples for 2013 to 2014.

Total Mercury

Total mercury was measured by Caltest (the “Target Lab”) in 2013 and 2014, and by Moss Landing Marine Labs (the “IC Lab”) in previous years. Seven total (unfiltered) water samples were split and analyzed for total mercury by both labs in the course of the study. For both labs, none of these split samples were analyzed as lab replicates, but precision on lab replicate analyses averaged 16% RSD in

2014 for the target lab and 3% in 2014 for the IC lab. Similar to copper, the IC lab generally reported concentrations higher than the target lab for any given sample (Figure 11), although the bias is less consistent. For total mercury, the target lab result ranged 51% to 105% (average 82%) of the IC lab result; the average RPD was 25%. Much of this difference was driven by a single result pair in 2014, where the IC lab result was nearly double that of the target laboratory; without that pair, the slope would have been near 1:1 (1.03), so correction is not warranted given the overall deviation depending largely on that one sample pair.

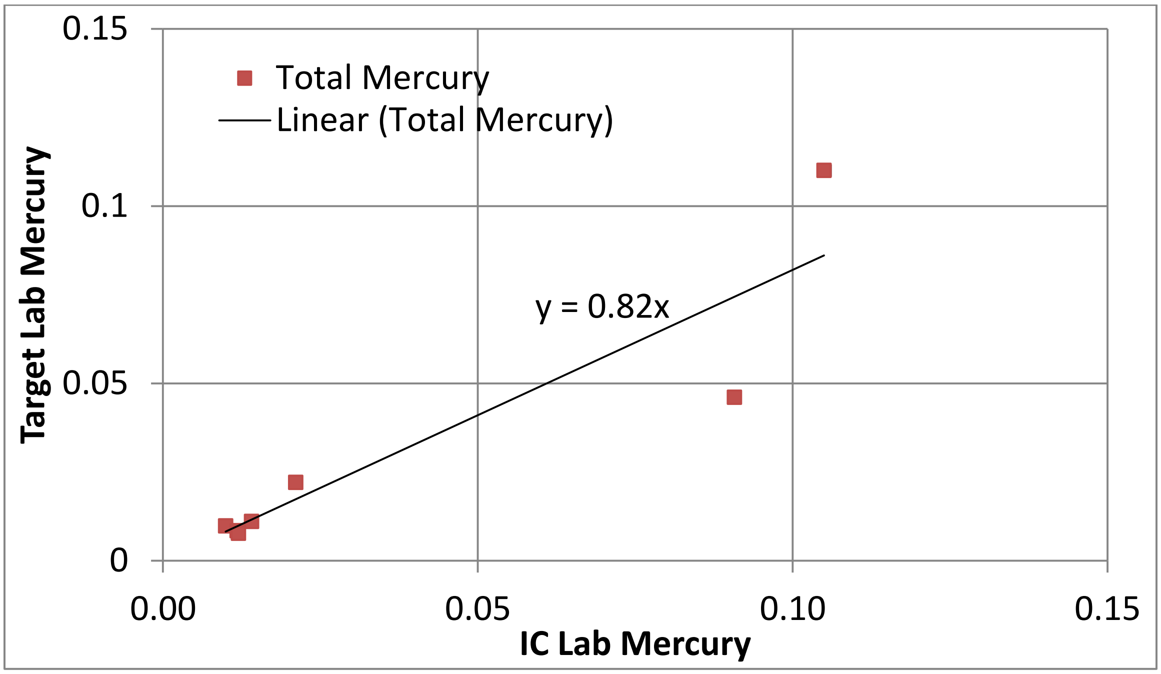

Figure 11. Target versus IC lab total mercury in split water samples for 2013 to 2014.

Methylmercury

Methyl mercury was measured by Caltest (the "Target Lab") in 2013 and 2014, and by Moss Landing Marine Labs (the "IC Lab") in previous years. Four total (unfiltered) water samples were split and analyzed for methylmercury by both labs in the course of the study. Only the IC lab analyzed one of these split samples directly in lab replicates, with <1% RSD, but the target lab also had acceptable precision with average 16% RSD in 2014 for other samples in the project. Unlike the other metals, the results for the IC lab averaged slightly lower than the target lab (Figure 12). For methylmercury, the target lab ranged 90% to 132% (average 105%) of the IC lab result. The average RPD was 12%, with some points both above and below the 1:1 line. Similar to copper, the differences are neither large enough nor consistent enough to warrant a correction factor.

