# Bioretention Monitoring at the Daly City Library

Final Report

by

Nicole David, Michelle Lent, Jon Leatherbarrow, Don Yee and Lester McKee

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SAN FRANCISCO ESTUARY INSTITUTE

7770 Pardee Lane, Second floor, Oakland, CA 94621

p: 510-746-7334 (SFEI), f: 510-746-7300, www.sfei.org

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# Bioretention Monitoring at the Daly City Library
# Final Report

**David, N., Lent, M., Leatherbarrow, J., † Yee, D., McKee, L.**

† Deceased August 6, 2010

Summary

The negative impacts of urbanization and increasing impervious surface area on water quantity and quality have been well documented (Dietz and Clausen 2006, USEPA 2002). The objective of this study was to evaluate the effectiveness of a bioretention system consisting of four rain gardens and one bioswale to reduce peak flow and treat stormwater runoff from a parking lot and recreation area in Daly City, CA, USA. Stormwater flow and concentrations of selected metals, organic contaminants, and suspended sediment in stormwater were measured before and after the construction of the gardens. Our ability to precisely quantify the effectiveness of the bioretention system was difficult because of rainfall variations over the two winters and uncertainties in the way rainfall triggered the transport of contaminants from sources such as atmospheric deposition, structural degradation (buildings and pavement), car emissions and supply line leaks, and trash. Nevertheless, contaminant load reductions (estimated instantaneous loading rate) between 59% for HgT and over 90% for PAHs, Zn, Cu, and Cd were observed during the first year after construction. Contaminant concentration reduction and particle concentration reduction showed similar results indicating the successful removal of the monitored contaminants. The balance of evidence suggested an improvement in water quality due to the bioretention system installation at the Daly City site. The results indicated that such features, when designed and functioning correctly, can be highly effective in reducing pollutant loads in stormwater runoff.

Additionally, a simple model was developed to estimate the benefits of installing combination rain garden and bio-swale bioretention systems on a broader scale. The model provided baseline average annual stormwater loads for the local watershed, Colma Creek watershed, and for the region. Against this baseline, loads reductions were calculated assuming runoff from similar land use areas was treated in the same manner as the Daly City demonstration site. Several implementation scenarios were developed: 1) treat parking lots only, 2) treat streets only, and 3) treat all transportation-related land uses (e.g., highways, airports). To provide for a greater level of realism in the scenarios tested, feasibility parameters including slope and space (without conflicting uses) were included in the model. The implementation scenarios were assessed on the local watershed scale and the regional scale under theoretical/upper-bound conditions and under practical/realistic conditions. This study estimated that broadly applying bioretention, just one tool in the management or LID palette, could result in transportation land load reductions of 27-56% theoretically (i.e., not constrained by cost and space) and 5-9% practically. The model framework developed for this study could be used to estimate benefits from other LID techniques, e.g. permeable pavement, and to test which combinations of LID techniques among other management practices could be most advantageous and cost effective for local or regional scale application.

## Introduction

Stormwater runoff from streets and parking lots is known to cause water quality impacts in receiving water bodies (Dietz and Clausen 2006) and according to the Environmental Protection Agency (EPA) is the leading cause of impairments to the nation's waterways (USEPA 2002), deteriorating water quality at local and regional spatial scales. Compared to traditional stormwater treatment that usually only addresses peak flow rates, low impact development (LID) and redevelopment infrastructure can be used to mitigate runoff velocity, runoff volume, and water quality across a range of flow rates (Dunnett and Clayden 2007). LID implies an environmentally sensitive approach to site development and stormwater management that minimizes the effect of development on runoff by preserving the hydrologic function of a site. The potential for green infrastructure to trap contaminants can be attributed to the processes of adsorption, decomposition, volatilization, and ion exchange (Dietz and Clausen 2006). However, the functionality can degrade significantly due to poor design, poor initial construction, poor maintenance, or a combination of these factors. Despite these issues, rain gardens and grassy bioswales have been recommended as measures to reduce contaminants in runoff from building roofs and transportation infrastructure (Dunnett and Clayden 2007). Their usefulness under a wider variety of conditions and over the longer term has yet to be studied in more detail.

The Daly City bioretention system is just one of many LID projects of this kind that adds a green foundation to infrastructure improvements in the Bay Area. These systems show great potential in improving water quality and in providing important protection to San Francisco Bay. In this study, the pollutant removal capabilities and pollutant load reductions were investigated to quantify the effectiveness of bioretention systems and estimate the improvements for the receiving water body. The study utilized interpretation of field data using EPA recommended methods (described in Section 1 of the report) and a simple rainfall-runoff modeling technique to test and discuss the potential for wider application in the Bay Area (described in Section 2 of the report).

## *Section 1: Empirical Field Observations and Interpretation*

### Methods

#### *Daly City Site Description and Bioretention Cell Design*

This demonstration project was located at the main library in Daly City, San Mateo County, CA (Figure 1), and included 4,600 $ft^2$ (427 $m^2$) of a bioretention system comprised of four rain gardens and one bioswale. It was financed by the San Mateo Countywide Water Pollution Prevention Program through a vehicle registration fee increase of four dollars and the City of Daly City with the desire to improve water quality and beautify the library area. The design and construction of the bioretention system amounted to $380,000.


Figure 1. Location of the Daly City Library study site.

The drainage area is approximately four acres (16,200 $m^2$) and includes a parking lot (70% of the drainage area) and recreation area (basketball and tennis courts and a community area that together account for 30% of the drainage area), all of which are impervious. The drainage area represents a controlled system that does not receive runoff from any other areas in the vicinity such as residential driveways, yards, or homes. Following guidance provided by the San Mateo County Sustainable Green Streets and Parking Lots Design Guidebook (San Mateo Countywide Water Pollution Prevention Program 2009), the bioretention system accounts for approximately 3% of the entire drainage area and is divided into four separate retention cells (rain gardens) that receive runoff from different sections of the parking lot and recreation area (Figure 2). The site is heavily used year round with an estimated 20,000 visitors per month. This high use rate and the library facility give the project high public visibility and provide a platform for outreach and education.


Figure 2. Overview of study area at Daly City library.

The individual bioretention cells at this site were constructed with 2 inches (5 cm) of 3/8 inch (0.95 cm) gravel mulch over 15 inches (38 cm) of select filter media soil material over a pea gravel drainage gallery (Figure 3). The biofiltration soil blend contained green waste compost and had a loamy soil classification by USDA standards. The organic content was specified by the manufacturer at 5.3% and the percolation rate was designed to be 7.8 inches per hour (19.8 cm/hr). With a pH of 7.4, the soil mix was slightly alkaline designed to provide sufficient nutrients (nitrogen, potassium, copper, manganese, and sulfate) for optimal plant growth. Below the soil material, the pea gravel created protected space around a perforated pipe at the bottom of the cell that transports the filtered water off-site. The natural subgrade in most of the area is of high clay content and the installation of subdrains is essential for the system to work properly. The system predominantly discharges to downstream stormwater conveyance, but volume reduction is also promoted through infiltration below subdrains and soil soaking and drying. Only native plants were used for the rain gardens and bioswale on top of the different layers of filter media. Mostly wetland plants were employed, including *Juncus*, *Scirpus,* and *Stipa spp*. as well as various flowering species like *Iris, Mimulus, Leonotis, Narcissus, Lupinus, and Ceanothus spp.* The bioretention system required some irrigation during the summer months but the use of fertilizer was limited to the first year only.

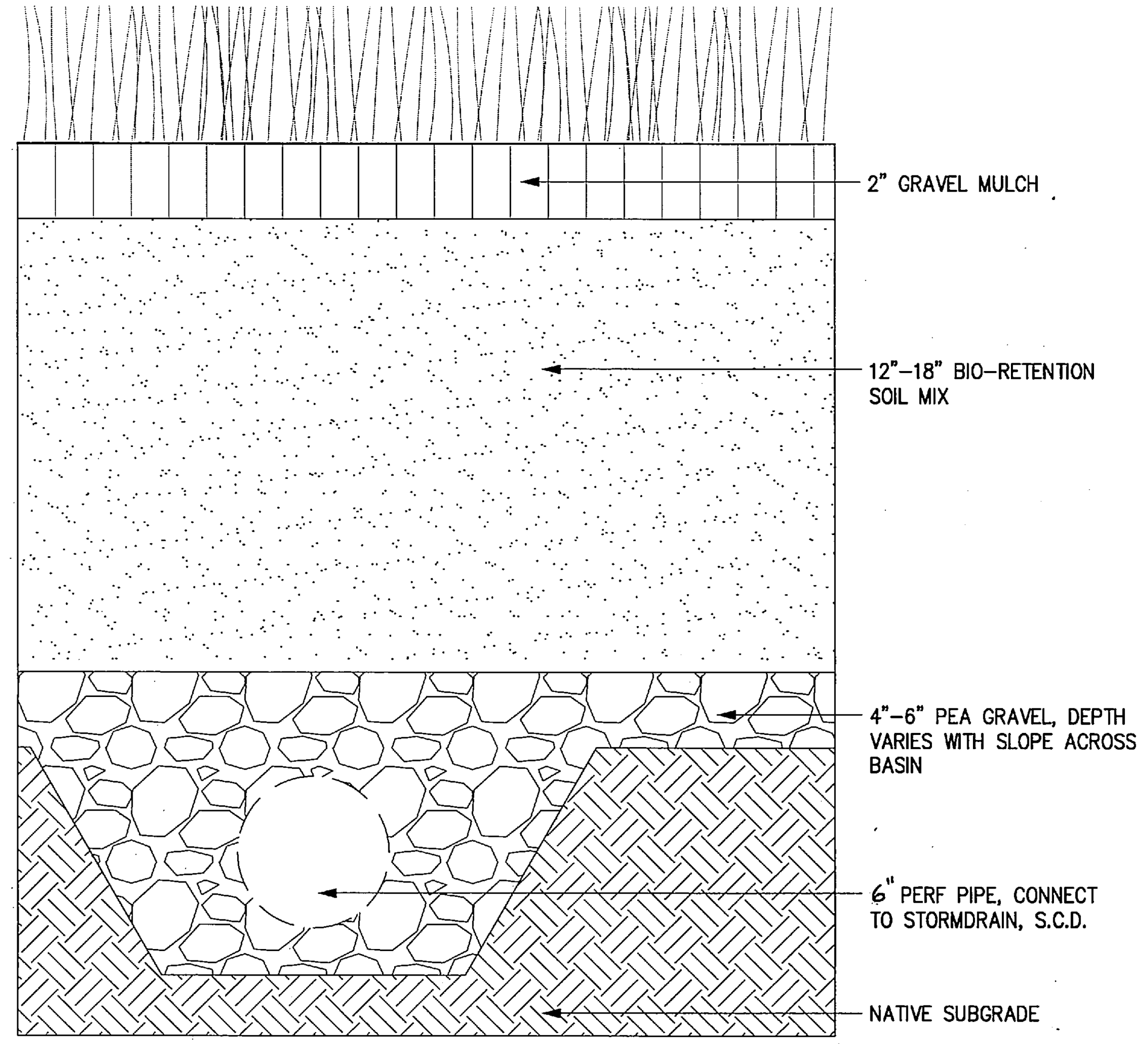


Figure 3. Design and layers of bioretention cells.

Unfortunately, trash caused some problems at the site when coffee cups and plastic bags clogged the inlets to one or more bioretention cells, and runoff was redirected to the remaining inlets. This occasionally compromised the capacity of the system as a whole since some of the cells were not filled with water while others were overflowing. When the rainfall intensity is too high and the water level in one cell reaches a certain elevation, water will spill into the adjacent cell via a drain connected to an unperforated pipe. When all cells reach their capacity, water spills over into the old stormdrain and is transported off-site without treatment to avoid flooding of the parking lot. The observed magnitude of a rainfall event that was above the capacity of the system was 0.2 inches per hour (0.5 cm/hr). Even though rain gardens have often been described as an additional tool for groundwater recharge (Dietz and Clausen 2006), the predominantly clayey soils in San Mateo County or the greater Bay Area with infiltration rates of 0 to 0.15 inches per hour (0 – 0.38 cm/hr) (Bay Area Stormwater Management Agencies Association 1999) likely do not allow much infiltration below the layers of the bioretention system.

During the planning phase of the project the engineers divided up the impervious area of the site into four main drainage management areas from which water drained into each bioretention cell due to the slope of the parking lot/recreation area (Figure 4). Responding to site topography and drainage patterns in the planning phase was important so that each bioretention cell in the final design received a flow volume proportional to its treatment capacity. Local rainfall data were used to calculate how much runoff each area would generate depending on imperviousness and slope, and the bioretention cells and positioning of the gutters and diversion channels were designed accordingly.

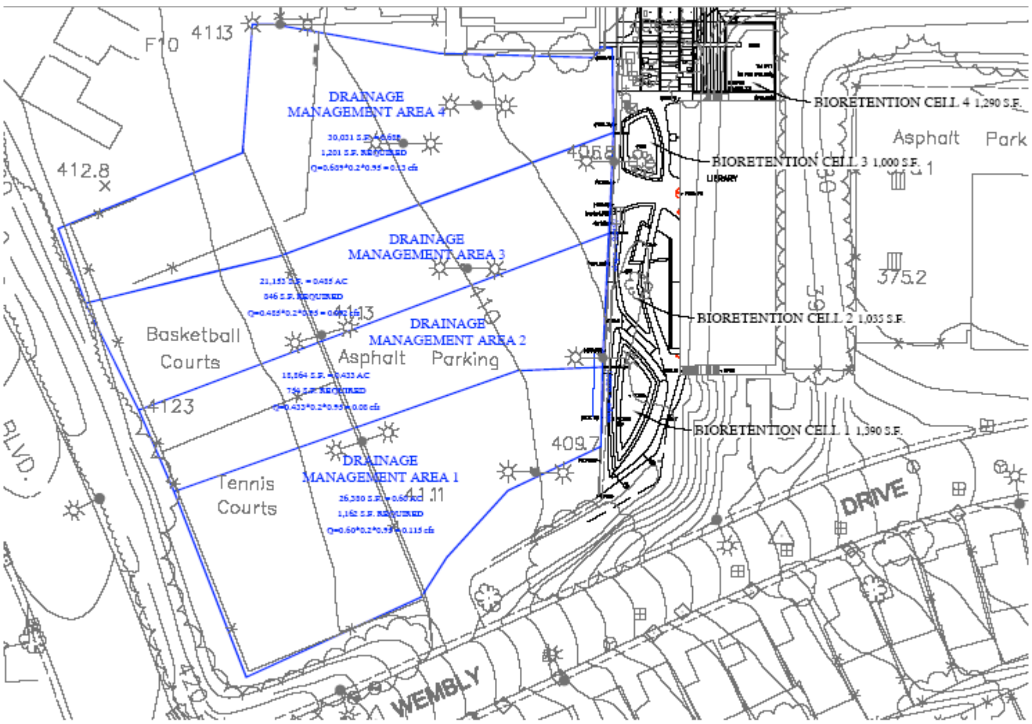


Figure 4. Drainage management areas of the Daly City library parking lot and recreation area.

*Field Methods*

Consistent with the objective of this study, we collected water samples during a variety of storm events (early and late in the season, high intensity storms and smaller rainfall events, saturated and unsaturated conditions of the bioretention cells) to study the site thoroughly before and after the installation of the bioretention system. A great amount of effort was invested in making qualitative and quantitative observations and capturing samples during different flow conditions (Table 1). Due to the imperviousness of the site the response time (time between the beginning of rainfall to the beginning of runoff in the pipe draining water away from the site) ranged from five minutes (before installation of the bioretention system) to 15-20 min (after installation), so the SFEI sampling team had to be ready on site when the rain started.

