Potential Benefits of Nutrient and Sediment Practices to Reduce Toxic Contaminants in the Chesapeake Bay Watershed

Part 1: Removal of Urban Toxic Contaminants

FINAL REPORT

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The following is a list of common acronyms used throughout the text:

ADT	Average Daily Traffic
BMP(s)	Best Management Practice(s)
CBP or CBPO	Chesapeake Bay Program Office
CBWM	Chesapeake Bay Watershed Model
EMC	Event Mean Concentration
Hg	Mercury
HMW	High Molecular Weight
HPCP	Household and Personal Care Products
HUC	Hydrologic Unit Code
LID	Low Impact Development
LMW	Low Molecular Weight
NPDES	National Pollutant Discharge Elimination System
OCP	Organochlorine Pesticides
OPP	Organophosphate Pesticides
OTM	Other Trace Metal
PAH	Polycyclic Aromatic Hydrocarbons
PBDE	Polybrominated Diphenyl Ether
PCB's	Polychlorinated Biphenyls
PEC	Probable Effects Concentration
MS4	Municipal Separate Storm Sewer System
Rv	Runoff Coefficient
TEC	Threshold Effects Concentration
TMDL	Total Maximum Daily Load
TOC	Total Organic Carbon
TPH	Total Petroleum Hydrocarbons
TSS	Total Suspended Solids
USGS	United States Geological Survey
UTC	Urban Toxic Contaminant
UTM	Urban Trace Metal

Foreword

This project was developed by the Toxic Contaminant Work Group to evaluate whether best management practices (BMPs) used to reduce nutrient and sediment loads for the Bay pollution diet might also offer additional benefits to reduce toxic contaminants. The results of this one year research synthesis are summarized in two technical memos. This memo is the first installment in the series, and looks at how stormwater BMPs remove urban toxic contaminants.

The second memo examines how toxic contaminants are influenced by the agricultural and wastewater sectors in the Chesapeake Bay, with an emphasis on croplands, animal feeding operations and manure application, as well as discharges from wastewater treatment plants and land application of municipal biosolids. The second memo focuses on the following toxic contaminants:

- Pesticide applications (especially herbicides used for no-till)
- Biogenic hormones produced by both sectors
- Human and veterinary pharmaceuticals and antibiotics generated from both sectors.

In compiling both memos, we tried to keep the technical jargon and organic chemistry to a minimum in order to make the findings more accessible to the general reader. Given the topics being explored, however, it is hard to avoid the complex and confusing terminology used to describe both toxic contaminants and stormwater best management practices.

Acknowledgements

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Executive Summary

A group of 12 toxins were classified as urban toxic contaminants (UTC), based on six unique criteria:

1. The toxin is primarily associated with urban land use, compared to other sectors in the watershed.
2. The toxin is either generated within the urban sector or is deposited from the atmosphere onto impervious surfaces and subsequently washes off.
3. Urban stormwater runoff is the predominant pathway for transporting the toxin in the watershed.
4. The toxin has "sediment-like characteristics" and can be removed by settling or filtering practices.
5. The toxin is generated or produced in an upland landscape position in the watershed where it can be effectively treated by an urban BMP that captures surface runoff.
6. Physical evidence exists that the toxin is captured and/or retained within an urban stormwater BMP.

Table E-1 Degree to Which the Toxin Categories Meet the Six Criteria for Urban Toxic Contaminants

Toxin Category	1. urban land use?	2. urban sources ?	3. stormwater pathway ?	4. Sediment characteristics	5. Upland Position ?	6. Urban BMP Capture or Retention?
PCBs	Y	Y	Y	Y	Y	y
PAH	Y	Y	Y	Y	Y	Y
TPH	Y	Y	Y	Y	y	Y
Hg	Y	Y	Y	Y	Y	y
UTM	Y	Y	Y	Y	Y	Y
OTM	Y	Y	Y	Y	y	y
PP	Y[a]	Y	Y	Y	y	y
OCP	Y[a]	Y	Y	Y	y	y
OPP	Y[a]	Y	Y	Y	y	ND
Plasticizer	y[w]	y	y	Y	y	y
PBDE	y[w]	y	y	Y	y	y
Dioxins	Y	y	y	Y	ND	ND