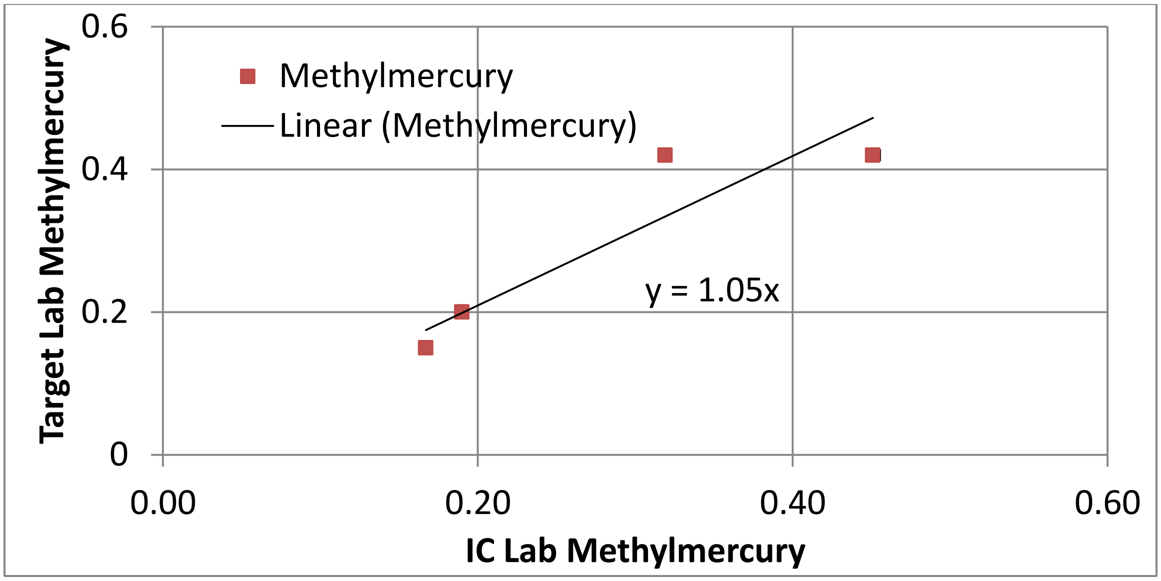

Figure 12. Target versus IC lab methylmercury in split water samples for 2013 to 2014.

Selenium

Selenium was measured by Caltest (the “Target Lab”) in 2013 and 2014, and by Brooks Rand (the “IC Lab”) in previous years. Two samples each of dissolved and total selenium were split and analyzed by both labs in the course of the study. For both labs, the within lab replicate RPDs were good, within 10% or better for these split samples. In general, the IC lab reported concentrations very slightly higher than the target lab for any given sample (Figure 13), but results were nearly identical among labs, and very similar between dissolved and total phase for any given sampling event. For dissolved selenium, the target lab results were 89% to 97% (average 92%) of the IC lab, and for total selenium 88% to 98% (average 95%). Averages of individual result pair RPDs were 9% for dissolved selenium, and 5% for total selenium. Corrections for selenium are clearly not warranted given the very good agreement.

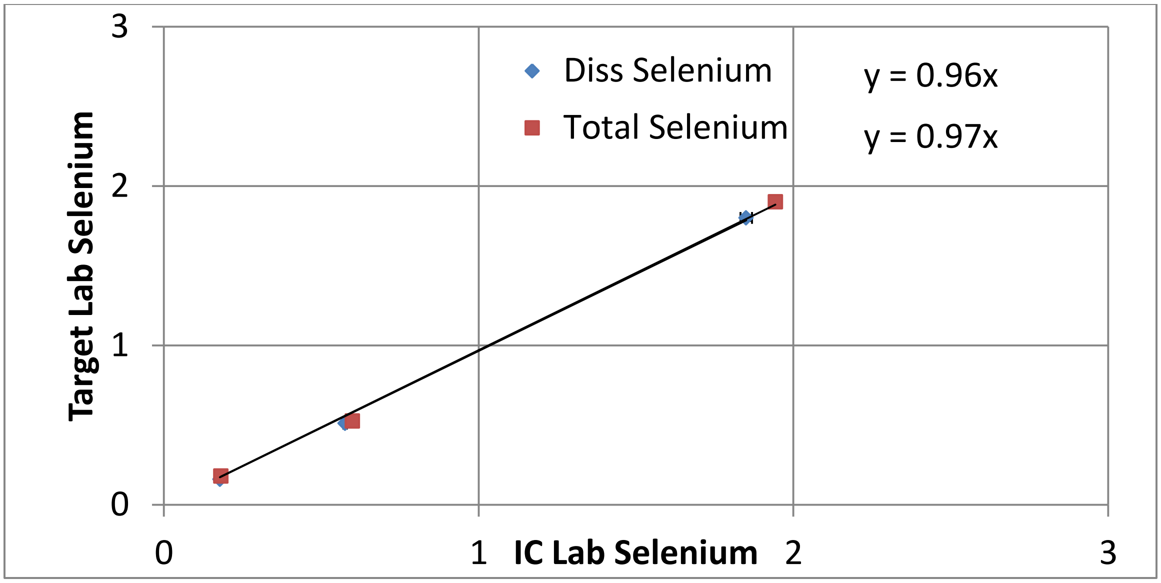

Figure 13. Target versus IC lab dissolved and total selenium in split water samples for 2013 to 2014.

Hardness

Hardness was measured by Caltest (the "Target Lab") in 2013 and 2014, and by Brooks Rand (the "IC Lab") in previous years by a calculation from Ca and Mg concentrations. Three samples were split for analysis by both labs in the course of the study. For the target lab, the within lab replicate RPDs or RSDs were 6% to 12% for these split samples, and for the IC lab 3% on the one sample they analyzed in replicate. There was no consistent bias, with the target lab reporting 85% to 116% (average 100%) of the IC lab result. Although recovery errors in lab control samples (a clean lab matrix) by the target lab were generally within 10% or better of the target value, for field sample matrix spikes, recoveries were highly variable, as low as 30% recovery (70% error), averaging over 20% error. The moderately large average recovery error and sporadic large excursions suggest uncertainties in the target lab hardness data, leading 2013 results to be censored (although raw results remain in the database, and are plotted in Figure 14). The IC lab did not report recovery on hardness directly, but recovery was good on Ca and Mg, with modest errors (from 8% to 12%). Given a lack of consistent bias, a correction factor is not warranted for hardness measurements.