Table 1. All sampling dates for the first and second wet season of monitoring at the Daly City library.

| Dates | Times | Rainfall (inches) | Sampling Events | Stage | SSC (mg/L) | Storms | Average Flow (cfs) | Duration (hr) of Flow |
|---|---|---|---|---|---|---|---|---|
| Pre-bioretention | | | | | | | | |
| 3-Mar-09 | 22:15 | 0.08 | Not Sampled | - | - | - | - | - |
| 22-Mar-09 | 0:55 | 0.10 | 1 | peak | 14 | 1 | 0.03 | 10 |
| 22-Mar-09 | 2:09 | 0.01 | 2 | falling | 2.9 | | | |
| 7-Apr-09 | 10:00 | 0.03 | 3 | rising | 42 | 2 | 0.03 | 19 |
| 7-Apr-09 | 10:52 | 0.08 | 4 | peak | 26 | | | |
| 1-May-09 | 15:10 | 0.16 | 5 | peak | 26 | 3 | 0.03 | 11 |
| 1-May-09 | 16:11 | 0.04 | 6 | falling | 16 | | | |
| Post-bioretention | | | | | | | | |
| 20-Nov-09 | 12:05 | 0.46 | 7 | peak | 110* | 4 | 0.06 | 18 |
| 20-Nov-09 | 14:03 | 0.00 | 8 | falling | 39 | | | |
| 7-Dec-09 | 9:12 | 0.55 | 9 | falling | 12 | 5 | 0.08 | 5 |
| 16-Dec-09 | 8:20 | 0.12 | 10 | peak | 7.1 | 6 | 0.04 | 3 |
| 29-Dec-09 | 20:37 | 0.15 | 11 | peak | 11 | 7 | 0.02 | 11 |
| 30-Dec-09 | 7:23 | 0.02 | 12 | falling | 7.7 | | | |
| 12-Jan-10 | 2:03 | 0.32 | 13 | peak | 15 | 8 | 0.01 | 33 |
| 13-Jan-10 | 8:15 | 0.15 | 14 | falling | 8.8 | | | |
| 19-Jan-10 | 10:02 | 1.84 | 15 | falling | 6.9 | 9 | 0.02 | 170 |
| 20-Jan-10 | 6:40 | 1.28 | 16 | peak | 7.3 | | | |
| 31-Mar-10 | 2:24 | 0.05 | 17 | peak | 44 | 10 | 0.05 | 6 |
| 31-Mar-10 | 4:35 | 0.03 | 18 | falling | 4.5 | | | |

*bioretention system overflowed

*Water Sample Collection*

Water samples were collected at a depth of 4 inches (0.1 m) in the manhole that had access to the pipe discharging runoff from the sub-drains and the overflow pipes. The bottom of the manhole is approximately 5 inches below the pipe that transports water to the storm drain, so the water level in the manhole never rose much over 5 inches. A portable peristaltic pump was used to transfer well-mixed water from the manhole, using trace-metal clean tubing, into the sample containers. To avoid aerosol and contact contamination prior to sampling, the sample tubing and all containers for collection of trace metals were double bagged. The inlet of the sampling pump tubing was attached to a small double-bagged weight and deployed in the manhole. Before filling sample containers, tubing was flushed with site water for at least one minute. Each sample container was triple rinsed with site water unless the container contained a preservative. The containers were filled completely to eliminate any headspace, and care was taken to minimize exposure of samples to sunlight. Immediately after collection, the containers were closed and placed on ice in a cooler. Samples were shipped to and received at the laboratories in good condition (defined in the Quality Assurance Project Plan) between March 2009 and March 2010. All of the coolers containing water samples for trace metals and trace organic analysis were received at the lab at the recommended temperature of approximately 4ºC.

*Ancillary Measurements*
Dissolved oxygen, pH, temperature, specific conductance, and salinity were determined with a multiparameter water quality meter (e.g., WTW Multi 340, Weilheim, Germany). At a minimum, surface readings were taken at the 4 inches (0.1 m) sampling depth once during each sample collection. Turbidity was measured either in the field or in the laboratory with a HACH® 2100p Turbidimeter (Loveland, CO). Turbidity samples, if not measured in the field, were stored at 4°C and processed within two days of collection. Water flow was measured and recorded in the pipe that transports all runoff off-site with a Global Water Instrumentation, Inc. (Sacramento, CA) FL 16 water flow logger. Flow measurements were recorded in 1 min intervals for the entire storm duration (between four and 170 hrs) of a total of 10 storms. Three storms were monitored before the bioretention system was installed between March and May 2009 (a total of six samples) and seven storms were monitored after installation was completed between November 2009 and March 2010 (a total of 12 samples) (Table 1).

*Analytical Methods and Quality Assurance*
Water samples were analyzed for total PCBs, total PAHs, and dioxins/furans by AXYS Analytical Laboratories. The East Bay Municipal Utility District laboratory analyzed samples for gasoline, diesel, motor oil composites (C21 – C32), biological and chemical oxygen demand, and suspended sediment concentration (SSC). Concentrations for total mercury (HgT), dissolved mercury (HgD,) total methylmercury (MeHgT), cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) were analyzed by Brooks Rand Laboratories.

PAHs were analyzed using high resolution gas chromatography/ low resolution mass spectrometry (HRGC/ LRMS) by AXYS Analytical (Sidney, BC Canada) method MLA-021, a variant of EPA Methods 1624 and 8270. Samples were spiked with a suite of deuterated surrogate standards and solvent extracted. Extracts were reduced in volume, solvent-exchanged to hexane, treated for sulphur and columned on deactivated silica. The extracts were spiked with a labeled recovery (internal) standard prior to instrumental analysis. PAH concentrations were analyzed in extracts using HRGC/LRMS performed on an Agilent 6890N GC / 5973 MS / 7683 Autosampler. A Restek Rtx-5 chromatography column (30 m, 0.25 mm internal diameter (i.d.), 0.25 mm film) was coupled directly to the MS source. The MS was operated at a unit mass resolution in an electron ionization (EI) multiple ion detection (MID) mode, acquiring two characteristic ions for each target analyte and surrogate standard. A splitless/split injection sequence was used.

Samples were analyzed for 40 PCB congeners by AXYS Analytical Method MLA-010, with lab specific modifications to EPA Method 1668 Revision A. Samples were spiked with isotopically labeled surrogate standards, solvent extracted, reduced in volume, and cleaned up on a series of chromatographic columns, which may include silica, Florisil, alumina, carbon/Celite and gel permeation columns. The final extract was spiked with isotopically labeled recovery (internal) standards prior to instrumental analysis. Analysis of the extract was performed on high-resolution mass spectrometer (HRMS) coupled to a high-resolution gas chromatograph (HRGC) equipped with a SPB-Octyl chromatography column (30 m, 0.25 mm i.d., 0.25 µm film thickness).

Dioxins and furans were analyzed by AXYS Analytical Method MLA-017, equivalent to EPA Method 1613B with some lab-specific modifications. Samples were spiked with a suite of isotopically labeled surrogate standards prior to analysis, solvent extracted, and cleaned up through a series of chromatographic columns that may include gel permeation, silica, Florisil, carbon/Celite, and alumina columns. The extract was concentrated and spiked with an isotopically labeled recovery (internal) standard. Analysis was performed using a high-resolution mass spectrometer coupled to a high-resolution gas chromatograph equipped with a DB-5 capillary chromatography column (60 m, 0.25 mm i.d., 0.1 µm film thickness). A second column, DB-225 (30 m, 0.25 mm i.d., 0.15 µm film thickness), was used for confirmation of 2,3,7,8-TCDF identification.

Gasoline in samples was analyzed by EBMUD Organics SOP #333, generally following the California LUFT Manual guidance, using purge and trap concentration with GC/MS quantitation (EPA Method 8015). Samples were collected in pre-acidified sample vials and sealed. Surrogate and internal standards were added by the autosampler system, then purged and trapped on a Tekmar Dohrmann 3000 or Teledyne Tekmar Velocity system equipped with a Supelco Trap K (VOCARB 3000). Samples were then thermally desorbed and quantified on a Varian Saturn 2100T GC/MS/MS with a capillary GC column (Restek TRX-624 or equivalent).

Diesel and motor oil range organics were analyzed by EBMUD Organics SOP #336, using extraction by EPA Method 3520C, with GC/MS separation and quantitation. A measured volume of sample, usually 1 liter was added to a one step extractor. Surrogate/internal standard was added to each sample. The samples were then extracted with methylene chloride for 5.5 hours, and concentrated by drying to a final volume of 1 mL. Analysis was then conducted on a Saturn 2100T GC/MS with a split/splitless injector, equipped with a J&W DB-5ms capillary GC column.

Biochemical oxygen demand (BOD) was determined by Standard Methods 5210 B (20th Edition), which measures the oxygen utilized during a five-day incubation period for the biochemical degradation of organic material (carbonaceous demand) and the oxygen used to oxidize inorganic material such as sulfides, ferrous iron, and reduced forms of nitrogen (nitrogenous demand) unless an inhibitor prevents their oxidation. Known amounts of sample, seed, and dilution water were added to BOD bottles. The initial dissolved oxygen (DO) was measured and recorded. The prepared samples were incubated at 20°C for five days ± 2 hours. The DO measurements after the incubation period were taken and the BOD calculated.

Chemical oxygen demand (COD) was quantified by Standard Methods 5220 D. Samples were introduced to sample vials containing a premade mixture (Hach COD digestion vials, high range) of sulfuric acid, mercuric sulfate, chromic acid, and silver sulfate. Capped vials were heated in a block digester at 150°C. After two hours, vials were removed from the digester, cooled, and measured spectrophotometrically at 600 nm.

Suspended sediment concentration (SSC) was determined by ASTM D 3977. Samples were filtered through tared Gooch crucibles containing glass fiber filters, with a deionized water rinse of the sample container to remove adsorbed particles, and three 10 ml rinses of the filter to remove entrapped dissolved solids. Crucibles were dried overnight at 103°C. The increase in the weight of the crucible represents the suspended sediment in the sample, which was divided by the initial sample volume to obtain the suspended sediment concentration.

Concentrations of total and dissolved mercury in water were analyzed by Brooks Rand Laboratories using BR-0006, a lab specific variant of EPA Method 1631 Revision E. Dissolved mercury samples were filtered in the field using an acid-cleaned 0.45 µm polypropylene capsule filter in-line on the outlet of the peristaltic pump. All mercury species in the samples were converted to $Hg^{2+}$ by addition of excess BrCl. Mercuric ions in the samples were reduced to Hg(0) with stannous chloride ($SnCl_2$), and then purged onto gold-sand traps or gold wire traps as a means of pre-concentration. Trapped Hg was then thermally desorbed, and transported by carrier gas into a fluorescence cell for quantitation.

Methylmercury samples were analyzed by Brooks Rand Laboratories method BR-0011, a lab specific variant of EPA Method 1630. Samples were acidified to a final concentration of 0.4% v:v hydrochloric acid (HCl). Methylmercury samples were stored in the dark at 4°C until analysis. Sample aliquots were distilled to pre-concentrate samples, distillates collected, and ethylated using sodium tetraethyl borate, purged from solution onto a graphitic carbon trap, then thermally desorbed, with detection and quantification by CVAFS.

Concentrations of cadmium (Cd), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn) were analyzed by Brooks Rand Laboratories using BR-0060, a lab specific modification of EPA Method 1638. Samples were first digested in a closed vessel in the presence of strong nitric acid in a 85°C oven. Particulates were allowed to settle or were centrifuged to remove from suspension, and the extract run on a Perken Elmer ELAN DRC II ICP-MS (dynamic reaction cell inductively coupled plasma mass spectrometer).

*Data QA/QC*

Eighteen whole water grab samples were collected and analyzed (along with up to two field replicates and several lab replicates) for seven trace elements (Cd, Cu, Pb, Hg, MeHg, Ni, and Zn). Seventeen field-filtered (dissolved) samples were also analyzed for Hg along with two replicate field samples. Certified Reference Material (CRM), Matrix Spike (MS), and Blank Spike samples were run to evaluate accuracy. Lab and field replicates were run to evaluate precision, and lab blanks were run to screen for contamination.

This dataset showed no QA/QC issues in trace element analyses. Sensitivity was sufficient for all field sample results to be reported above the detection limits. Trace element results were blank corrected and the blanks subtracted were consistent with the variability (stdev) of blanks below the MDL. Lab replicates were run for most trace elements except Hg and MeHg, for which field replicate samples were evaluated instead. Precision was good for all trace elements with the average lab and field replicate relative standard deviations (RSDs) well below the target objective of 25%. CRM and MS samples run with these datasets had generally good accuracy results. The average error percentage for both CRM and MS samples were well within the target of 25% (75-125% recovery). MeHg accuracy was reviewed based solely on the MS results as no CRMs were reported for that analyte. No data were qualified for accuracy issues based on this review. The results for mercury dissolved and total fractions reported were also internally consistent, showing the total concentration higher than the dissolved fraction concentrations, as would be expected.

AXYS Analytical reported a total of 18 field grab samples of water for water organics (PAHs, PCBs, and Dioxins), and EBMUD reported samples for gasoline, diesel, and motor oil range organics. Lab blanks, field blanks, blank spikes, and matrix spike samples were also reported. In addition, field replicates were analyzed and lab replicates were run on several QA samples. Note that one PAH sample arrived at the lab broken and was not processed.

The PAH, PCB and Dioxin/Furan data from AXYS Analytical were in accordance with QA/QC requirements. Sensitivity was generally good for the PAHs and PCBs with the majority of analytes detected in most samples, with exception of Dibenz(a,h)anthracene; over 50% of results were not detected. The motor oil composite (C21-C32) reported by EBMUD also had more than 50% of the results reported as non-detect (ND). Four of the 17 Dioxin compounds were frequently detected (HpCDD, 1,2,3,4,6,7,8-; HpCDF, 1,2,3,4,6,7,8-; OCDD, 1,2,3,4,6,7,8,9-; and OCDF, 1,2,3,4,6,7,8,9-; MDL was 0.47 pg/L), while the remaining compounds were rarely detected. However, frequent non-detects are not unusual for dioxins and furans in environmental samples from this region (personal communication with Don Yee, SFEI). Blank contamination was evaluated for all analytes; many of the analytes were found at low levels in blanks, but given typically very low environmental concentrations, blank contamination sometimes comprised a major portion of the overall analyte measured in samples. Samples in which the blank contamination accounted for over 1/3 of any given analyte were censored and not reported for that analyte.

Field samples were generally not large enough to analyze in replicate for organic analyses, and this project's field samples were collected during runoff events that were expected to have a lot of variation over time, so blank spike samples were deemed most suitable for evaluating analytical precision. Blank Spike results were reported at 50 times their respective MDLs for all analyte groups except the Alkylated-PAHs. The average RSDs for Blank Spikes were within the organics target of 35% except for OCDD, and OCDF, which had average RSDs of 36% and 47% respectively and were each qualified but not censored. Accuracy was evaluated using matrix spike samples with average percent error for all reported analytes which were below the organics target of 35% error. Blank spike errors also averaged below the target of 35% for all reported analytes. No results were qualified for accuracy issues. It was not possible to compare the concentration range of this dataset to earlier site data as the site was not previously studied.

## Results and Discussion

### *Precipitation*

The study started late in the wet season of 2008/09, and six samples were collected during that first wet season of the study, prior to installation of the bioretention system. Precipitation ranged between 0.08 and 0.2 inches (0.2 and 0.5 cm) over durations lasting between four and 23 hr for the events studied. The second year of this study was classified as an above normal wet year and brought much more precipitation to the study site. Precipitation in the second year ranged between 0.08 and 1.8 inches (0.2 and 4.6 cm), with storm durations lasting between 45 min to 17 hr. Twelve samples were collected after the installation of the bioretention system in the second wet season. The post-installation sampling included a wider range of storm types. A couple of second winter season storms had similar characteristics to the first winter season storms, with similar rainfall intensity and duration, and also similar duration of dry weather antecedent conditions before the beginning of the storm. The remaining storms sampled spanned a variety of weather characteristics.

The average precipitation during the first winter season of the study was 1.4 inches per month (3.6 cm) with a rainfall total of 14 inches (36 cm) during the entire first wet season. The winter of 2009/10 was much wetter and had an average precipitation of 2.4 inches per month (6.1 cm). The total amount of rain for the second winter of the study added up to 24.2 inches (61.5 cm), more than twice as much as the previous year. This much wetter winter was classified as a mild El Niño year. Winters, during the El Niño effect, are usually wetter winters in the southwest United States including central and southern California (NOAA 2000). The first monitored storm in November 2009 of the second winter season brought 0.5 inches (1.3 cm) of rain in approximately 45 min (the largest storm monitored at the study site) and caused the stormwater to overflow into the old stormdrain. The runoff was untreated and the contaminant results of this event were not included in the post-installation averages. It is estimated that the return interval of this storm was three to five years.

### *Flow*

Flow measurements in the outflow pipe ranged from below the 0.05 cubic feet per second (cfs) (0.0014 $m^3/s$ (cms)) detection limit of the flow meter to a maximum of 2.5 cfs (0.071 cms). The average flow for each of the 10 monitored storm events was measured between <0.05 cfs (0.0014 cms) in January of 2010 during a series of storms lasting 33 hr and 0.08 cfs (0.0023 cms) in December of 2009 during a storm lasting five hours. Storm events of similar magnitude showed only a slight decrease in overall flow volume after the installation of the bioretention system. However, the flow period was prolonged for all storms during the second year of this study with the bioretention system in place, allowing for absorption of contaminants to the filter media. The system demonstrated the ability to delay and reduce runoff peak flow velocities and volumes through infiltration and evapotranspiration. However, since the soil and the filter media stayed wet and often saturated (new rainwater started pooling quickly since pores between soil particles seemed to still be filled) when there were very brief time periods between storm events, only a slight decrease (approximately 10%) in flow volume was observed when compared to similar first wet season storms. It is likely that if the second wet season had comparable precipitation to the first wet season, flow volume reduction would have been greater than 10%. Additionally, the maturing of the plants in the bioretention system can be expected to result in a higher absorption of water and a further decreased flow volume in future years. Therefore, at this time we cannot say what the average performance of the system is with regard to flow volume reduction.