UTM: Urban Trace Metals (Cd, Cu. Pb and Zn)
OTM: Other Trace Metals (As, Cr, Fe and Ni)
PP: Pyrethroid Pesticides, OCP: Organochlorine pesticides, OPP organophosphate pesticides.
PBDE: Polybrominated diphenyl ethers

Y = Yes, based on strong evidence
y = Yes, supported by limited monitoring data
ND = no data available to assess
a: moderate loads are also produced by the ag sector
w: wastewater can also produce moderate loads

Table E-1 shows the strength of evidence for classifying the twelve groups of toxins as urban toxic contaminants or UTCs. The UTC designation is important as it enables watershed managers to target urban watersheds with effective stormwater BMPs to reduce toxin loads to the estuary.

Overall Findings For All Urban Toxic Contaminants

Despite differences in their origin and chemical characteristics, the 12 UTCs shared some common findings when it came to removal by urban stormwater BMPs.

- Most urban BMPs in the Bay watershed have a high capability to remove suspended sediment from urban runoff. Suspended sediments and UTCs share many common characteristics- they are hydrophobic, non-soluble, have a strong affinity for organic matter, and bind, adsorb or otherwise become attached to sediment particles. In addition, both sediments and UTCs are relatively inert, persistent and have low rates of biodegradation. Both are also associated with fine and medium grained particles that can be entrained in urban stormwater runoff. Most importantly, both are subject to the same BMP removal mechanisms (i.e., settling and filtering) and frequently achieve reductions on the order of 50 to 90% in most urban BMPs.

Table E-2: Comparison of BMP Treatability for the 12 Urban Toxic Contaminant Groups

Toxin Category	BMP Removal Rate?	Measured or Estimated?	Behaves like Sediment?	BMP Retention?	Sediment Toxicity Concern?
PCBs	TSS	E	Y	Y	PR
PAH	> TSS	E	Y	Y	CR
TPH	> TSS	M	Y	Y	MR
Hg	> TSS	E	Y	Y	PR
UTM	< TSS	M	Y	Y	PR
OTM	< TSS	M	Y	Y	PR
PP	TSS	E	Y	y	CR
OCP	> TSS	E	Y	y	MR
OPP	< TSS	E	Y	ND	MR
Plasticizers	< TSS	E	Y	y	ND
PBDE	< TSS	E	Y	Y	ND
Dioxins	< TSS	E	Y	ND	ND

Removal Rate:

>TSS: Higher than TSS Removal
TSS: Similar to TSS Removal
< TSS: Less than TSS Removal

M= Measured
E= Estimated

Y = Yes, based on strong evidence
Y = Yes, limited monitoring data provides support
ND = no data available to assess

PR: Potential Risk
CR: Clear Risk
MR: Minimal Risk

- Given the close relationship between suspended sediment and UTCs, it is possible to link UTCs to a benchmark sediment removal rate. This is helpful because it allows users to infer UTC removal rates based on known TSS removal rates that have been developed by more than a dozen expert panels and approved for use by the CBP partnership. Table E-2 compares the relative treatability of the 12 urban toxic contaminants.

- It is clearly evident that existing urban BMPs are effectively removing UTCs in urban runoff and are preventing many of them from reaching the rivers and estuaries of the Chesapeake Bay. Currently, about a third of urban areas are treated by urban BMPs in the Chesapeake Bay watershed, and future BMP coverage should steadily increase as states and localities implement the Bay TMDL.

- While a precise estimate is not possible, rough calculations indicate that urban BMPs are reducing UTCs by about 25% now and perhaps by as much as 40% by 2025. This finding suggests that efforts to reduce nutrients and sediments for the Bay TMDL can produce other significant water quality benefits, such as reducing toxicity to fish, wildlife and humans. Continued implementation of BMPs in the urban sector for the Bay TMDL is a key element of a comprehensive strategy to reduce loads of UTCs across the Bay watershed (along with existing strategies such as pollution prevention and product substitution).

- Due to major data gaps, this review could not evaluate the degree of UTC reduction due to implementation of on-site pollution prevention practices that are required under industrial and municipal stormwater permits. The potential impact of these practices could be considerable, as more than 2700 industrial sites have stormwater permits in the watershed, and more than a thousand MS4 facilities and public works yards are also subject to the regulations.