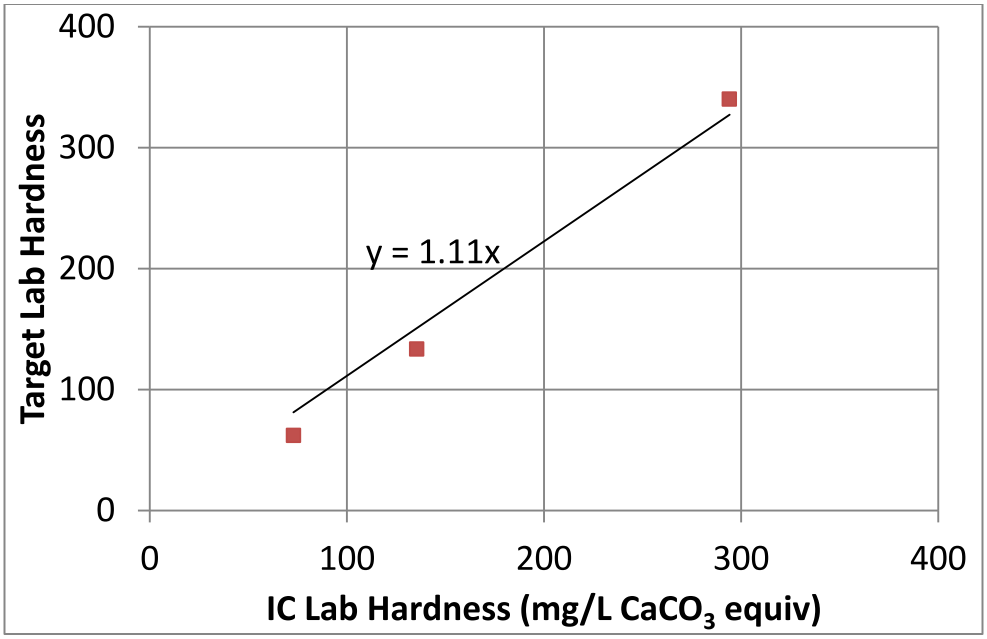

Figure 14. Target versus IC lab hardness in split water samples for 2013 to 2014.

Suspended Sediment Concentration

Suspended sediment concentration (SSC) was measured by Caltest (the "Target Lab") in 2013 and 2014, and by EBMUD (the "IC Lab") in previous years. Three samples were split for analysis by both labs in the course of the study. For the target lab, the lab replicate RSDs were 6% to 12% for these split samples, and for the IC lab 3% on the one sample they analyzed in replicate. There was no consistent bias between labs (Figure 15). The target lab reported results 41% to 150% (average 101%) those of the IC lab, with the largest relative differences on the lower concentration samples. Recoveries on LCS samples by the target lab were within 10% of the expected values. The IC lab reported recovery on performance testing reference materials, with recovery errors for different materials of 1% to 19%. Despite the large variations in the comparison of results between labs, the differences were not consistently biased and thus would not justify application of a correction factor.

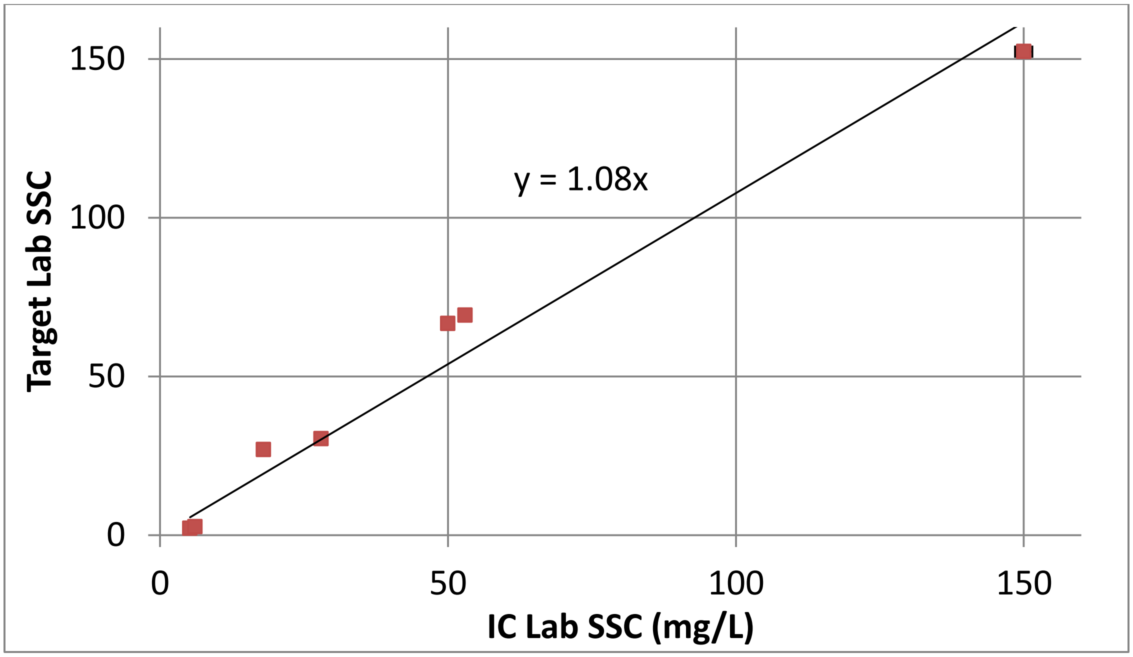

Figure 15. Target versus IC lab SSC in split water samples for 2013 to 2014.