*Suspended Sediment*

The concentrations of suspended sediment ranged from 2.9 to 43 mg/L (average 21 mg/L) before the installation of the bioretention system and from 4.5 to 44 mg/L (average 15 mg/L) after the bioretention system was constructed. One sample that was collected during the larger storm, when the capacity of the bioretention system was exceeded and water spilled over into the old stormdrain, was measured at 110 mg/L SSC, but that event was not used in deriving the second season average. Average suspended sediment loads were calculated for the first and the second winter. Loads were estimated to be reduced by 84% even though it is likely that small amounts of sediment in the outflow originated from the bioretention soil mix that is not separated from the drainage gallery by any filter material or other fabric, rather than coming off of the parking lot. Some parking lot sediment was probably able to trickle through the filter media since it is assumed that water will form channels within the soil mix.

Since the second winter brought more frequent and more intense rainfall events, it can be assumed that the amount of sediment entering the system was higher than in the first year. Therefore, the achieved load reductions for sediment and associated contaminants may not even represent the full treatment efficiency of this bioretention system.

*Metals*

Pre-installation trace metal suspended particle concentrations in parking lot/recreation area runoff were high, averaging between 17 (Pb) and 660 times (Zn) higher than long-term average suspended particle concentrations measured in Central San Francisco Bay (SFEI 2010) (Table 2). This indicates that transportation infrastructure, like the parking lot/recreation area at the Daly City library, could potentially contributing to water quality problems in San Francisco Bay. For example, Hg is a major problem in San Francisco Bay because it accumulates to high concentrations in invertebrates and fish and impacts wildlife species and people who eat fish caught from the Bay. As such, Central San Francisco Bay is included on the 303(d) List for impaired water bodies for Hg. HgT concentrations ranged from 3.5 to 47 ng/L in runoff before the bioretention system was installed, measuring 50 times the particle concentration when compared to particles in San Francisco Bay (Table 2). MeHg particle concentrations were 17 times higher in the parking lot/recreation area runoff relative to San Francisco Bay. Average concentration for MeHg and HgD ranged from 0.19 to 1.5 ng/L and 2.4 to 43 ng/L, respectively.

Table 2. Average pre-installation particle concentrations for metals in runoff from Daly City site and from San Francisco Bay.

| | Cd | Cu | Pb | Ni | Zn | HgT | MeHg |
|---|---|---|---|---|---|---|---|
| Particle concentration (pre) (mg/kg) | 59 | 3,420 | 330 | 690 | 59,120 | 0.4 | 33 |
| Particle concentration in SF Bay (mg/kg) | 0.1 | 52 | 21 | 54 | 88 | 0.2 | 2.0 |
| Degree of elevation to SF Bay (pre) | 590 | 66 | 16 | 13 | 670 | 2 | 17 |

Post-installation results for trace metals were mostly lower when the majority of runoff water was captured by the bioretention system. The only exception was MeHg. The MeHg average concentration increased from 0.63 to 1.6 ng/L, perhaps due to methylation processes within the bioretention system. However, HgT concentrations decreased and ranged from 5.8 to 27 ng/L post-installation (Figure 5), and HgD concentrations ranged from 2.1 to 18 ng/L post-installation. Due to miscommunication during the construction phase, a subdrain was left out when the bioretention system was completed. The subdrains underneath the filter media are supposed to transport water out of the system fast enough so that the bioretention cells remain predominantly aerobic. The missing subdrain may have caused some anaerobic conditions to occur on the bottom of one cell, where water was not freely draining, or draining very slowly, which may have created conditions favoring MeHg production by bacteria. During rain events, anaerobic water from underneath the filter media could have possibly commingled with rainwater filtering through the system, causing the higher MeHg concentrations after the implementation of the bioretention system. MeHg production can vary tremendously over short time periods and the results of this study each represent only snapshots in time. However, the recommendation was made that the missing subdrain was added in December of 2009. If this correction occurs, it is possible that future sampling would show decreased MeHg discharge from this system. Unfortunately, the addition of a subdrain after project completion is expensive and the City of Daly City resolved the problem with an extended retaining wall on the site of the bioretention cell where the water was pooling. With this added stability access water is now draining through and overflow drain.

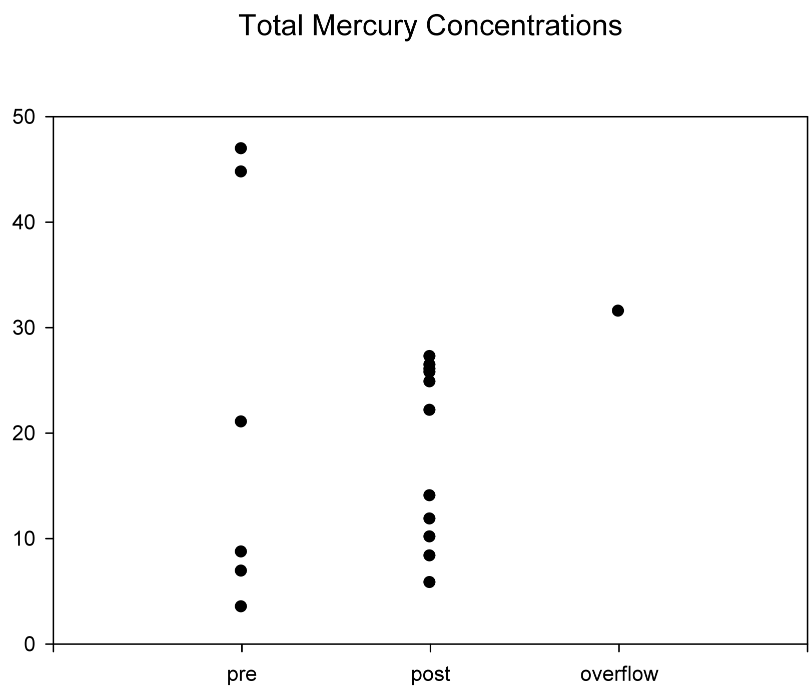


Figure 5. Total mercury concentrations before and after the bioretention system installation. Average HgT concentration before the installation was 22 ng/L and 18 ng/L afterwards.

Other metal concentrations were consistently lower after the bioretention system was installed. For example, average Cu concentrations decreased from 46 µg/L to 7.7 µg/L (Figure 6), and Zn concentrations dropped from 690 µg/L to 46 µg/L. Average Ni concentrations went from 15 µg/L before the system installation to 12 µg/L afterwards, while Pb concentrations decreased from 3.5 µg/L to 1.7 µg/L, and Cd concentrations dropped from 0.56 µg/L to 0.09 µg/L after the implementation of the bioreteneion system. For Cu and Zn, 83% (n = 6) of the pre-bioretention system samples were above the Criterion Maximum Concentration (CMC) for Water Quality recommended by EPA and only 8% (n = 12) for Cu were above the CMC after the biorentention system was installed. All post-installation samples were below the CMC for Zn. The recommended CMC for Cu is 13 µg/L for water hardness ranging from 100 to 130 mg/L, the CMC for Zn is 120 µg/L. The only Cu sample slightly exceeding the CMC after the bioretention system was installed was measured at 13.8 µg/L. All other monitored metal concentrations were below the recommended CMC during both years of this study. The CMC is an estimate of the highest concentration of a contaminant in surface water to which an aquatic community should be exposed. However, due to dilution the concentrations in runoff will be lower once the water reaches the receiving water body. Overall, the contaminant concentration reductions followed the order Zn>Cd>Cu>Pb>SSC>Hg>Ni. Considering that no filter fabric was used between the filter media and the drainage gallery, it is not surprising that SSC was not reduced more drastically. It is possible that sediment particles were added to the runoff from the soil mix during the second wet season storms.

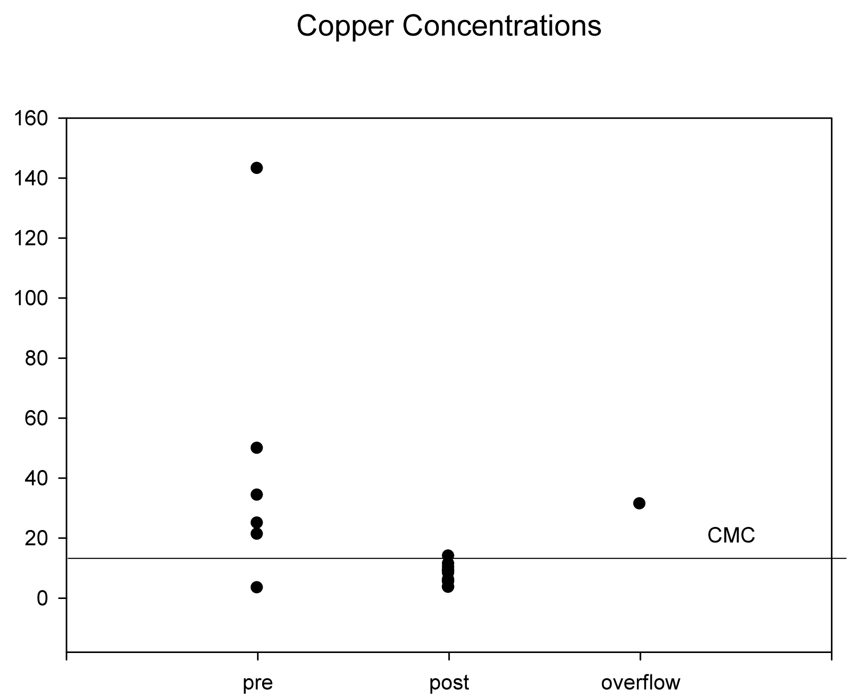


Figure 6. Total copper concentrations before and after the installation of the bioretention system. The line indicates the Criterion Maximum Concentration of 13 µg/L.

Previous studies examining prototype rain gardens (Davis et al. 2001a) have documented that mulch especially acted as a sink for trace metals, retaining up to 98%, 36%, and 16% of Cu, Pb, and Zn concentrations, respectively. Plants only removed between 0.1 and 0.2% of metals. In the Daly City study, the plants were just placed into the ground before the rain and the monitoring started in October of 2009. As the root systems of these plants develop in future years their metal retention potential may increase, since it will aid the physical particulate filtering process. However, according to the study of Davis et al. it can be hypothesized that if similar processes were operating at the Daly City site, metal removal in the Daly City bioretention system is primarily due to mulch or other layers of the filter media rather than by metal adsorption or uptake by the plants.

Compared to other studies investigating metal concentrations in stormwater runoff from pavement in residential areas (Legret and Pagotto 1999, Baeckstroem et al. 2003, Gnecco, et al. 2005), only Zn concentrations were higher in this two-year study (Table 3). Although Zn does not pose human health risks, it can be highly toxic to aquatic life. The ambient water quality guidelines for marine life are 90 µg/L and for freshwater aquatic life between 30 and 120 µg/L, depending on hardness (USEPA 2005). The extremely high concentration (2,500 µg/L) that was measured on April 7, 2009, during a moderate spring storm, was one order of magnitude higher than the average Zn concentration prior to the installation of the bioretention system and more than two orders of magnitude higher than the post bioretention system installation average. Possible sources of Zn include tire wear, wear of wheel balancing weights, leaking batteries, galvanized metal or scrap metal dumping, or dumping of Zn containing fluids, e.g., wood preservative. Other metals are contributed to urban runoff predominantly by vehicles (brakepads, tires, auto body wear, and various fluid leaks).

Table 3. Metals in runoff from paved surfaces.

| | This Study | France | Sweden | Italy | National Stormwater Quality Database |
|---|---|---|---|---|---|
| Usage | parking lot/ recreation area | motorway | highway | road | freeway |
| Size | 16,190 $m^2$ | 275 $m^2$ | 12 $m^2$ | 2,800 $m^2$ | NA |
| Range or average in | µg/L | µg/L | µg/L | µg/L | µg/L |
| Cd | 0.02-1.6 | 0.2-4.2±0.86 | 0.056-0.53 | NA | 1 |
| Cu | 3.3-140 | 11-150±27 | 39-130 | 19±20 | 35 |
| Ni | 0.73-49 | NA | NA | NA | 9 |
| Pb | 0.16-14 | 14-190±44 | 9.0-21 | 13±6 | 25 |
| Zn | 9.8-2,500 | 104-1,500±290 | 120-290 | 81±33 | 200 |
| Reference | | Legret and Pagotto 1999 | Baeckstroem et al. 2003 | Gnecco et al. 2005 | Pitt et al. 2004 |

Results ± SD when available

Concentration reduction on its own can be misleading in the evaluation of bioretention effectiveness because it is highly dependent on other factors (e.g., input concentrations, rainfall, SSC). There have been a number of methods proposed over the last several decades as measures of performance of BMP and LID infrastructure (Roseen et al. 2006). If sufficient numbers of samples are collected, the EPA recommends the Effluent Probability Method (EPM) (USEPA 2009). However, when paired data (at the inlet and outlet) are not available, other methods might need to be employed. For LID features, the EPA commented on the greater interest of before and after studies as employed in this study (USEPA 2009, page 9-16). In addition, given the promise of volume reduction as part of the treatment process in LID as compared to more conventional BMPs, the EPA also recommended using formal quantification of volume reduction as a key performance measure (USEPA 2009, page 9-18). In this study, due to a number of factors such as limited budget, insufficient time to collect enough samples prior to installation, and a different number of samples between the pre- and post-installation data set, data collected were not appropriate for use of the EPM. When data are not sufficient, the EPA recommended the use of multiple calculation methods to show the impacts of method choice on the evidence for specifying efficiency and providing some understanding of the ranges and trends in the evidence. In this study we used three methods of evaluation effectiveness that are described by the EPA and useful for building a weight-of-evidence but not recommended for use individually: Volume reduction, mean concentration, and loads. In addition, we use another method not discussed by the EPA but described by Whyte and Kirchner (2000) for an analysis of before and after concentrations at a mine remediation site near San Francisco that arguably is also suitable as part of the weight-of-evidence approach. To estimate the particle concentration of metals before and after bioretention system installation, correlations of metals to suspended sediment concentration were plotted as linear regressions assuming the dissolved phase is minimal and the unfiltered metal concentrations are strongly related to SSC (Figure 7). These assumptions are unlikely consistently true across the suite of contaminants or even for differing concentrations of a single contaminant, however, for the purpose of assessing performance, the assumption that the ratio of total contaminant concentration to SSC provides an estimate of particle concentrations is very useful. In this way we used many methods of evaluation as part of the weight of evidence rather than just concentration alone.

Evaluation of the degree of contamination of the particles in runoff provides insights into treatment performance. If it is assumed that the relationship between unfiltered metal concentrations and SSC was constant during all of the pre-implementation rainfall events and all of the post-implementation rainfall events, the slopes for the pre-and post-scenarios can be used to estimate the metal concentrations in relation to sediment particles. Changes in grain size during storms can cause a slight deviation from this linear relationship representing changes of energy in the system at different times of each storm. In the case of Cu (Figure 7), the pre-installation concentration was 3,420 mg/kg and the post-installation concentration was 30 mg/kg, a reduction of 99%. This difference was statistically significant with $p = 0.0006$, t-test ($n = 17$). The negative intercept can be explained with the large confidence intervals typically seen for regressions with small data sets; the 95% confidence interval of the regression y-intercept ranges from -78 to 25 and thus includes the origin. Similarly large ranges are seen for Cd, Pb, and Zn regressions. Another possible cause of deviations from linearity and the non-zero intercept in these regressions of metals to SSC are the non-uniform distributions of metal pollutants on road particulate matter; previous studies have generally shown increasing metal concentrations with decreasing particle sizes in stormwater runoff (Li et al. 2006, McKenzie et al. 2008). Different size fractions being preferentially transported in different parts of the hydrograph would lead to changing slopes for the aggregated data.

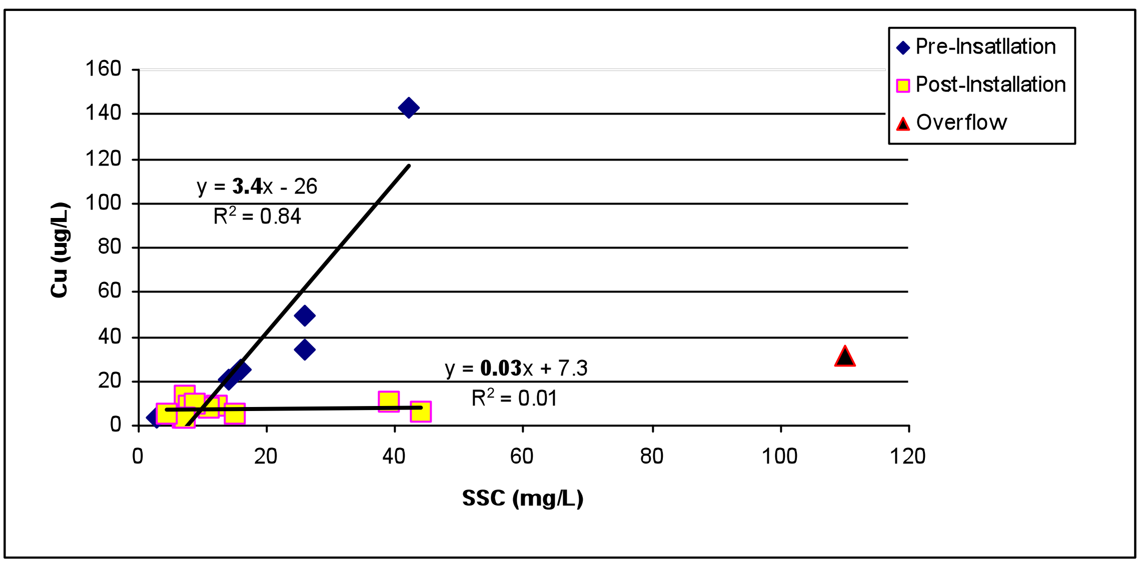


Figure 7. Correlation of suspended sediment concentrations and copper concentrations before and after the bioretention system installation. One high magnitude rainfall event is graphed separately because runoff spilled over into old stormdrain and left site untreated.