- The highest UTC levels tend to be generated in older urban watersheds, especially those with extensive industrial, commercial or transport land uses. Communities should target these UTC "hotspots" as they retrofit their watersheds to meet the Bay TMDL in order to promote even greater toxin reductions. Greater UTC reduction might be triggered for these potential hotspots if a modest numerical TSS reduction requirements were attached as a permit condition in the next generation of stormwater permits.

- The environmental benefits of UTC reductions may not be immediately realized, since they experience a long lag time from when they are first deposited in the watershed, slowly cycle through the stream network and then ultimately reach the Chesapeake Bay. Researchers project watershed lag times of several decades for the most persistent UTCs, which include PCBs, PAH, mercury, urban trace metals, flame retardants and OC pesticides. Other UTCs are expected to have watershed lag times measured over multiple years (petroleum hydrocarbons, OP pesticides and pyrethroid pesticides).

- It is important to keep in mind that while urban BMPs are effective at trapping and retaining UTCs, they are not necessarily being removed from the environment -- these persistent compounds could accumulate in BMP sediments over many decades to the point that they might trigger sediment toxicity guidelines.

- Older stormwater ponds built in the 1980s and 1990s appear to have the greatest risk of sediment toxicity. Monitoring has revealed that as many as 8 UTCs could potentially reach toxic levels in pond sediments, including PCB, PAH, Hg, Ni, Cr, Cu, Cd, and Zn. Some UTCs appear to be slowly declining in pond sediments (e.g., legacy pesticides), whereas the potential risk associated with other UTCs is simply not known at this time (e.g., PBDE, dioxins and pyrethroid pesticides).

- Despite these risks, pond sediments remain an acceptable option to (temporarily) trap toxics in the urban landscape for several reasons. First, the actual toxicity risk to aquatic life in the stormwater pond environment may be limited. The simplified food webs and low species diversity found in ponds may reduce the potential for bio-accumulation in urban fish and wildlife tissues. In particular, the benthic community in pond sediments that would be most exposed to UTCs is already highly degraded. Lastly, human fish consumption is extremely limited in stormwater ponds and recreational contact with sediments is uncommon.

- More research is urgently needed to measure UTC concentrations in pond sediments to fully assess the real toxicity risk and develop safer methods to maintain BMPs and clean out their sediments. Work is needed to determine which types of stormwater ponds pose the greatest risk (e.g., age, contributing land use, surface area or other factors) and to define the optimal places in the urban landscape where pond sediments can be safely disposed after they are cleaned out (e.g., fill, mix w/ bio-solids, landfill, etc.). In addition, further tissue tests are recommended to determine if toxins are bio-accumulating in the fish and wildlife that utilize the habitat created by urban BMPs.

- On a more positive note, recent research indicates that LID practices that are now required by all Bay states (e.g., bioretention, biofilters and swales) are very effective at trapping UTCs and may actually break them down as a result of microbial biodegradation and phytoremediation processes that occur in the soil media and/or vegetation. The risk of UTC bioaccumulation also appears to less pronounced in LID practices such as bioretention. These smaller practices do not create aquatic habitat and their maintenance schedule calls for frequent removal and replacement of surface mulch and sediments where most UTCs will be preferentially trapped.

- While the urban BMPs required by the Bay states are effective in trapping UTCs, the ultimate strategy is to keep the toxins out of the environment. Several other UTC management strategies could be very effective in reaching this goal. These include:

  - Past bans and/or product substitution efforts have been effective (e.g., lead, PCB, DDT).

  - New bans and product substitution may be warranted in some cases:

    - Coal tar sealant (for PAH)
    - Brake pads and rotors (for UTMs)
    - More sustainable roofing materials (for UTMs)
    - Further restrictions on the use of dichlorvos and other urban insecticides

  - Enhanced recycling and disposal outreach (batteries, thermostats fluorescent light bulbs).

  - Targeted street cleaning at older watersheds and industrial sites.

  - Enhanced air quality controls at power plants and incinerators to reduce UTC emissions.

Introduction

Background for the Study

One of the key outcomes under the Toxic Contaminant goal in the 2014 Chesapeake Bay Watershed Agreement is to "identify which best management practices (BMPs) might provide multiple benefits of reducing nutrient and sediment pollution as well as toxic contaminants in waterways".