Nutrients

Nutrients were measured by Caltest (the “Target Lab”) in 2013 and 2014, and by EBMUD (the “IC Lab”) in previous years. Seven samples were split for analysis of nitrate by both labs. For the IC lab, the lab replicate RSDs were 1% or better for these split samples. The target lab did not analyze any of these split samples in replicate, but RSDs for lab replicates on other field samples averaged 5%. The target lab generally reported lower concentrations except for the highest sample (Figure 16), ranging 76% to 108% (average 90%) of those from the IC lab, with the largest relative differences mostly on the lowest concentration samples (RPDs on paired splits of 2% to 28%, averaging 15%). Recoveries on LCS samples by the target lab averaged within 3% of the expected values, while the IC lab LCS sample recovery errors averaged 24%. The IC lab spiked at much lower levels however (around 0.05 mg/L vs ~4 for the target lab) which may in large part explain the seemingly poorer recoveries. Differences among the labs results were not systematic and do not warrant a correction factor for comparison.

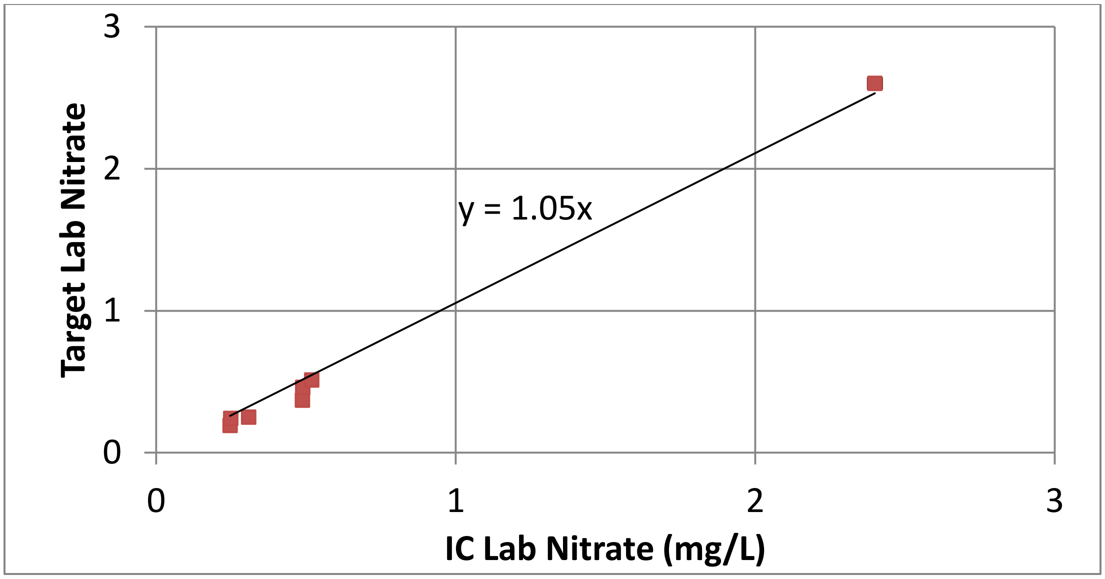

Figure 16. Target versus IC lab nitrate in split water samples for 2013 to 2014.

Orthophosphate was measured in seven split samples by both labs. For both labs, lab replicate RSDs were <1% for these split samples. The target lab reported a much lower concentration (69% of the IC lab result on one sample), but otherwise had similar results (Figure 17), around 92% to 101% of those from the IC lab (average 93% including all samples). Reported as RPDs on paired splits, the differences ranged from 0% to 37%, averaging 8%. Recoveries on IC lab LCS samples were biased high an average 14%, which may explain in part the differences among labs, but without the one sample with the target lab at 69% of the IC results, results would be near 1:1 between the labs.