Using the same method to estimate particle concentrations for all monitored metals in this study, reductions ranging from 63% (HgT) to 99% (Zn) were calculated (Table 4). This estimation includes all but one post-installation data points collected during the study. The outlier not included in the particle concentration estimate was the sampling event on November 20, 2009, when water spilled over into the old storm drain, with rainfall exceeding 0.2 in/hr (0.5 cm/hr). The spill over water was untreated with high contaminant concentrations and very high SSC of 110 mg/L.

Table 4. Particle concentrations for before and after the installation of the bioretention system.

| Contaminant | Particle Concentration (pre) | Particle Concentration (post) | Estimated Reduction (%) |
|---|---|---|---|
| Cadmium | 59 mg/kg | 1.1 mg/kg | 98 |
| Copper | 3,420 mg/kg | 30 mg/kg | 99 |
| Mercury | 0.38 mg/kg | 0.14 mg/kg | 63 |
| Nickel | 690 mg/kg | No relationship with SSC | NA |
| Lead | 330 mg/kg | 5.0 mg/kg | 98 |
| Zinc | 59,120 mg/kg | 240 mg/kg | 99 |

However, most of the metals only have a strong relationship with SSC during the pre-construction phase and do not exhibit a correlation with SSC after the bioretention system was constructed. One possible explanation for that is that dissolved metals remained after the installation since the contaminant concentration relative to SSC was drastically reduced. Even if all particulate metals were successfully removed through the bioretention system there could still be dissolved phase contamination, which would not show strong correlation to SSC. This issue may be of concern to managers as dissolved phase metals and trace organics may be more bioavailable; treatment performance of LID systems in relation to dissolved phase should be a subject of future investigations. Also, the consensus of experts is that surface flows through treatment cells cannot be reduced in sediment and contaminant concentrations beyond a rather low level of irreducible concentration (Kadlec and Knight 1996).

Another index of bioretention system performance was the reduction of contaminant loads. The load estimates were based on instantaneous samples in a fast changing system and represent snapshots in time during different stages of storms. Load estimates were calculated by multiplying concentrations at a given time by instantaneous flow at a given time, using the following equation:

Contaminant Concentration (ng/L) x Flow (cfs) x (Unit Conversion Factor) 0.00245 = Load (g/day)

Averages of first order estimates of loads generated in this manner before and after the system installation suggest reductions of approximately 59% (HgT) to over 90% for PAHs, Zn, Cu, and Cd after the bioretention system was installed (Table 5). Although the method of calculation does influence the quantity of the reduction estimated, the conclusions for the effectiveness of the system were very similar regardless of the method used.

Table 5. Metals load reduction from before (pre) to after (post) the installation of the bioretention system at the Daly City library.

| Contaminant | Load (mg/min) pre | Load (mg/min) post | Estimated Reduction (%) |
|---|---|---|---|
| Cadmium | 0.146 | 0.0056 | 96 |
| Copper | 11 | 0.49 | 96 |
| Total Mercury | 0.003 | 0.0013 | 59 |
| Mercury dissolved | 0.000076 | 0.000028 | 64 |
| Nickel | 3.7 | 0.78 | 79 |
| Lead | 0.9 | 0.12 | 87 |
| Zinc | 170 | 4.0 | 98 |

*Organic Contaminants*

PCB contamination remains one of the greatest water quality concerns in San Francisco Bay, and the measured pre-installation estimated average particle concentration in the parking lot/recreation area runoff was 26 times as high (0.026 mg/kg) as the long-term RMP San Francisco Bay suspended particle average (SFEI 2010) assuming the majority of the measured PCB concentrations in the untreated pre-installation samples were mostly particulate phase. PAHs are included on the 303(d) List for several locations in San Francisco Bay, and concentrations along the western shoreline of Central Bay, the segment of the Bay that Daly City runoff drains into, have been the highest in the Bay for almost a decade. Average estimated PAH particle concentration in the parking lot/recreation area runoff was 44 times higher (89.7 mg/kg) than the average measured in Central Bay suspended particulates, again assuming the ratio of PAH concentration to SSC concentration is equivalent to particulate concentration in the pre-installation system.

After the installation of the bioretention system average PCB and PAH concentrations decreased substantially, from 730 pg/L to 410 pg/L and from 2,300 ng/L to 235 ng/L, respectively (Figure 8 and 9). Average PCB concentrations after the installation of the bioretention system were 44% lower than before the installation, with some indication that the higher-chlorinated PCBs were decreased more after the installation. This could be explained by the bioretention system more effectively removing the PCB congeners that are more strongly associated with particles. One sample (March 22, 2009) had a relatively high proportion of lower-chlorinated PCBs in the bioretention system effluent. This suggests mobilization of a source in the watershed with larger amounts of the less-chlorinated Aroclor mixtures (Aroclor 1242 and Aroclor 1248). The sample collected on January 12, 2010 (DC-13) had a similar profile to the March sample of the first wet season, suggesting a similar source or better removal of the higher-chlorinated PCBs that associate more strongly with particles. The sample collected during the November storm (DC-7) that caused the system to overflow had the highest PCB concentration by far, and also an unusual profile dominated by higher-chlorinated PCBs characteristic of Aroclor 1260. This could be due either to mobilization of a more highly-chlorinated source in the watershed and a bypass of the bioretention system during this overflow event. The most toxic PCBs (especially PCB 126) are moderately chlorinated and the moderately chlorinated congeners did not appear to be selectively affected by the bioretention system.

The overflow sample collected in November of 2009 is one of the most important samples of this study. It provides evidence that the catchment and sources of contaminants were very similar between the two winters of sample collection. It is truly the intervention of the bioretention system that caused the changes in concentrations observed. The November 2009 overflow sample (DC-7) is a good indicator of system performance as shown in Figure 10 and 11.

PAH concentrations were reduced on average by 90% after the installation of the bioretention system. There was a distinct general difference in PAH concentrations after installation, with the higher molecular weight PAHs consistently reduced more. This suggests that the bioretention system captured these more particle-associated PAHs more effectively than the more soluble low weight PAHs. Prior to the installation, the low weight PAHs accounted for 19% of the totals, and high weight PAHs accounted for 81%. After the installation, the low weight PAHs accounted for 44% of the totals, and high weight PAHs accounted for 56%. The PAH profile prior to the installation was dominated by high weight pyrogenic PAHs generated from combustion of fossil fuels. However, the bioretention system appeared to provide relatively effective treatment of the high molecular weight PAHs. This is important because many high molecular weight PAHs have been shown to be toxic to aquatic organisms (USEPA 2002).

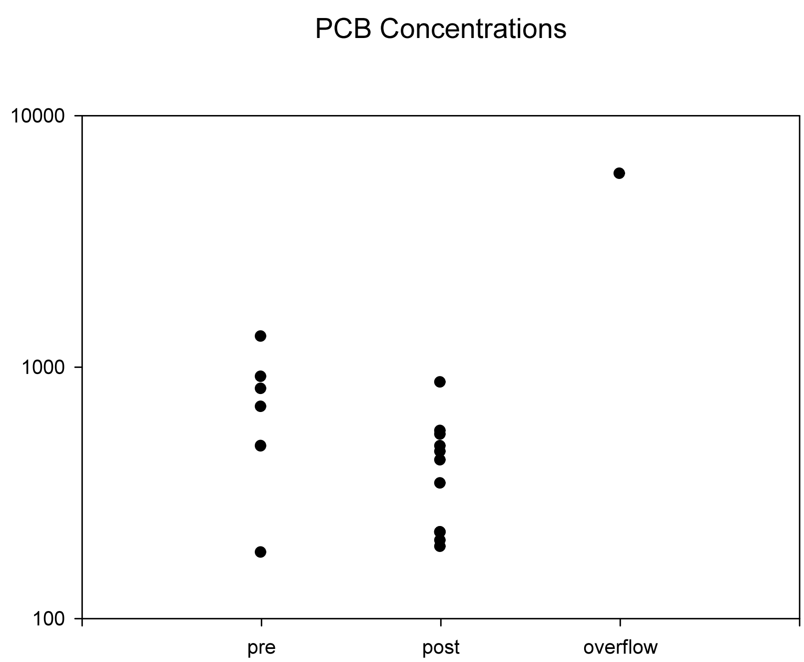


Figure 8. Total PCB concentrations before and after the installation of the bioretention system and from overflow sample when capacity of the system was exceeded and water spilled into the old stormdrain without treatment. Note log scale.

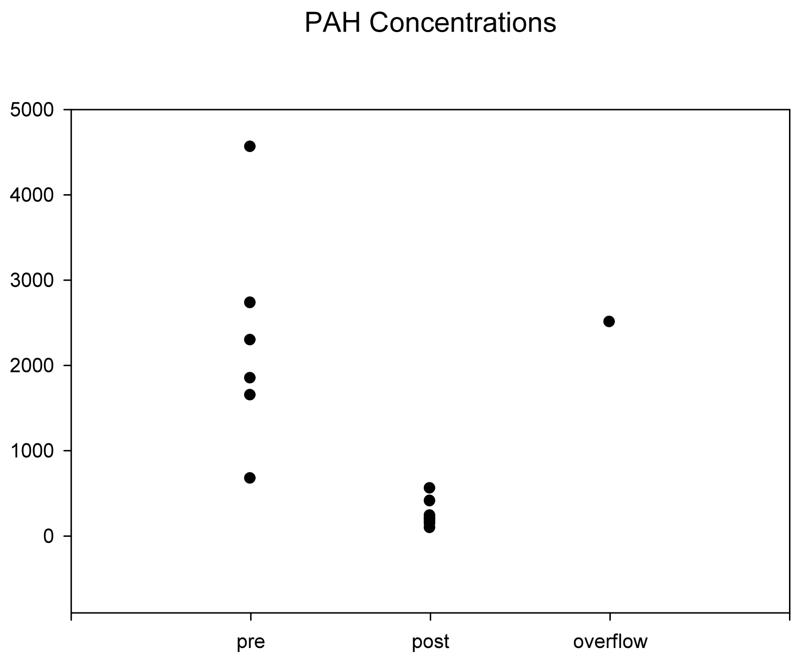


Figure 9. Total PAH concentrations before and after the installation of the bioretention system and from overflow sample when capacity of the system was exceeded and water spilled into the old stormdrain without treatment.

Again, in a similar fashion to the trace metals, assuming most PAHs and PCBs were in particle phase, the average PAH particle concentration ratio was reduced from approximately 90 mg PAH/kg SSC to 3.9 mg PAH/kg SSC, which is almost a 96% reduction (Figure 10). PCB particle concentration was reduced from 0.026 mg PCB/kg SSC to 0.004 mg PCB/kg SSC (Figure 11) during the second year of the study, a reduction of 85%. This suggests that contaminants were partitioning off of the suspended sediment particles and onto the filter media particles or that particles were trapped by the filter media completely and other cleaner particles were released into the draining water from the bottom layer of the filter media. The positive intercepts may indicate the portion of these organics that were in dissolved phase and if so, we might speculate that even some dissolved phase was captured by the system. Again, since dissolved phase contaminants may be more bioavailable, this component of treatment deserves more study. Regardless, the highly efficient removal rates for PAH that were observed in the Daly City bioretention system were consistent with other studies that examined influent concentrations and percent removal for contaminants in rain gardens (USEPA and ASCE 2002).

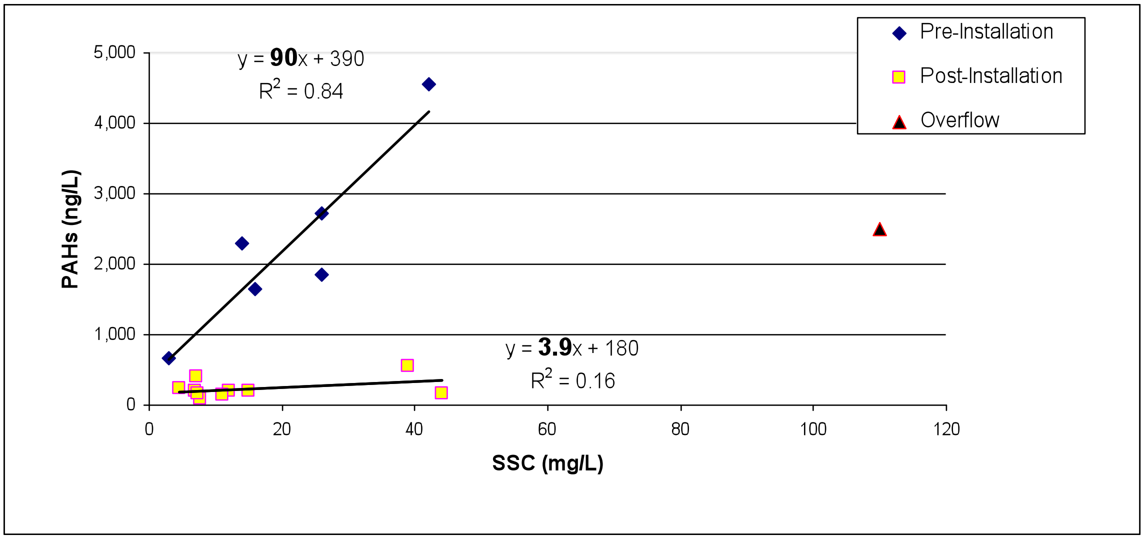


Figure 10. Correlation between suspended sediment concentrations and PAH concentrations. Pre- and post-installation results are statistically significantly different ($p = 0.001$, t-test). Graph was generated from all data points collected in this study ($n = 16$; one PAH sample broke during the transport to the laboratory). Overflow sample was not included in the regression data series.

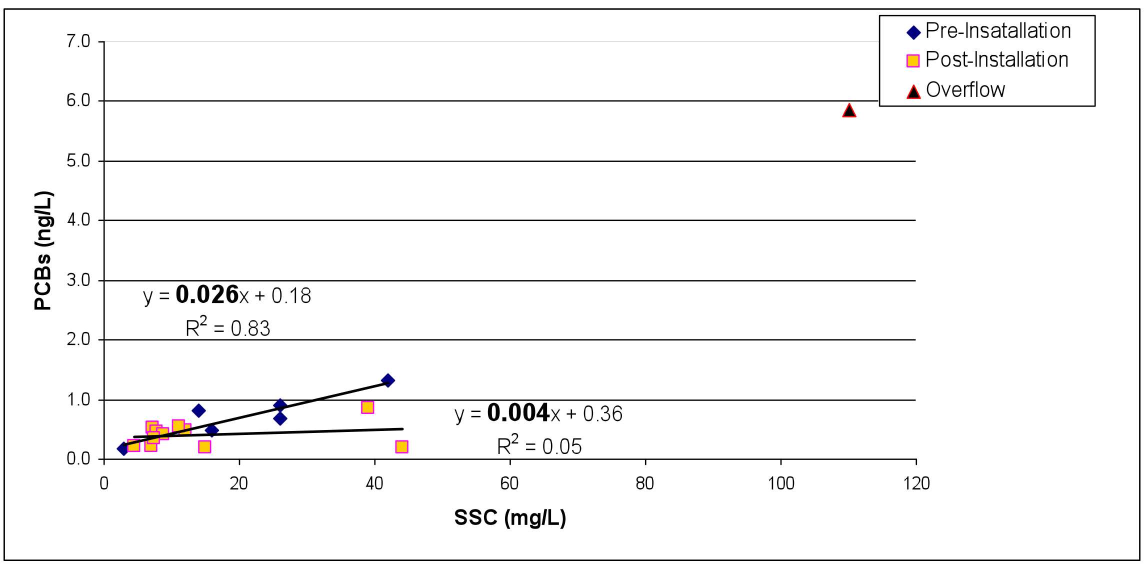


Figure 11. Correlation between suspended sediment concentrations and PCB concentrations. Pre- and post-installation results are statistically significantly different ($p = 0.009$, t-test). Graph was generated from all data points collected in this study, with the exception of the overflow sample.