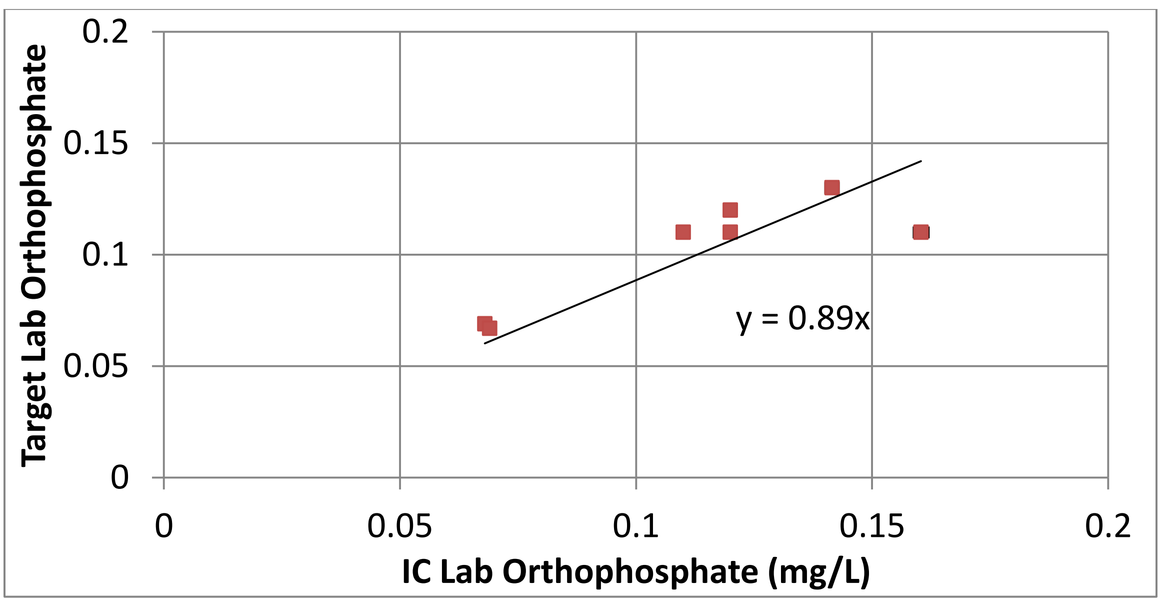

Figure 17. Target versus IC lab orthophosphate in split water samples for 2013 to 2014.

Phosphorus was reported for four split samples. For 3 of the 4 samples results generally agreed (target lab results 70% to 96% of the IC lab's), but for one, the concentration for the target lab was 4x lower (Figure 18). RPDs ranged from 140% for the latter sample pair, to 4% for the best paired results. Although the IC lab 2013 sample batch was flagged for low recovery (86%), below the target MQO of 10% error (90% recovery), that would not explain the discrepancy between the labs since the IC lab result was biased high relative to the target lab. The lab replicate precision was good for both the target and IC labs for these split samples (RSDs <5%), so measurement variation also seems unlikely to explain the difference, but the specific pair with the largest difference was not analyzed in replicate by either lab. Field sampling variation (more likely with total phase samples) might also contribute to differences in inter-lab splits, which are taken sequentially in the field rather than by truly splitting a larger sample. Again, aside from the poor agreement on one pair, the results show no clear bias among labs and do not require any adjustment.

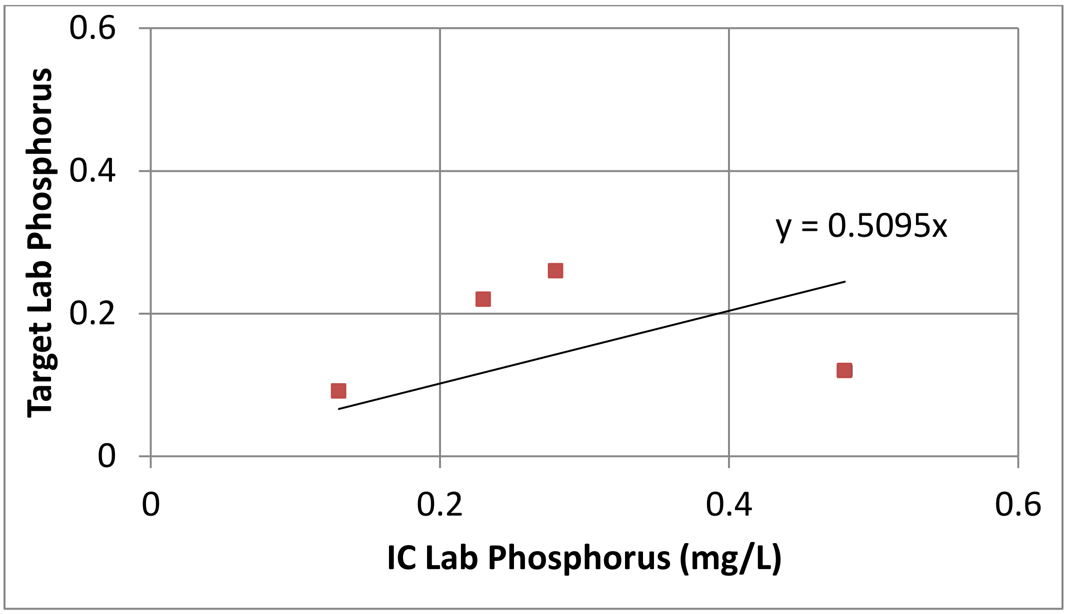

Figure 18. Target versus IC lab orthophosphate in split water samples for 2013 to 2014.

Pyrethroids

Pyrethroids were measured by Caltest (the "Target Lab") in 2013 and 2014, and by Axys Analytical (the "IC Lab") in previous years. Three water samples were split and analyzed for pyrethroids by both labs in the course of the study. For both labs, none of these split samples were analyzed as lab replicates. Some field replicates were analyzed by the target lab with RSDs 31% or better for analytes detected over 3x the MDL; the IC lab reports an ongoing precision and recovery (LCS) sample replicated across batches, with recovery errors 23% or less in 2014 samples. Only three analytes were detected in at least two of the split samples: bifenthrin, deltamethrin/tralomethrin, and total cypermethrin (Figure 19). The target lab reported higher concentrations slightly over half the time, but the ratio of target to IC lab concentrations was highly variable between samples for any given analyte; 54% to 120% for bifenthrin, 38% and 86% for deltamethrin/tralomethrin, and 105% and 149% for total permethrin. These differences are equivalent to an RPD range of 5% to 90%; as would be expected, the worst correspondence occurred in lower concentration samples where the relative impact of a nominal difference is larger. A larger number of samples would be needed to state with certainty, but within this small set of samples there does not appear to be any consistent bias, with the few results with concentrations above 10,000 pg/L (10 ng/L) being generally very similar between labs.