The calculation of contaminant loads for PAHs and PCBs showed a reduction of approximately 82% for PCBs and 97% for PAHs in stormwater runoff (Table 6). Even though this calculation is based on instantaneous samples and only represents a snapshot of the overall storms, the results suggest that contaminants originating from a nearby source, e.g., PAH from engine combustion and weathered road surfaces, can be significantly reduced as the water infiltrates through the bioretention system, with concentrations in effluent remaining low in a variety of storm conditions. PCB particle concentrations were somewhat elevated in runoff before the bioretention system was installed relative to average particle concentrations in San Francisco Bay but did not indicate a large PCB sources in the parking lot/recreation area. Atmospheric deposition or particulate transport from either local or longer range sources is likely the main source of PCBs in the runoff from this parking lot and recreation area.

Table 6. Loads for PAHs and PCBs in runoff from the Daly City parking lot/recreation area before and after the installation of the bioretention system.

| Contaminant | Load (mg/min) pre | Load (mg/min) post | Estimated Reduction (%) |
|---|---|---|---|
| PAHs | 0.91 | 0.02 | 97 |
| PCBs | 0.000229 | 0.000042 | 82 |

Octa-chlorinated dioxin (OCDD) was the most abundant dioxin in runoff water, with the most toxic tetra-chlorinated dioxins not detected throughout this study. OCDD concentrations ranged from below the 6.8 pg/L detection limit to 110 pg/L before the system was installed. Only one pre-installation sample (April 2009) had a detected concentration of total tetra-furans with 0.56 pg/L. The post-installation results for OCDD were all below the method detection limit of 0.47 pg/L likely due to effective removal by the bioretention system, consequently loads were not calculated.

*Fuel and Fuel Additives*

All results for gasoline for pre- and post-implementation were below the detection limit for the analytical method. The toxic gasoline constituents benzene, toluene, and xylene, or BTX, were also not detected in the collected samples. This is in part likely due to the high volatility of these compounds and the relative lack of on site sources that would lead to measurable amounts in runoff. Diesel concentrations ranged from below the method detection limit of 20 µg/L to 5,000 µg/L before the construction of the bioretention system to 20 and 150 µg/L with the system in place (Table 7). The average of diesel concentrations dropped sharply from 1,700 µg/L to 37 µg/L. Acute toxicity of diesel fuel to the waterflea *Daphnia magna* was described by Das and Konar (1988) with an $LC_{50}$ value of 1,500 µg/L. The same study published an $LC_{50}$ of 2,000 µg/L for the mollusk *Viviparus bengalensis* and an $LC_{50}$ of 5,000 µg/L for *chironomid* larvae, indicating that there is a potential risk for aquatic organisms in the receiving water body when stormwater leaves the parking lot/recreation area untreated. Results for motor oil composites (C21-32) in water samples ranged from 330 to 6,800 µg/L before the implementation of the bioretention system to all results afterwards being below the method detection limit of 260 µg/L (Table 7).

Table 7. Average concentrations of motor oil, diesel, and chemical oxygen demand before and after the installation of the bioretention system.

| Contaminant | Average Conc. Pre | Average Conc. Post |
|---|---|---|
| Motor oil composites (C21-C32) | 2160 μg/L | <260 μg/L |
| Diesel | 1680 μg/L | 38 μg/L |
| Chemical Oxygen Demand | 289 mg/L | 64 mg/L |

*Ancillary Measurements*

In environmental chemistry, the chemical oxygen demand (COD) test is commonly used to indirectly measure the amount of organic compounds in water. The average COD concentration dropped from 290 mg/L before the bioretention system was installed to 64 mg/L after installation, indicating less organic material transported in effluent.

Dissolved oxygen concentrations in parking lot/recreation area runoff dropped from an average of 8.8 mg/L before the installation of the bioretention system to 7.0 mg/L after the installation, while the average temperature remained similar (12.7°C before and 11.6°C after). These results suggest that there may be some oxygen consumption in the bioretention system, probably due to slower flow allowing some consumption by materials with COD retained by the system, which might also explains the increase in MeHg due to methylation by bacteria in an anaerobic environment.

## Conclusion

Three different approaches were used in this study to evaluate the effectiveness of the bioretention system. Contaminant concentration reduction, particle concentration reduction, and load reduction all showed similar results indicating the successful removal of the monitored contaminants. Although there were differences in treatment efficiency for all contaminants depending on the tool of performance evaluation, it can be stated that even with the bigger storms occurring during the second winter, contaminant concentrations and loads were still smaller in that year likely due to treatment in the bioretention system. This also became apparent when event mean concentrations (EMC) and flow-weighted mean concentrations (FWMC) were calculated to characterize the contaminant load for the receiving water body (Table 8). The EMC is the mean concentration of a particular contaminant over the flow of the hydrograph, while the FWMC is the total load of a particular contaminant over the flow of the hydrograph. These are two common methods used to characterize contaminants in runoff over the course of a storm event. Additionally, the reduction of peak effluent concentrations due to the slower flows in the second winter resulted in a delayed and flattened hydrograph response and was probably as important of a benefit to the downstream water body as the concentration and load reduction of the contaminants. However, while Colma Creek is mostly culverted or channelized in concrete, bioretention systems resulting in flow and volume reductions may be more beneficial in watersheds with less hydromodification, consistent with municipal stormwater program permit requirements.

Table 8. Comparison of performance evaluation for average concentration (Conc.), particle concentration (Part. Conc.), average load, event mean concentration (EMC), and flow-weighted mean concentration (FWMC) expressed in % reduction from pre- to post-installation of the bioretention system.

| Efficiency | Conc. | Part. Conc. | Load | EMC | FWMC |
|---|---|---|---|---|---|
| Cadmium | 84% | 98% | 96% | 90% | 96% |
| Copper | 83% | 99% | 96% | 90% | 94% |
| Mercury T | 18% | 63% | 59% | 53% | 78% |
| Mercury D | 50% | NA | 64% | 47% | 55% |
| Nickel | 20% | NA | 79% | 55% | 74% |
| Lead | 51% | 98% | 87% | 70% | 86% |
| Zinc | 93% | 99% | 98% | 95% | 97% |
| PAHs | 90% | 96% | 97% | 93% | 97% |
| PCBs | 44% | 85% | 82% | 61% | 78% |
| SSC | 29% | NA | 84% | 49% | 82% |
| COD | 78% | NA | 93% | 86% | 92% |

The Daly City library bioretention system (rain gardens and bioswales) was highly effective in removing metals and organic contaminants from infiltrating stormwater. Contaminants that originate from local sources (vehicles and weathering pavement) and have relatively high concentrations in runoff, like most metals and PAHs, seemed to be effectively treated through the system. Pollutants originating more from atmospheric sources, like PCBs and Hg, and are not as elevated in stormwater runoff were not reduced to the same amount as locally contributed contaminants.

While PCBs were removed to a larger extent (approximately by 82%), HgT was only removed by an average of 59%, likely due to dissolved, colloidal, or ultrafine particulate transport carried in the slow and steady flow through the bioretention system. This finding is consistent with the observation made by Yee and McKee (2010) that in settling experiments a larger fraction of total PCBs (30-70%) settled out of suspension in stormwater within 20 minutes whereas a smaller fraction of HgT (10-13%) was removed within the same time. Since the bioavailablility of contaminants may be higher when they are in the dissolved phase, the ability for LID to remove dissolved phase is an area that deserves further research. The current results are very promising and suggest that unlike conventional structural BMPs, LID systems have at least moderate dissolved phase removal.

According to the Impervious Indicator Model (Figure 12), which describes the strong negative relationship between subwatershed impervious cover and various indicators of stream health, degradation of the receiving water body quality is highly correlated to increased imperviousness in urban areas (Finkenbine et al. 2000). LID measures like rain gardens and bioswales seem to mitigate the detrimental effects of stream degradation by removing contaminants and reducing both peak and total stormwater flows from parking lot and road runoff for precipitation events that do not exceed system capacity. Low flow events are treated especially effectively because the chemical and physical processes occurring in the bioretention system are optimized and adequate reaction time is provided for contaminant attenuation (Davis et al. 2003).

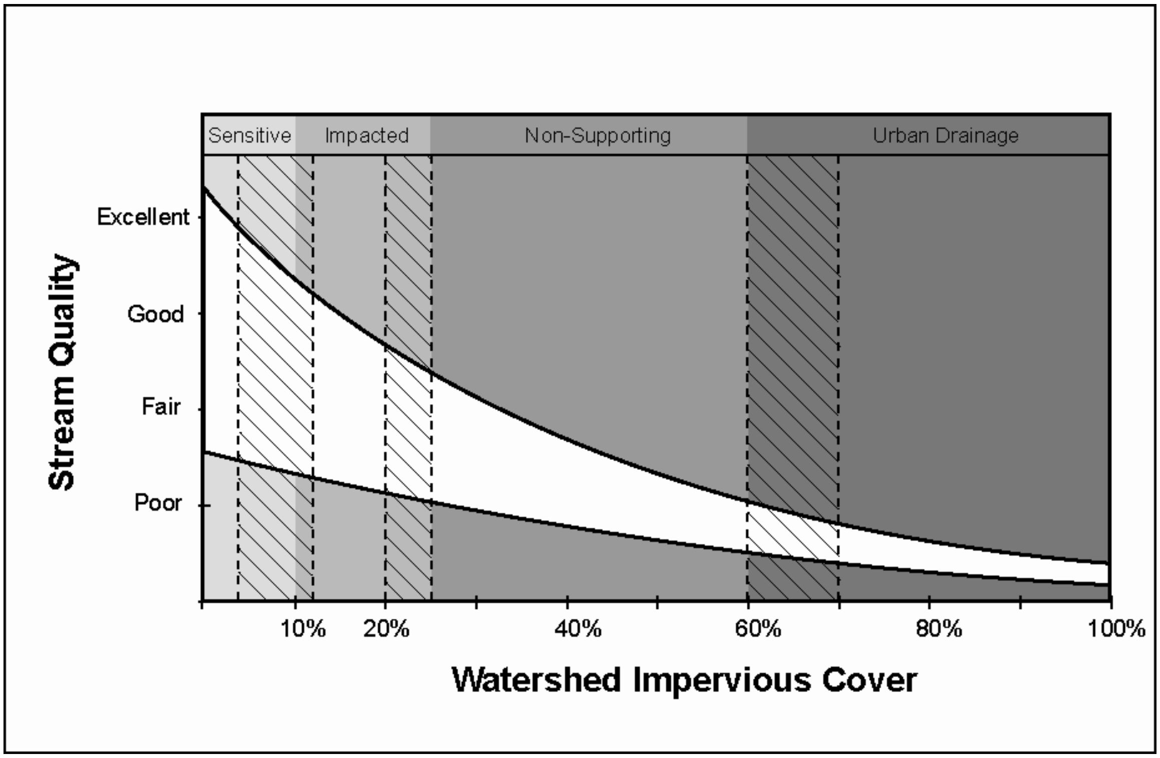


Figure 12. Impervious Cover Model by Chesapeake Stormwater Network 2009 http://www.chesapeakestormwater.net/all-things-stormwater/the-reformulated-impervious-cover-model.html

Mass balance studies in other systems suggest that the build-up of heavy metals will not pose a problem to the site for approximately 10-15 years (Davis et al. 2003, MacDonald et al. 2008). After that time there is the possibility of metal accumulation to a degree of concern and soil testing is recommended. Certain plants have exhibited the ability to hyperaccumulate heavy metals and could be considered for rain gardens with above average metal inputs for the continuous removal of metals (Wu et al. 1999 and Sarret et al. 2001), as long as the vegetative material does not become a significant food source for resident biota of concern.

*Lessons Learned*

The Daly City bioretention system effectively treated stormwater and removed contaminants for the majority of storms in 2009/10. The site was designed to treat runoff from the parking lot and recreation area during storms with rainfall less than 2 inches of rain per hour. Despite the fact that the curb-cut to the southernmost bioretention cell was frequently blocked by plastic bags and coffee cups, the system still performed at the desired level without the full benefit of one bioretention cell.

Future monitoring efforts at this site should consider sample collection at the north end of the parking lot before stormwater enters the biorentention cell and in the pipe where water leaves the site after treatment. Even though it is challenging to collect water running off the pavement directly and to measure flow this way, this approach would result in data that have less noise due to other factors, e.g., interannual rainfall variation. SFEI will likely have the opportunity to conduct a follow-up study at the Daly City library site in the winter of 2013/14 and with some site modifications could possibly change the sampling design to influent/effluent comparison.

Other recommendations for monitoring bioretention systems in a cost-effective way and obtaining long-term trends of monitored contaminants were made by Strecker et al. (2001). This approach combines chemical measures of effectiveness with physical habitat and biological assessments of the receiving water body. However, the latter may sometimes be influenced or vary by climactic or other factors (invasive species, disease) not directly related to LID application and would have to be interpreted carefully. Especially for the comparison of larger watersheds, this coupled approach would provide a better tool for performance evaluation of the LID. For the Daly City library site, the number of contaminants for long-term monitoring could be reduced to contaminants from local sources (e.g., PAHs and dissolved and particulate metals) that can describe the effectiveness of the system as well as its finite capacity. Additionally, a streamlined protocol for data collection would also produce measurable data that are reliable and comparable between states.

## ***Section 2: Application of Results. Estimating LID Benefits on a Broader Scale***

Methods

*Model Overview*

A simple rainfall/runoff model was developed for the San Francisco Bay Area region (Figure 13). The rainfall/runoff model assumes a linear relationship between annual stormwater volume and annual precipitation (Gunther et al. 1987; BCDC 1991; Maidment 1993; Davis et al. 2000), where a runoff coefficient determines the fraction of the precipitation that becomes runoff. Stormwater contaminant loads were calculated by multiplying runoff volume by average concentration of contaminant in stormwater runoff for each distinct land use type.

$$L = \sum C_j * r_j * I * A_j \qquad \text{Equation (1)}$$

where L = contaminant load, C = stormwater contaminant concentration for land use *j*, r = runoff coefficient for land use *j*, I = average rainfall, and A = area of land use *j*.

*Hydrologic Model Development*

Data from CALWATER (version 2.0) were used for model delineation (Figure 13). The spatial extent of the model was State Water Resources Control Board (SWRCB) Region 2. However, the model extent was modified to remove drainage areas greater than 20 mi$^2$ (52 km$^2$) behind dams from analysis, which resulted in about 20% of total area being excluded. The rationale was that significant retention of particles and chemical transformations occur in reservoirs (Davis et al. 2000). The remaining model extent was divided into hydrologic units based on land use type, hydrologic soil group, and slope classification (Table 9). Runoff coefficients were assigned to each of these hydrologic units based on a look-up table (Browne 1991; Table 10). Gridded long-term average annual precipitation data (OCS 2008; Table 9) were applied and were multiplied by hydrologic unit areas and runoff coefficients to generate annual runoff volumes.

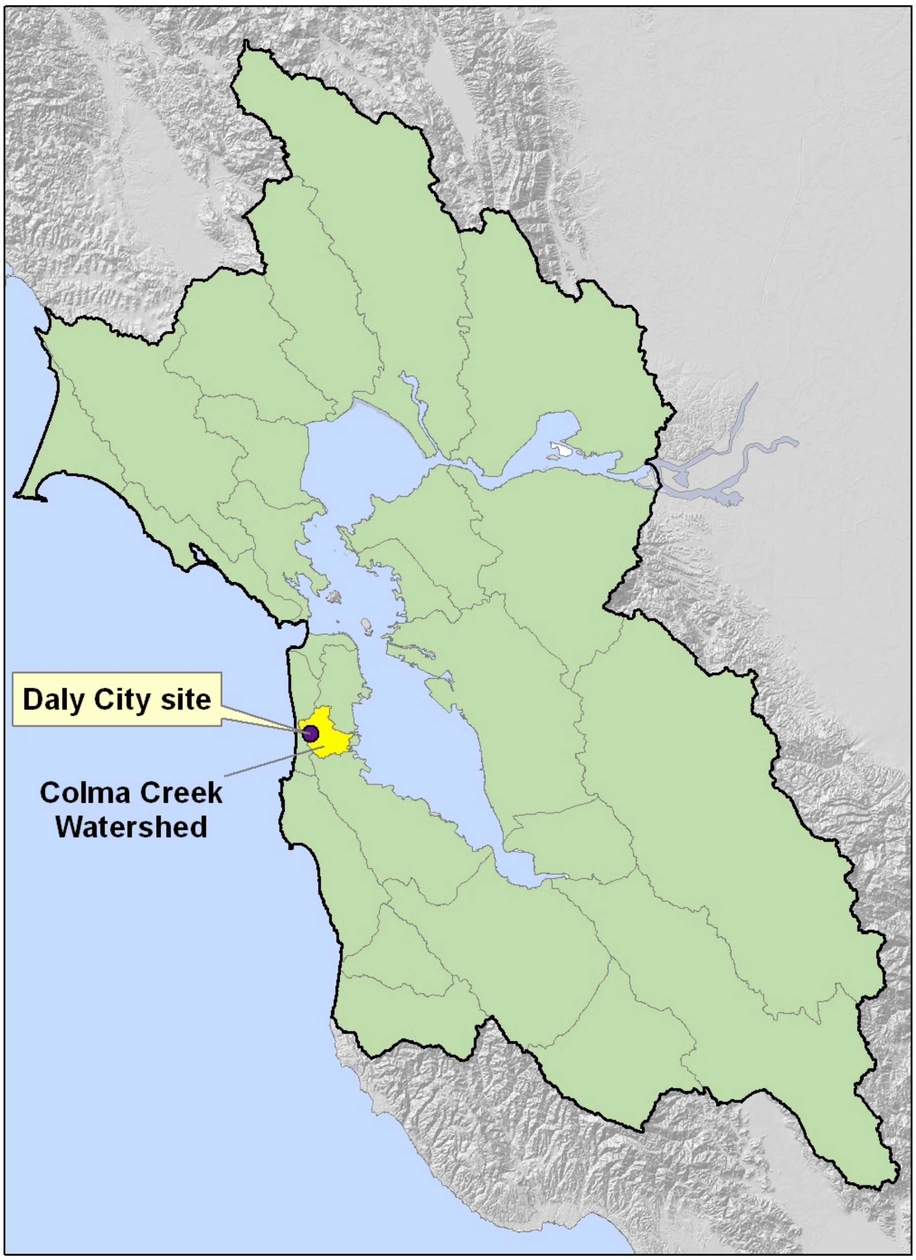


Figure 13. Spatial extent of model with the treatment site and Colma Creek watershed marked.