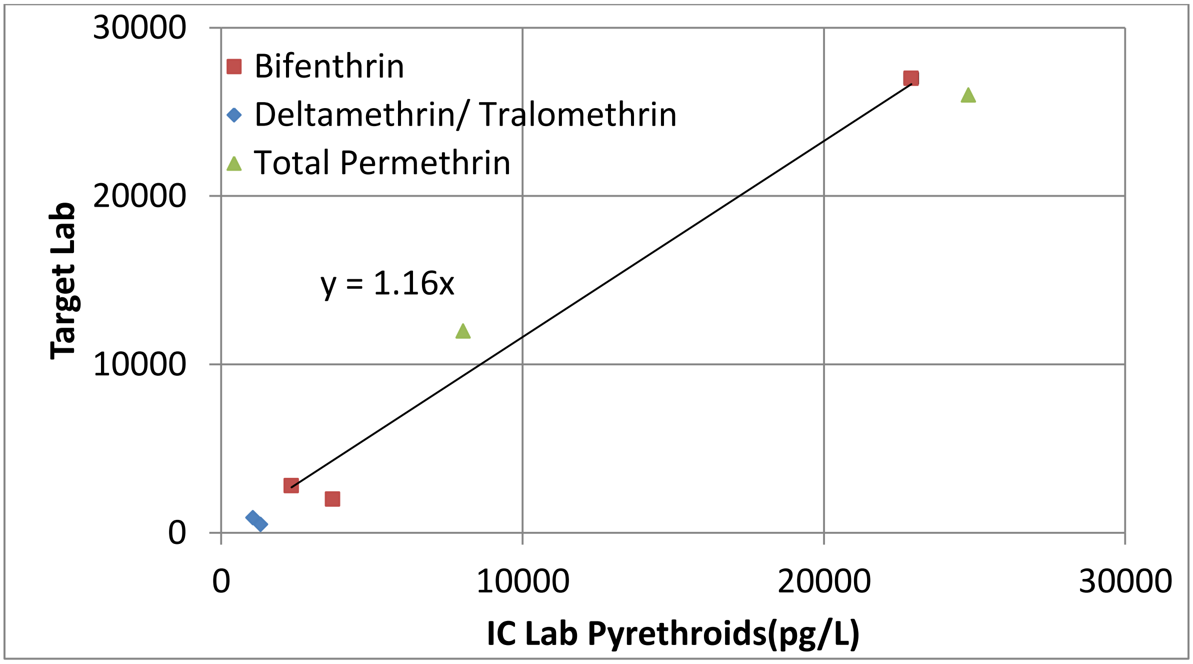

Figure 19. Target versus IC lab pyrethroids in split water samples for 2013 to 2014.

Overall the results of these intercomparison samples show general agreement between labs. For most analytes, there was not consistent bias between the labs; even where there seemed to be some bias, many of the results still showed nearly a 1:1 correspondence, so with the small number of split samples reported for most analytes, one or two random measurement errors could create the appearance of a net bias. If there are needs to more definitively quantify differences in sites or among events reported by different labs in different years, a greater number of split samples would be needed to assure a lack of bias from changing labs, but the current data suggest other than for sporadic excursions for individual samples, the data generally agree between labs, within the usual intra-lab acceptance ranges for precision and recovery for the various analytes. As noted before, most of the field sample data for this study will be considered in aggregated statistics, so even in cases where sporadic large differences appeared, the net impact will be small so long as these excursions are not the rule rather than the exception. Data subsampling techniques (e.g., including and excluding subsets of the best or worst data) can be used to further explore the need to reduce uncertainty of inter-lab differences for decision-making, before devoting time and resources to more rigorously quantify these differences.

Attachment 3. Additional Boxplots of Concentration Data

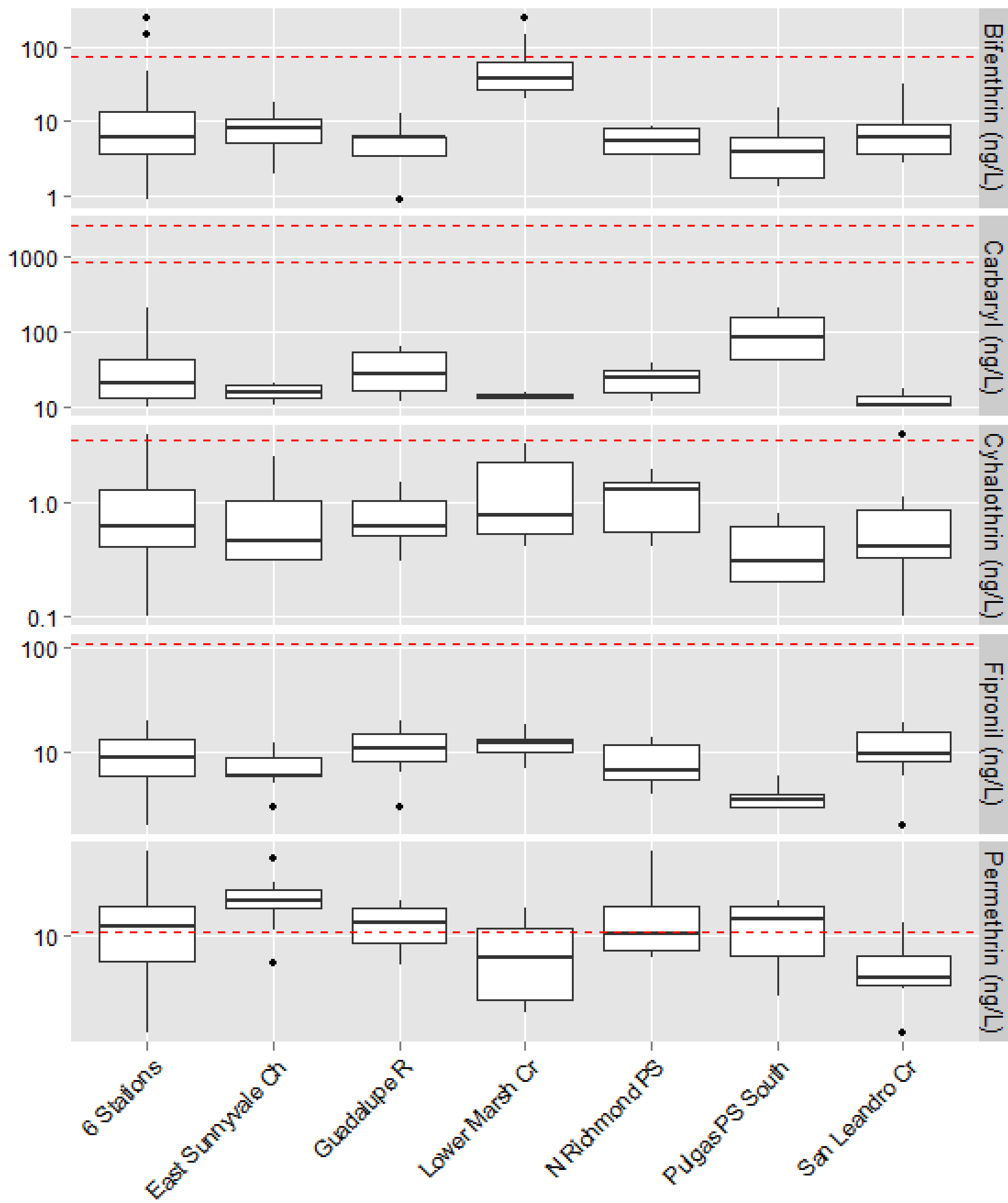

Figure 20. Summary boxplots of data collected across the six sampling stations for select analytes. Dashed red lines denote TMDL target, water quality criteria or other benchmark noted in Table 8.