Table 9. Data used to generate base hydrologic model

| Data Type | Data Set | Reference |
|---|---|---|
| Land use | ABAG 2000 land use | ABAG 2000 |
| Soil | STATSGO | USDA 1993 |
| Slope | USGS National Elevation Dataset (NED) 10-m grid | Gesch et al. 2002 |
| Precipitation | PRISM 1971-2000 average precipitation 800-m grid | OCS 2008 |
| Runoff coefficients | Coefficients by land use, soil, and slope | Browne 1991 |

*Hydrologic Calibration and Performance*

Paired precipitation and flow gauge data were compiled for 18 watersheds within the model extent. For each watershed, regression relationships were established for annual total precipitation versus annual flow volume. These regression relationships were used to scale the observed flow volume data to the long-term average PRISM precipitation data. This scaling allowed comparison between the (adjusted) observed annual flow volume and the simulated annual flow volume, despite being generated from different years with different climate characteristics. Based on the model hydrologic performance, some runoff coefficients (RC) were modified to better represent local conditions (Table 10).

Table 10. Runoff coefficient ranges by land use

| Land Use | Initial RC range (from Browne 1991) | Final RC range (after calibration) |
|---|---|---|
| Open | 0.07-0.29 | 0.09-0.34 |
| Agriculture | 0.10-0.41 | 0.12-0.46 |
| Residential | 0.20-0.39 | 0.20-0.39 |
| Commercial | 0.71-0.72 | 0.50-0.60 |
| Industrial | 0.67-0.70 | 0.50-0.60 |
| Transportation | 0.78-0.83 | 0.78-0.83 |

Following hydrologic calibration, the simulated annual flow volumes for the 18 watersheds ranged between -45% and +56% of the scaled observed values, with a mean bias of -4% and a median bias of -7%. The model simulated an annual flow volume of 8.99 million $m^3$ for Colma Creek watershed while the scaled observed annual flow volume was 8.33 million $m^3$ (8% over-simulation).

*Land-use Specific Contaminant Concentrations*

Contaminant concentrations in stormwater runoff were compiled from the literature for the modeled land use categories (Table 11). Specifically, event mean concentrations (EMCs) were used. For sediment, total suspended sediment (TSS) EMCs were used because they were more prevalent in the literature than SSC EMCs and the bias introduced by using TSS as a proxy for SSC (Grey et al. 2000) is small relative to the uncertainties associated with the model (C. Sommers, personal communication). For some contaminants, land-use specific EMCs were not available, but land-use specific concentrations on sediment were available (Table 12). In these cases, land-use based runoff concentrations were estimated by multiplying concentrations on sediment by TSS EMCs for each land use category. Some of the contaminants, namely dioxins and motor oil, could not be included in this analysis since the post-LID implementation monitoring data set was dominated by non-detects, and so ratios of pre- to post-LID concentrations could not be calculated. In the case of diesel, insufficient land use specific concentration data were found for either water or sediment particles and so this contaminant could not be included in the analysis.

Table 11. Median event mean concentrations in stormwater runoff by land use category

| Land Use | TSS (mg/L) | COD (mg/L) | Cd (ug/L) | Cu (ug/L) | Ni (ug/L) | Pb (ug/L) | Zn (ug/L) |
|---|---|---|---|---|---|---|---|
| Open | 85 | 40 | 0.5 | 9 | 7 | 4 | 25 |
| Agriculture | 210 | 170 | 9.9 | 64 | 109 | 30 | 270 |
| Residential | 100 | 77 | 0.9 | 16 | 13 | 22 | 110 |
| Commercial | 67 | 58 | 1.3 | 29 | 25 | 33 | 250 |
| Industrial | 110 | 82 | 2 | 29 | 27 | 45 | 370 |
| Transportation | 95 | 80 | 1 | 21 | 20 | 30 | 120 |
| References | 1, 2, 4, 5, 7-9, 11-15, 17-19, 21, 22, 24-33 | 4, 8, 9, 11-13, 16, 17, 21, 22, 26, 30, 32 | 1, 3, 5, 6, 17, 20, 22, 24, 26, 31 | 1-6, 8-11, 13, 17, 20, 22-25, 27-32 | 1, 5, 6, 20, 22, 24, 31 | 1, 3-6, 8-13, 17, 20-25, 27-29, 31, 32 | 1-6, 9-11, 13, 17, 20-25, 27-31 |

References: 1 - Ackerman and Schiff 2003; 2 - ACWA 1997; 3 - Baeckstroem et al. 2003; 4 - Barrett et al. 1998; 5 - BASMAA 1996; 6 - BCDC 1991; 7 - Charbeneau and Barrett 1998; 8 - Choe et al. 2002; 9 - City of Austin 1995; 10 - Davis et al. 2001b; 11 - Driscoll et al. 1990; 12 - Ellis and Mitchell 2006; 13 - Gnecco et al. 2005; 14 - Goodson et al. 2006; 15 - Goonetilleke et al. 2005; 16 - Khan et al. 2006; 17 - Legret and Pagotto 1999; 18 - Line et al. 2002; 19 - Moore et al. 2002; 20 - this study; 21 - Pagotto et al. 2000; 22 - Pitt et al. 2004; 23 - Sansalone and Buchberger 1997; 24 - SCCWRP 2000; 25 - Shinya et al. 2000; 26 - Sorour 2000/2002; 27 - Stein et al. 2007; 28 - Stein et al. 2008; 29 - Tiefenthaler et al. 2008; 30 - USEPA 1983; 30 - WCC 1991; 31 - Wu et al. 1998; 32 - www.newriverwetlands.com

Table 12. Median concentration in local sediments by land use (estimated from KLI 2002 and ACCWP 2002)

| Land Use | t-Hg (ppm) | t-PAHs (ppm) | t-PCBs (ppb) |
|---|---|---|---|
| Open | 0.30 | 0.10 | 1.0 |
| Agriculture | 0.30 | 0.10 | 1.0 |
| Residential | 1.1 | 5.0 | 50 |
| Commercial | 1.1 | 5.0 | 50 |
| Industrial | 2.0 | 30 | 450 |
| Transportation | 2.0 | 30 | 50 |

*Applying Contaminant Concentrations*

The contaminant concentrations shown in Table 11 and generated from Table 12 were applied by land use type to the Colma Creek watershed and to the overall San Francisco Bay Area region to generate baseline contaminant loads. In the model, transportation runoff was calculated as being purely sourced from transportation land use, although, in reality, some transportation areas can have contributions from surrounding land uses, especially streets in developed areas. The land use distribution for the watershed and the region are shown in Table 13.

Table 13. Land use distributions

| Land Use | Colma Creek Watershed Area in km$^2$ (% of total) | San Francisco Bay Area region* Area in km$^2$ (% of total) |
|---|---|---|
| Open | 10.4 (25.7%) | 4587 (54.7%) |
| Agriculture | 0.3 (0.7%) | 1048 (12.5%) |
| Residential | 13.2 (32.5%) | 1620 (19.3%) |
| Commercial | 6.3 (15.5%) | 477 (5.7%) |
| Industrial | 1.6 (3.9%) | 214 (2.6%) |
| Transportation | 8.8 (21.7%) | 437 (5.2%) |
| Total | 40.6 | 8383 |

*with dammed areas greater than 20 mi$^2$ (52 km$^2$) removed

*LID Implementation Scenarios*

For the implementation scenarios, similar areas to the Daly City site, specifically transportation-related land, were divided into several categories and assessed for suitability for bioretention. The areas meeting the basic bioretention site suitability criteria were then included in the LID implementation scenarios. Additionally, the impacts of site criteria that impacted implementation costs were considered.

For testing different levels of implementation, transportation land use was divided into parking, streets, and all other uses. The parking lot category included parking lots, parking garages, and municipal vehicle yards. The streets category contained local streets and roads. The overall transportation category included all transportation-related land uses, including the first two categories, plus highways, airports, railways, and ports. Regionally, transportation land use contains 69% local streets, 16% highways, 10% airports, 2% railways, 1% ports, 1% parking lots, and 1% unspecified transportation. Colma Creek watershed transportation-related land consists of 89% local streets, 5% highways, 4% parking lots, and less than 1% of each for the other transportation categories.

Although much of transportation land would be suitable for some type of LID, only some of the transportation land would be suitable for bioretention retrofits (Table 14). Implementation of bioretention would be restricted by available space (that does not conflict with other uses such as utilities) and steepness of slope. Additionally, the site design would be impacted by site topography (e.g., drainage paths and pooling points), soil type and stability, depth to impermeable layer, and depth to water table (Table 14).

Table 14. LID site suitability criteria (LIDC 2010)

| LID type | Hydrologic Soil Group | | | | Depth to groundwater (ft) | | Depth to impermeable layer / bedrock (ft) | | Slope (% rise) | | | High landslide risk |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | A | B | C | D | <10 | >10 | <5 | >5 | <5 | 5-15 | >15 | |
| Bioretention | ✓ | ✓ | | | | ✓ | | ✓ | ✓ | ✓* | | |
| Bioretention with underdrain | | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓* | | ✓ |
| Permeable pavement | ✓ | ✓ | | | | ✓ | | ✓ | ✓ | | | |
| Permeable pavement with underdrain | | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | | | ✓ |
| Capture/reuse | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| Vegetated roofs | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ |
| Soil amendments | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | |
| Downspout disconnection | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | | |
| Filter strips | ✓ | ✓ | ✓ | ✓ | | ✓ | | ✓ | ✓ | | | |
| Vegetated swales | ✓ | ✓ | ✓ | ✓ | | ✓ | | ✓ | ✓ | ✓ | | |
| Infiltration (retention) basins | ✓ | ✓ | ✓ | | | ✓ | | ✓ | ✓ | | | |
| Infiltration trenches | ✓ | ✓ | ✓ | | | ✓ | | ✓ | ✓ | | | |
| Dry wells | ✓ | ✓ | ✓ | | | ✓ | | ✓ | ✓ | | | |
| Dry ponds (detention basins) | ✓ | ✓ | ✓ | | | ✓ | | ✓ | ✓ | | | |
| Constructed wetlands | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | | | |
| Wet ponds | | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | ✓ | | |

*if a terraced design is used

Available space is one of the most important site suitability criteria, but is the hardest to determine on a large scale. Bioretention is highly flexible in terms of site design geometry and scale, but is limited to areas that do not conflict with other uses (e.g., utilities) or to areas that can potentially change use (e.g., parking) as long as the system is situated downslope or subgrade relative to the treatment catchment. For parking lots, bioretention can be retrofitted into existing lot islands, medians and perimeters. For streets, linear and "cell" bioretention can be implemented in medians, sidewalk planters, and side swales. However, suitable locations are limited because the bioretention facility needs to be at a low point of the curbline profile for it to capture runoff from most of the street.

Additionally, implementation in medians is limited to roads that are not crowned and adding sidewalk planters generally entails loss of parking. For highways or railways, linear bioretention can be implemented in medians when not crowned or side swales when not conflicting with utilities. For any site type, multiple cells can be connected to scale up treatment area. Ideally, a combination of methods and data sets would be used to assess available space, such as interviews with local LID experts plus assessment of aerial photography, land ownership data, and utility maps. Since this level of effort was beyond the scope of this project, only discussions with local LID experts were used to estimate percentages of different land use areas with enough space to accommodate bioretention facilities (Table 15).

Table 15. Experience based estimates of technical feasibility of retrofitting sites for bioretention provided by local low impact development experts. The experts stated which region their estimates were based on. The estimates are shown as percent of area in each land use category.

| Land use category | Scott Durbin, Sustainable Watershed Designs San Francisco | Matt Fabry, San Mateo Countywide Pollution Prevention Program SF Bay Area Region | Dan Cloak, Dan Cloak Environmental Consulting California |
|---|---|---|---|
| Parking lots | 75% | 90% | 90% |
| Local streets | 50% | 75% | 25% |
| Other transport (non-urban) | - | 25% | 25% |
| Other transport (urban) | - | 10% | 10% |

The implementation feasibility estimates provided by the three local LID experts followed the same trend. They suggested that parking lots are the most amenable to bioretention retrofit, followed by local streets and other transport (mainly highways) in non-urban settings, and other transport (again mainly highways) in urban areas being the least amenable to bioretention retrofit. Parking lots are highly amenable since bioretention can be retrofitted into existing parking lot islands and perimeters. On the other end of the spectrum, urban highways are generally not conducive to bioretention retrofit since they often have no suitable green space and, even if they do have a green median, the road surface is generally crowned so that water will not run into it. The estimates provided by Scott Durbin (Sustainable Watershed Designs) were for San Francisco, which has unique challenges given its much higher density of development relative to the other cities in the region (including the City of Colma), the age of the city, and the use of a combined sewer system for wastewater and stormwater. As a result, the implementation feasibility estimates for the City of San Francisco, while informative, were deemed uncharacteristic of the likely conditions found in the wider Bay Area and were not treated any further in this study. The estimates provided by Matt Fabry (San Mateo Countywide Pollution Prevention Program) were averages for the San Francisco Bay Area region and the estimates provided by Dan Cloak (Dan Cloak Environmental Consulting) were averages for California. These estimates are identical except for local streets; Dan Cloak indicated that bioretention works well only where low points and availability of space coincide, which, in his experience, is rare for streets. He also noted that the tendency of streets to be crowned and resistance to loss of street lanes, parking, or sidewalk space would hinder bioretention retrofit feasibility in medians and sidewalk planters, respectively. Given there are other feasibility factors that could not be applied due to lack of GIS spatial data, we decided to use the most conservative applicable feasibility estimates: 90% of parking lots, 25% of local streets, and 10% of all other transport-related land were considered amenable to bioretention retrofit.

Aside from available space, the other critical factor for bioretention feasibility is slope. For bioretention facilities, the LID Center recommends avoiding sites with slopes greater than 15% (LIDC 2010). Accordingly, areas with average slopes greater than 15% were removed from the implementation scenario analysis. Excluding slopes greater than 15% also served to remove areas at risk of landslides (Nilsen et al. 1979), which is a consideration for designing bioretention facilities (LIDC 2010). Another slope-related consideration for bioretention is that terracing will be required for slopes greater than 5%. Since terracing a site dramatically increases construction costs (slope stabilization and other geotechnical costs, heavier equipment, and labor considerations), we made the assumption that in the near term early adoption period of bioretention systems, the realistic/practical implementation scenario would likely only include areas with slopes equal to or less than 5%. As shown in the results section, there is an abundance of theoretically feasible application sites that could be considered before moving upslope.

*LID Treatment Application*

The Daly City LID site monitoring results were applied using a reduction multiplier (1 - removal efficiency) on EMCs for transportation land use that met the ≤15% slope requirement for the theoretical set of scenarios and the ≤5% slope requirement for the practical set of scenarios. For each contaminant, the removal efficiency was calculated from the pre- and post-LID implementation EMCs (Table 16). To test the sensitivity of the model to the removal efficiencies, the impact of using a removal efficiency calculated by averaging all the different efficiencies shown in Table 8 was assessed.