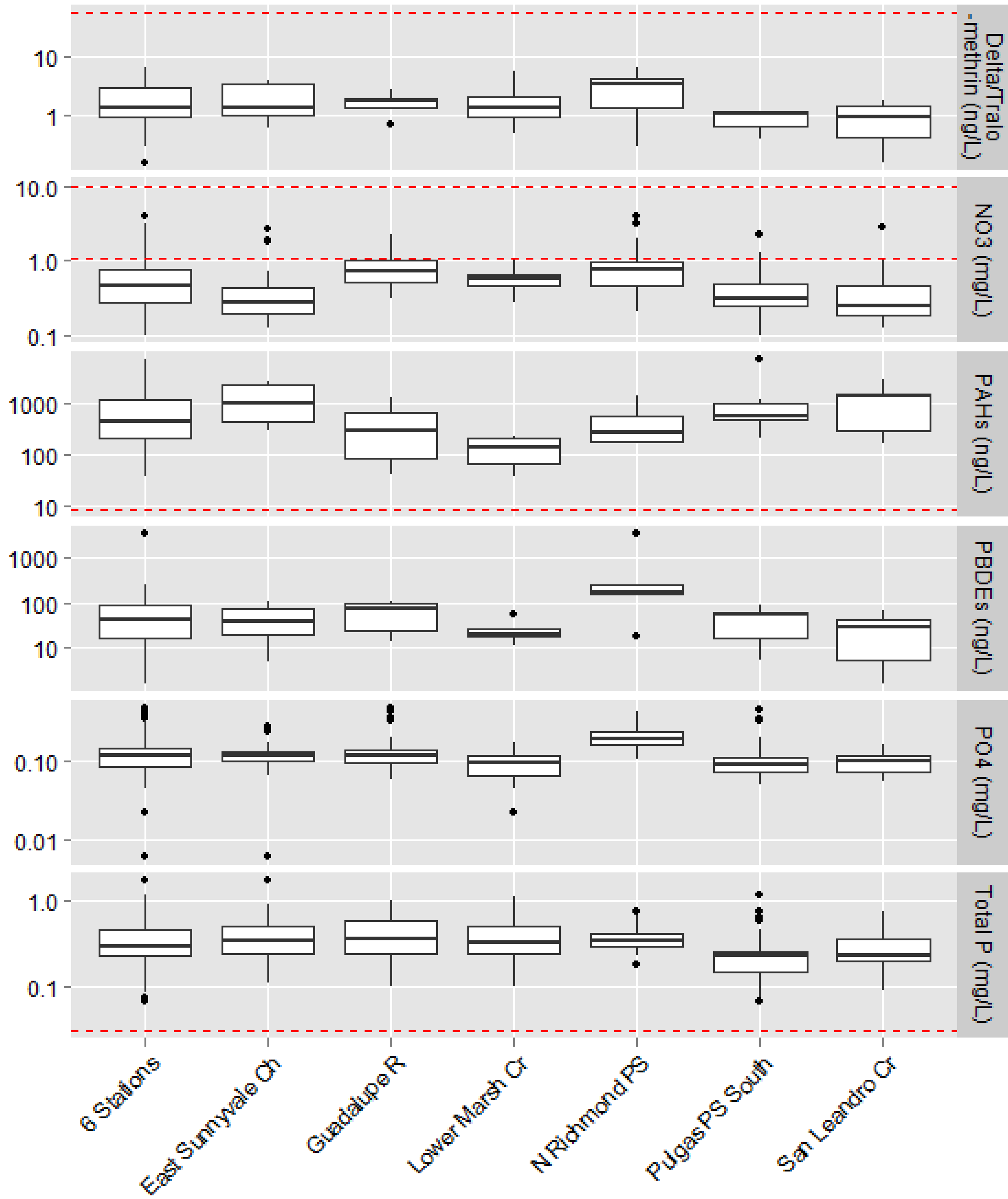

Figure 21. Summary boxplots of data collected across the six sampling stations for select analytes. Dashed red lines denote TMDL target, water quality criteria or other benchmark noted in Table 8.

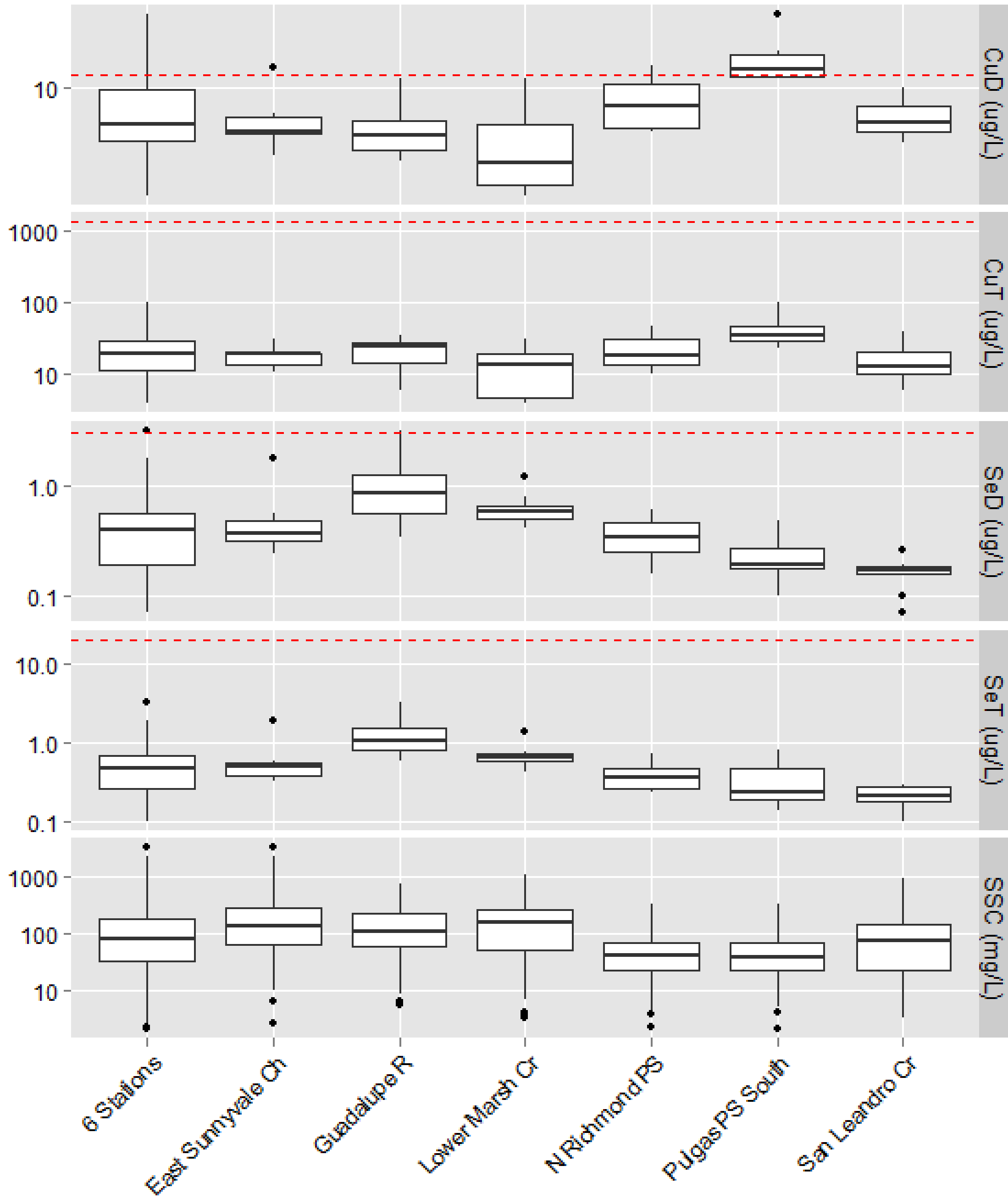

Figure 22. Summary boxplots of data collected across the six sampling stations for select analytes. Dashed red lines denote TMDL target, water quality criteria or other benchmark noted in Table 8.