To account for long-term average climatic conditions, the removal efficiencies were scaled to reflect the amount of treatment bypass on average at the decadal time scale. This scaling is necessary because, unless monitoring is performed over a long time frame (i.e., decades), the monitoring period is generally not representative of long-term average climatic conditions. To estimate the percent of flow that would bypass treatment, first the percent of rainfall that would exceed the site design storm rating needed to be calculated. For this purpose, a rainfall intensity cumulative distribution curve (Figure 14) was developed based on an hourly precipitation record from San Francisco Airport (WY 1980-2007) that was scaled to Colma Creek watershed's long-term average rainfall. The site design bypass threshold rate of 0.2 in/hr was applied to the rainfall intensity distribution to determine that the threshold rate was surpassed 7% of the time. A histogram of the hourly rainfall intensity was developed (Figure 14). Contributions of each binned range of rainfall intensities to the total precipitation were generated by multiplying the probability density by mid-point of the bins. After normalizing the rainfall contributions, the contribution of the rates below 0.2 in/hr was summed. Following these steps, it was estimated that, over a decadal time scale, 28% of the total amount of rainfall (and the corresponding runoff) would bypass treatment in the Daly City site. Table 16 shows both the ideal (no bypass) treatment efficiencies, which would apply in low rainfall intensity years, and the more realistic (some bypass) treatment efficiencies, which would apply in average rainfall intensity years.

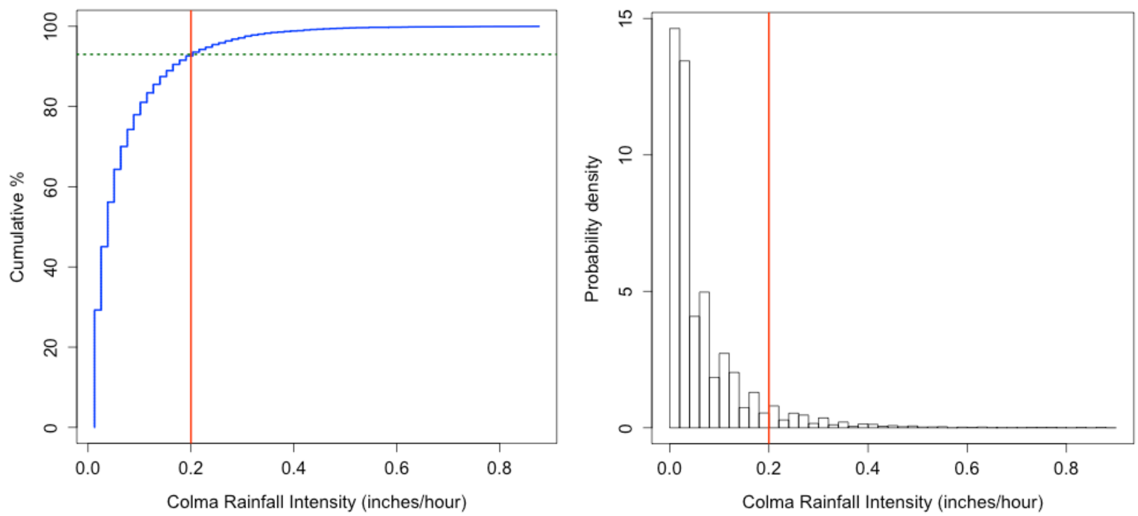

Figure 14. Cumulative distribution and histogram of hourly rainfall intensities for Colma Creek watershed.

Table 16. EMCs for runoff pre- and post-LID implementation, and resulting removal efficiencies for 100% and 72% of runoff treated.

| | Data from Daly City bioretention system | | Calculated treatment efficiencies | |
|---|---|---|---|---|
| Constituent | Pre-LID EMC | Post-LID EMC | No bypass | 28% bypass* |
| Flow volume | 100 | 90 | 0.1 | 0.07 |
| SSC (mg/L) | 24 | 13 | 0.46 | 0.33 |
| COD (mg/L) | 313 | 46 | 0.85 | 0.61 |
| Cd (ug/L) | 0.67 | 0.07 | 0.9 | 0.64 |
| Cu (ug/L) | 59 | 5.9 | 0.9 | 0.65 |
| Ni (ug/L) | 20 | 8.8 | 0.56 | 0.4 |
| Pb (ug/L) | 4.9 | 1.5 | 0.69 | 0.5 |
| Zn (ug/L) | 610 | 48 | 0.95 | 0.68 |
| Hg (ug/L) | 0.03 | 0.014 | 0.53 | 0.38 |
| PAHs (ug/L) | 2.65 | 0.188 | 0.93 | 0.67 |
| PCBs (ng/L) | 0.886 | 0.346 | 0.61 | 0.44 |

*based on the decadal scale climatic adjustment of the Daly City data

For all the implementation scenarios the long-term average removal efficiencies were used since the base hydrology model was created using long-term average precipitation and, moreover, the long-term average removal efficiencies represent more realistic treatment levels. Additionally, it was assumed the LID sites would be sized according to local rainfall and would result in similar treatment levels.

*Simplifications*

Because of the breadth of this analysis, in terms of spatial extent and number of contaminants included, numerous simplifications were required, which should be noted. The analysis could be considered limited in that it was based on a simple rainfall-runoff model combined with land use specific concentrations to generate loads; however, the modeling approach is reasonable for approximating loads on a long-term average annual scale for a regional spatial extent (Davis et al. 2000). While outside the scope of this project, ideally a sensitivity analysis would be performed to provide a range of load estimates; for example, the land use specific EMCs used were the median literature values, but the 0.1 and 0.9 percentile values could also be tested. Additionally, due to lack of local bioretention monitoring data, only the Daly City bioretension system data were used to estimate the change in EMCs after bioretention implementation. If more bioretention sites are monitored in the San Francisco Bay Area, the results could be used to estimate the range in removal efficiencies and their impact on load reductions.

Another simplification was that only direct drainage retrofits were considered, while storm drain retrofits were not included in this analysis. The potential to retrofit portions of the storm drain system so that storm drain pipes discharge into bioretention facilities may provide opportunities to apply LID to a greater amount of impervious area (D. Cloak, personal communication), suggesting that the results presented here may underestimate the achievable benefits.

## Results and Discussion

Baseline loads were calculated for Colma Creek watershed and for the San Francisco Bay Area region to provide a context for the loads reductions associated with broader implementation of LID sites similar to the Daly City site. The loads estimated using the simple rainfall/runoff model described above represent the total amount of a given contaminant that would be delivered from local watersheds to receiving waters (bay or ocean) for all modeled areas except San Francisco. Unlike the rest of the Bay Area region, San Francisco has a combined sewer system, which delivers stormwater along with wastewater to treatment plants. Accordingly LID implementation in San Francisco would result in loads being averted from wastewater treatment plants, instead of being averted from receiving waters (except in cases of overflow conditions).

The baseline loads were separated into those originating from transportation land use and those sourced from all other land uses. The changes in transportation land use loads were calculated for the different LID implementation scenarios. The scenarios represent increasing levels of LID implementation associated with treating parking lots only, treating streets only, and treating all transportation-related land uses. Each LID implementation scenario was considered under two conditions; the first was a theoretical implementation where all relevant land use with slopes equal to or less than 15% was treated without consideration of technical feasibility issues (e.g., space limitations) and the second was a more realistic implementation where relevant land use with slopes equal to or less than 5% was treated and the percent of land treated was scaled by the technical feasibility estimates provided by the local LID experts.

### *Baseline Loads*

The baseline loads for Colma Creek watershed are shown in Figure 15 with loads from transportation land use separated out from all other land uses. While less than a quarter of Colma Creek watershed is dedicated to transportation-related land uses, the loads from transportation land use contribute more than half the total load for all contaminants except Zn and PCBs. For Zn and PCBs, which both have a stronger industrial signal, transportation land use contributes about a third of the load. The disproportionate contribution of transportation land use to contaminant loads suggests that applying LID techniques to this land use can have significant effects on total loads generated in Colma Creek watershed.

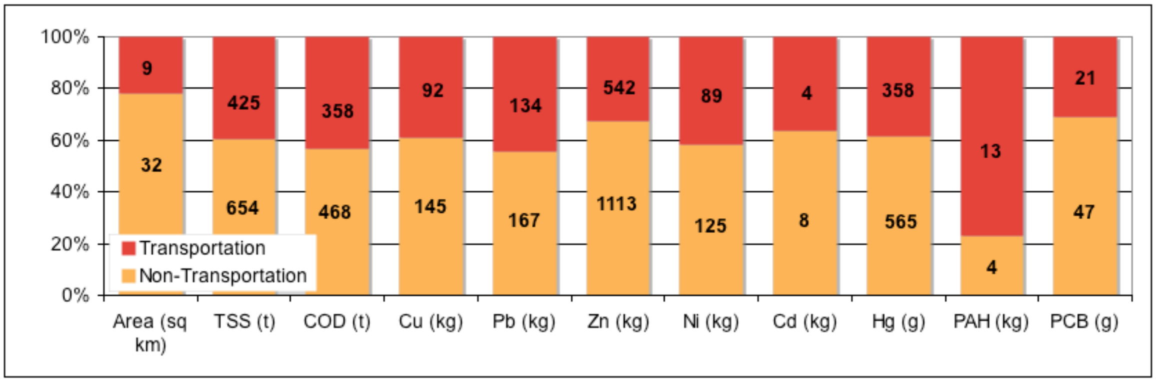

Figure 15. Areas and long-term average loads for Colma Creek watershed shown for transportation and non-transportation land use as percentages. The actual area and loads are denoted numerically on the graph, and their associated units are shown along the x-axis.

The baseline loads for the San Francisco Bay Area region are shown in Figure 16 with loads from transportation land use separated out from all other land uses. The percent of transportation land use is much lower for the entire region than for Colma Creek watershed, so the percent of loads from transportation land use are also much lower. However, the transportation land use still contributes disproportionately; it is only 5% of the entire area, but contributes about 15% of total loads of mercury and PCBs. Additionally, transportation land use contributes 20% of the Pb load and 60% of the PAH load.

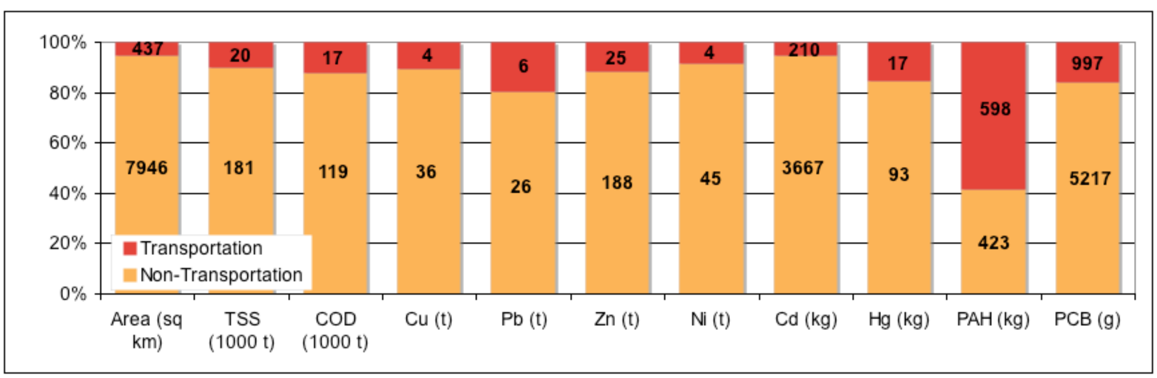

Figure 16. Areas and long-term average loads for the San Francisco Bay Area region shown for transportation and non-transportation land use as percentages. The actual area and loads are denoted numerically on the graph, and their associated units are shown along the x-axis.

*Theoretical (Upper-Bound) Post-Treatment Loads*
For the theoretical scenarios, bioretention was applied to the relevant transportation land uses in areas meeting the slope criteria. Table 17 shows the amount of area meeting the ≤15% slope criteria for installing bioretention for both Colma Creek watershed and the entire region. Additionally, Table 17 shows the amount of area meeting the ≤5% slope criteria, which are used later in the practical/realistic implementation scenarios to avoid more expensive terraced site designs.

Table 17. Transportation land use and slope-based bioretention site suitability

| Land Use | Colma Creek Watershed | | | San Francisco Bay Area region | | |
|---|---|---|---|---|---|---|
| | Area ($km^2$) | Area with ≤5% slope ($km^2$) | Area with ≤15% slope ($km^2$) | Area ($km^2$) | Area with ≤5% slope ($km^2$) | Area with ≤15% slope ($km^2$) |
| Transportation | 8.8 | 3.8 | 6.4 | 440 | 270 | 360 |
| Parking lots | 0.4 | 0.3 | 0.4 | 6.2 | 4.4 | 5.4 |
| Streets | 7.8 | 3 | 5.4 | 300 | 170 | 240 |

The results of the theoretical LID implementation scenarios are shown in Tables 18 and 19. These scenarios represent bioretention retrofit of all relevant land use with slopes equal to or less than 15%. The scenarios do not take into account technical feasibility issues such as space limitations or conflicts with utilities. While these scenarios are unrealistic, they provide an upper bound on potential benefits of applying bioretention to transportation land use for the areas under consideration. Additionally, these scenario results allow others to apply their own technical feasibility estimates and generate their own post-treatment load estimates.

Table 18 shows the simulated contaminant loads from transportation land use in Colma Creek watershed for baseline conditions (no runoff treatment) and for the three levels of LID implementation scenarios under theoretical conditions. These scenarios show the outcomes of treating 0.4 $km^2$ of parking lots, 5.4 $km^2$ of local streets and 6.4 $km^2$ of transportation land relative to the untreated (baseline) load generated by 8.8 $km^2$ of transportation land.

In this theoretical analysis of retrofitting transportation land in Colma Creek watershed with bioretention, treating 0.4 $km^2$ of parking lots (4.5% of the total transportation land use) would result in 2 to 3% reduction in transportation-related loads. Retrofitting the local streets (5.4 $km^2$ or 61% of total transportation land use) would result in 23 to 43% reduction in transportation-related loads. Treating all the transportation land with ≤15% slope (6.4 $km^2$ or 73% of total transportation land use) would result in 27 to 51% reduction in transportation-related loads. It is, of course, not realistic to assume all transportation land would be amenable to retrofit with bioretention, but these results provide a great contextual framework and allow others to apply their own feasibility scaling factors.

Table 18. Average annual loads from Colma Creek watershed transportation land for baseline scenario and for theoretical LID implementation scenarios (≤15% slope and assumption of technical feasibility). Percent reductions from baseline are also shown.

| Contaminant | Baseline (untreated) | Scenario 1: Parking lots treated | Scenario 2: Streets treated | Scenario 3: All transportation land treated |
|---|---|---|---|---|
| | Loads from transportation land use | | | |
| | Load in kg | Load in kg (% Reduction) | Load in kg (% Reduction) | Load in kg (% Reduction) |
| TSS | $425*10^3$ | $418*10^3$ (2%) | $327*10^3$ (23%) | $310*10^3$ (27%) |
| COD | $358*10^3$ | $348*10^3$ (3%) | $218*10^3$ (39%) | $192*10^3$ (46%) |
| Cd | 4.5 | 4.3 (3%) | 2.7 (41%) | 2.3 (48%) |
| Cu | 92 | 89 (3%) | 54 (41%) | 47 (49%) |
| Hg | 0.36 | 0.35 (2%) | 0.27 (26%) | 0.25 (30%) |
| Ni | 89 | 88 (2%) | 65 (27%) | 61 (32%) |
| Pb | 134 | 131 (2%) | 90 (33%) | 82 (39%) |
| Zn | 542 | 525 (3%) | 310 (43%) | 267 (51%) |
| PAHs | 12.8 | 12.4 (3%) | 7.4 (42%) | 6.4 (50%) |
| PCBs | $21*10^{-3}$ | $21*10^{-3}$ (2%) | $15*10^{-3}$ (29%) | $14*10^{-3}$ (35%) |

Table 19 shows the simulated contaminant loads from transportation land use in San Francisco Bay Area region for baseline conditions (no runoff treatment) and for the three levels of LID implementation scenarios under theoretical conditions. These scenarios show the outcomes of treating 5.4 $km^2$ of parking lots, 237.9 $km^2$ of local streets and 355.2 $km^2$ of transportation land relative to the untreated (baseline) load generated by 436.9 $km^2$ of transportation land.

Table 19. Average annual loads from the San Francisco Bay Area region transportation land for baseline scenario and for theoretical LID implementation scenarios (≤15% slope and assumption of technical feasibility). Percent reductions from baseline are also shown.

| Contaminant | Baseline (untreated) | Scenario 1: Parking lots treated | Scenario 2: Streets treated | Scenario 3: All transportation land treated |
|---|---|---|---|---|
| | Loads from transportation land use | | | |
| | Load in kg | Load in kg (% Reduction) | Load in kg (% Reduction) | Load in kg (% Reduction) |
| TSS | $19.9*10^6$ | $19.8*10^6$ (0.5%) | $15.7*10^6$ (21%) | $14.0*10^6$ (30%) |
| COD | $16.8*10^6$ | $16.7*10^6$ (0.8%) | $10.8*10^6$ (36%) | $8.3*10^6$ (51%) |
| Cd | 210 | 208 (0.8%) | 132 (37%) | 99 (53%) |
| Cu | $4.3*10^3$ | $4.3*10^3$ (0.8%) | $2.7*10^3$ (38%) | $2.0*10^3$ (53%) |
| Hg | 16.8 | 16.7 (0.5%) | 12.8 (24%) | 11.2 (33%) |
| Ni | $4.2*10^3$ | $4.2*10^3$ (0.5%) | $3.2*10^3$ (25%) | $2.7*10^3$ (35%) |
| Pb | $6.3*10^3$ | $6.3*10^3$ (0.6%) | $4.4*10^3$ (30%) | $3.6*10^3$ (42%) |
| Zn | $25*10^3$ | $25*10^3$ (0.9%) | $15*10^3$ (39%) | $11*10^3$ (56%) |
| PAHs | 598 | 593 (0.8%) | 366 (39%) | 269 (55%) |
| PCBs | 0.996 | 0.991 (0.6%) | 0.729 (27%) | 0.618 (38%) |

In this theoretical analysis of retrofitting transportation land in the San Francisco Bay Area region with bioretention, treating the 5.4 $km^2$ of parking lots (1.2% of the total transportation land use) would result in 0.5 to 0.9% reduction in transportation-related loads. Retrofitting local streets (238 $km^2$ or 54% of total transportation land use) would result in 21 to 39% reduction in transportation-related loads. Finally, treating all the transportation land with ≤15% slope (355 $km^2$ or 81% of total transportation land use) would result in 30 to 56% reduction in transportation-related loads. The differences in the percent load reductions for the region as a whole versus for Colma Creek watershed reflect the differences in the distribution of different transportation land uses, namely percent of total transportation area and amount of precipitation received.

*Practical (Realistic) Post-Treatment Loads*

The results of the practical or realistic LID implementation scenarios are shown in Tables 20 and 21. These scenarios represent bioretention retrofit of the relevant land uses for areas that would not require terracing (i.e., slopes equal to or less than 5%). Table 17 shows that if bioretention implementation sites were limited to areas with 5% slope or less, the potential for application would be reduced to 25% for parking lots, 44% for streets, and 41% for transportation land use overall in Colma Creek watershed and would result in potential site area reductions of 19% for parking lots, 30% for streets, and 24% for transportation land use for the entire region. Additionally, for these practical scenarios, the areas of LID application were scaled by the conservative technical feasibility estimates: 90% of parking lots, 25% of local streets, and 10% of all other transport-related land.

Table 20 shows the simulated contaminant loads from transportation land use in Colma Creek watershed for baseline conditions (no runoff treatment) and for the three levels of LID implementation scenarios under the more realistic conditions. These scenarios show the outcomes of treating 0.27 $km^2$ of parking lots, 0.75 $km^2$ of local streets and 1.1 $km^2$ of transportation land relative to the untreated (baseline) load generated by 8.8 $km^2$ of transportation land.

In this practical or more realistic analysis of retrofitting transportation land in Colma Creek watershed with bioretention, treating 90% of parking lots situated on land with ≤5% slope (0.27 $km^2$ or 3% of the total transportation land use) would result in 1 to 2% reduction in transportation-related loads. Likewise, treating 25% of streets (0.75 $km^2$ or 9% of the total transportation land use) translates into a 3 to 6% transportation-related loads reduction. Combining the results from the feasibility-scaled treatment of parking lots and streets with loads from all other transportation land uses incorporating a 10% treatment feasibility factor results in a 5 to 9% load reduction for treating 1.07 $km^2$ (12%) of transportation land use.

Table 20. Average annual loads from Colma Creek watershed transportation land for the baseline scenario and for practical LID implementation scenarios (≤5% slope and scaled for technical feasibility). Percent reductions from baseline are also shown.

| Constituent | Loads from transportation land use | | | |
|---|---|---|---|---|
| | Baseline (untreated) | Scenario 1: Parking lots treated | Scenario 2: Streets treated | Scenario 3: All transportation land treated |
| | Load in kg | Load in kg (% Reduction) | Load in kg (% Reduction) | Load in kg (% Reduction) |
| TSS | $425*10^3$ | $419*10^3$ (1%) | $411*10^3$ (3%) | $405*10^3$ (5%) |
| COD | $358*10^3$ | $350*10^3$ (2%) | $338*10^3$ (5%) | $329*10^3$ (8%) |
| Cd | 4.5 | 4.4 (2%) | 4.2 (6%) | 4.1 (8%) |
| Cu | 92 | 90 (2%) | 86 (6%) | 84 (8%) |
| Hg | 0.36 | 0.35 (2%) | 0.34 (4%) | 0.34 (4%) |
| Ni | 89 | 88 (2%) | 86 (4%) | 84 (6%) |
| Pb | 134 | 132 (2%) | 128 (5%) | 125 (7%) |
| Zn | 542 | 529 (2%) | 509 (6%) | 494 (9%) |
| PAHs | 12.8 | 12.4 (2%) | 12.0 (6%) | 11.6 (9%) |
| PCBs | $21*10^{-3}$ | $21*10^{-3}$ (2%) | $20*10^{-3}$ (4%) | $20*10^{-3}$ (6%) |

Relative to the theoretical conditions, reducing treated parking lots by a third (0.4 $km^2$ to 0.27 $km^2$) to a practical level resulted in a 1-2% range in reduction, as opposed to the 2-3% load reduction under the upper-bound conditions. Reducing treated streets by 86% (5.4 $km^2$ to 0.75 $km^2$) to reflect feasibility resulted in a 3-6% range in reduction instead of a 23-43% load reduction under the upper-bound conditions. Introducing all the feasibility factors modified the treated transportation area from 6.4 $km^2$ to 1.07 $km^2$ (83% site area loss) and resulted in a 5 to 9% load reduction instead of a 27-51% load reduction under the upper-bound conditions.

The sediment loads exhibited the least sensitivity to changes in LID implementation, which is consistent with sediment concentrations being subject to lowest removal efficiency of the contaminants in monitoring data set. At the other end, Cd, Cu, Zn, and PAHs exhibited most sensitivity to changes in LID implementation, which reflected the high removal efficiencies for these contaminants in the bioretention system.

Table 21 shows the simulated contaminant loads from transportation land use in the San Francisco Bay Area region for baseline conditions (no runoff treatment) and for the three levels of LID implementation scenarios under more realistic conditions. These scenarios show the outcomes of treating 4.0 $km^2$ of parking lots, 42 $km^2$ of local streets and 56 $km^2$ of transportation land relative to the untreated (baseline) load generated by 440 $km^2$ of transportation land.

In this practical/realistic analysis of retrofitting transportation land in the San Francisco Bay Area region with bioretention, treating 90% of parking lots situated on land with ≤5% slope (4.0 $km^2$ or 0.9% of the total transportation land use) would result in 0.3 to 0.6% reduction in transportation-related loads. Treating 25% of non-steep streets (42 $km^2$ or 10% of the total transportation land use) translates into a 4 to 7% transportation-related loads reduction. Combining the results from the feasibility-scaled treatment of parking lots and streets with loads from all other transportation land uses incorporating a 10% treatment feasibility factor results in a 5 to 9% load reduction for treating 56 km$^2$ (13%) of transportation land use.

Relative to the theoretical conditions, reducing treated parking lots by a quarter (5.4 km$^2$ to 4.0 km$^2$) to a practical level resulted in a 0.3-0.6% range in load reduction, as opposed to the 0.5-0.9% load reduction under the upper-bound conditions. Reducing treated streets by 82% (240 km$^2$ to 42 km$^2$) to reflect feasibility resulted in a 4-7% range in reduction instead of a 21-39% load reduction under the upper-bound conditions. Introducing all the feasibility factors modified the treated transportation area from 360 km$^2$ to 56 km$^2$ (84% site area loss) and resulted in a 5 to 9% load reduction instead of a 30-56% load reduction under the upper-bound conditions.

Table 21. Average annual loads from the San Francisco Bay Area region transportation land for the baseline scenario and for practical LID implementation scenarios (≤5% slope and scaled for technical feasibility). Percent reductions from baseline are also shown.

| Constituent | Loads from transportation land use | | | |
|---|---|---|---|---|
| | Baseline (untreated) | Scenario 1: Parking lots treated | Scenario 2: Streets treated | Scenario 3: All transportation land treated |
| | Load in kg | Load in kg (% Reduction) | Load in kg (% Reduction) | Load in kg (% Reduction) |
| TSS | $19.9*10^6$ | $19.8*10^6$ (0.3%) | $19.2*10^6$ (4%) | $19.0*10^6$ (5%) |
| COD | $16.8*10^6$ | $16.7*10^6$ (0.5%) | $15.7*10^6$ (6%) | $15.4*10^6$ (8%) |
| Cd | 210 | 208 (0.6%) | 196 (7%) | 192 (8%) |
| Cu | $4.3*10^3$ | $4.3*10^3$ (0.6%) | $4.0*10^3$ (7%) | $3.9*10^3$ (8%) |
| Hg | 16.8 | 16.7 (0.3%) | 16.1 (4%) | 15.9 (5%) |
| Ni | $4.2*10^3$ | $4.2*10^3$ (0.4%) | $4.0*10^3$ (4%) | $4.0*10^3$ (6%) |
| Pb | $6.3*10^3$ | $6.3*10^3$ (0.4%) | $6.0*10^3$ (5%) | $5.9*10^3$ (7%) |
| Zn | $25*10^3$ | $25*10^3$ (0.6%) | $24*10^3$ (7%) | $23*10^3$ (9%) |
| PAHs | 598 | 593 (0.6%) | 557 (7%) | 545 (9%) |
| PCBs | 0.996 | 0.991 (0.4%) | 0.949 (5%) | 0.936 (6%) |

The sensitivity of loads to LID implementation followed the same trends as seen in the Colma Creek watershed results when comparing theoretical and practical implementation scenarios. As discussed earlier, the implementation sensitivity reflected the removal efficiencies; the contaminants experiencing low removal rates were relatively insensitive to LID implementation and the contaminants experiencing higher removal rates were more sensitive to LID implementation.

*Sensitivity of Results to Type of Removal Efficiency*

The removal efficiencies used in this analysis were based on the pre- and post-LID implementation EMCs collected at Gellert Park since this is an EMC-based model. However, one could alternatively use any of the computed removal efficiencies shown in Table 8 (e.g., percent load reduction). To test the impact of this choice, the model was run with the average removal efficiency computed from the five ‘lines-of-evidence’ (average concentration, particle concentration, load, EMC, and FWMC) for copper. Copper was chosen because it tends to be strongly associated with transportation land use due to local sources such as brake pads, and so the authors have a high degree of confidence in the results. The average removal efficiency for Cu was 0.92 versus the EMC-based Cu removal efficiency of 0.90. When scaled to long-term average treatment bypass conditions, the average removal efficiency was reduced to 0.66 relative to 0.65 for the EMC-based Cu removal efficiency. Table 22 shows the impact of the alternative removal efficiency on Colma Creek watershed results. A 2% difference in removal efficiency resulted in a similar change in the loads, which is the response one would expect from a model based on a linear approximations.

Table 22. Impact of removal efficiency choice on estimated load reductions. Example uses Colma Creek watershed with theoretical LID implementation scenarios.

| | Loads from transportation land use | | | |
|---|---|---|---|---|
| Removal efficiency used | Baseline (untreated) | Scenario 1: Parking lots treated | Scenario 2: Streets treated | Scenario 3: All transportation land treated |
| | Cu load in kg | Cu load in kg (% Reduction) | Cu load in kg (% Reduction) | Cu load in kg (% Reduction) |
| 0.65 | 92 | 89 (3%) | 54 (41%) | 47 (49%) |
| 0.66 | 92 | 89 (3%) | 53 (42%) | 46 (50%) |

*Load Reductions Relative to Watershed and Regional Scale*

While this analysis focused on changes in loads specifically from transportation land use to normalize the results to the amount of transportation land use, it is useful to frame the results in terms of the total loads from the watershed and the region to see the big picture. The load reduction for the Colma Creek watershed for the practical scenario with highest level of LID implementation drops from 5-9% total loads averted (relative to transportation loads) to 2-3% total loads averted (relative to overall loads). The exception is the PAH load, which maintains a similar level of reduction regardless of being normalized to loads off transportation land use or to loads off all land uses, since it is strongly associated with transportation land use. For the entire region, the overall load reductions range from 0.5-1.3%, except for the load reduction for PAH, which is 5%. The load reductions for the region are relatively small since transportation land use makes up only 5% of the total land use. However, transportation land use still remains a high-leverage source, especially for PAHs (Table 23). Table 23 shows the ratio of PAHs load contribution (%) to area contribution (%) for each land use. Industrial and transportation land use categories are notable for having a ratio greater than one, i.e., disproportionately contributing contaminants relative to the their area.

Table 23. Example of transportation land as a high-leverage source.

| Land Use | Colma Creek Watershed | | | San Francisco Bay Area region* | | |
|---|---|---|---|---|---|---|
| | Contributing Area (%) | PAHs load (%) | Ratio of load to area | Contributing Area (%) | PAHs load (%) | Ratio of load to area |
| Open | 25.7% | 0.1% | < 0.01 | 54.7% | 0.9% | 0.016 |
| Agriculture | 0.7% | 0.0% | < 0.01 | 12.5% | 0.5% | 0.040 |
| Residential | 32.5% | 8.5% | 0.26 | 19.3% | 15.6% | 0.81 |
| Commercial | 15.5% | 4.7% | 0.30 | 5.7% | 4.7% | 0.82 |
| Industrial | 3.9% | 9.9% | 2.5 | 2.6% | 19.6% | 7.5 |
| Transportation | 21.7% | 76.8% | 3.5 | 5.2% | 58.6% | 11 |

*with catchment areas greater than 20 $mi^2$ (52 $km^2$) upstream from dams and reservoirs removed

*Load Reductions Relative to other Management Actions*

The modeling results presented here are quite encouraging, demonstrating that just one type of LID system, if applied more widely, could provide measurable loads reductions, however, it should not be forgotten that stormwater managers already apply considerable effort annually using conventional structural and non-structural best management practices (SFEI 2010). These practices include recycling (particularly for Hg but also for other metals), street sweeping, conveyance system maintenance, redevelopment retrofit (contaminated soils cleanup and complying with the stormwater permit C.3 provisions for hydro-modification reduction), building demolition controls (PCBs), and illicit waste cleanup. Although these efforts don't account for much load reduction individually, together they do cumulatively add up (SFEI 2010).

Using a spreadsheet to organize the thought experiment, the load reductions theoretically possible under reasonable scenarios of increased BMP effort was tested for the Bay Area for Hg and PCBs (Mangarella et al. 2010). The outcomes of the Mangarella analysis were similar to that of scaling up the results for the LID system described in the present report. For example, increased of Hg recycling effort was estimated to result in a 7.8% load reduction, increased street sweeping effort could result in a further 1.1% capture for Hg and a 6.5% capture for PCBs, street washing in selected contaminated areas would result in <1% increased capture, refocusing conveyance maintenance on contaminated sites would actually result in increased loads (negative efficiency), and smart redevelopment of industrial areas using treatment controls that employ settling (e.g. detention basins) would result in <1% increased capture for Hg and 2% increased capture for PCBs. All these percentages were estimated in relation to the currently estimated annual loads (Hg: 160kg; PCBs: 20 kg) (Mangarella et al. 2010). Thus, it appears that the application of green retrofits to transportation infrastructure is as good and in many cases better than the reasonable scenarios of increased effort using conventional BMPs, an encouraging result if we consider Hg and PCBs, several of our highest ranked pollutants of concern.

## Conclusions

*Modeling Conclusions*

The modeling study showed that just applying bioretention, one tool in the LID palette and one tool amongst other more conventional municipal BMP options, to just one land use, albeit a high-leverage contaminant source area, could make an impact on San

Francisco Bay load reduction objectives. Without cost and space considerations, bioretention applied to all transportation land use could result in approximately 30-60% loads reduction for the various contaminants considered. More realistically, if cost and technical feasibility restrictions were included, the potential benefit would be 5-9% reduction in loads sourced from transportation land, at least as good and, in most cases, better than could be achieved through increased effort using conventional urban BMP scenarios.

Although this analysis has focused on one LID technique applied to one land use, the analytical framework provided here could be applied to a wider range of LID techniques and land uses. One could add in other LID tools, as supporting data is available, and expand the potential treatable areas; for example, permeable pavement could be applied when space is limited and filter basins could be applied on steep slopes. Finally, this framework could be used to test which combinations of LID techniques would be most advantageous for local or regional scale application under Bay Area specific climatic and landuse conditions. What is lacking presently is a thorough analysis comparing the costs of application and maintenance versus efficiency of LID with a similar analysis of conventional structural and non-structural municipal BMPs.

*Overall Conclusions*

There are several benefits from the widespread application of bioretention systems. With systems like this in place the reduction of runoff peak flows and volumes through biofiltration and evapotranspiration, as described in the first section of this report, will not just aid in treatment of contaminants but also make high magnitude storms more manageable downstream. In systems with little or no hydromodification costly flood control expansions can possibly be avoided especially in times of climate change and potentially wetter winters in the near future. Additionally, bioretention systems with mostly native plants, educational signs, and benches for public use like at the Daly City library will not only provide a platform for outreach and education to the public but also add to the aesthetic appeal of the area.

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