The key issue is whether BMPs and wastewater treatment upgrades used to comply with TMDL can also help to substantially reduce toxin inputs to local waterways and the Chesapeake Bay. Such multiple benefits could provide significant cost savings to the Chesapeake Bay Partnership to simultaneously meet the Bay TMDL and reduce toxic contaminants in the environment.

Therefore, the broad purpose of this study was to:

> (1) Investigate the potential toxic contaminant reduction benefits that could be associated with the implementation of BMPs for sediment and nutrient reduction under the Bay TMDL.
>
> (2) Provide water resource managers with better BMP data to develop more effective local TMDLs and action strategies to control toxic pollutants in the watershed.

Selection of Priority Toxins

Thousands of potential contaminants exist in the water environment, so it was necessary to screen them down to a manageable number based on environmental risk in the Chesapeake Bay watershed. The degree of environmental risk was broadly defined based on three primary criteria, as previously established by CBP (2012):

> (a) Relative extent of the individual toxic contaminant in the Bay watershed based on prior monitoring data that indicate it has been detected in water, sediment, and/or tissue samples, as summarized in CBP (2012).
>
> (b) Relative severity of the human health and/or fish and wildlife impacts caused by the toxin in localized hotspots or across the entire Bay watershed.
>
> (c) Toxins that Bay states have directly linked to water quality impairments and/or fish consumption advisories in specific receiving waters within the Bay watershed.

Based on this screening analysis, a priority list was developed for 45 toxic contaminants in two broad categories (Table 1).

Table 1:Priority Contaminants Based on Environmental Risk[1]

#	Toxic Category	Individual Contaminants
	Group A: Urban Toxic Contaminants	
#	*Toxic Category*	*Individual Contaminants*
1	PCBs	Total PCBs
2	PAH's	Total PAH, benzo(a)pyrene, napthalene
3	Petroleum Hydrocarbons	TPH, oil and grease, benzene
4	Mercury	Hg, Me-Hg
5	Urban Trace Metals	Cd, Cu, Pb, Zn
6	Other Trace Metals	As, Cr, Fe, Ni
7	Pyrethroid Pesticides	Bifenthrin
8	Legacy OC Pesticides [2]	DDT/DDE, dieldrin,
9	Legacy OP Pesticides [2]	Chlordane, diazinon, chloropyrifos
10	Plasticizers	Phthalates
11	Flame Retardants	PBDE
12	Dioxins	Dioxins and furans
	Group B: Agricultural, Wastewater and Biosolid Contaminants	
13	Cropland Herbicides	Atrazine, simazine, metolachlor, acetochlor, glyphosate
14	Veterinary Pharmaceuticals	Tetracyclines, sulfonamides
15	Biogenic Hormones	Estradiol, estrone, testosterone
16	Human Pharmaceuticals	acetaminophen, naproxen, ibuprofen, triclosan tetracycline and sulfonamides

Codes: PCB's = Polychlorinated Biphenyls, PAH= Polycyclic Aromatic Hydrocarbons, HPCP= Household and Personal Care Products, PBDE = Polybrominated Diphenyl Ether, TPH = Total Petroleum Hydrocarbons. OC= organochlorine, OP =organophosphate.

Notes:

[1] As defined by the extent and prevalence of the contaminant in the Bay watershed, as well as actual impairments or fish advisories, as defined in CBP (2012).

[2] Legacy pesticides refer to insecticides that have been banned or phased out, but have such long half lives that they are still detected in the environment; this list is based on a national assessment of pesticide prevalence in streams and groundwater by Gilliom et al (2006).

Scope of Literature Review

CSN conducted an international literature review to identify key research papers on the priority toxins. The review investigated:

- Key characteristics, sources, generating sectors and watershed pathways associated with priority toxins
- Measured concentrations in stormwater runoff, groundwater and sediments
- Measured or inferred removal of toxins associated with current urban and agricultural BMPs
- Measured concentrations and retention of toxins within BMP sediments
- Additional pollution prevention practices that can prevent how much of the toxin is released to the environment

More than 250 research papers and reports were discovered during the review, including several research databases and review papers that contained an additional citations (Figure 1). A spreadsheet was developed to organize the papers by the toxin, author, title and geographic region, which is available from the Chesapeake Stormwater Network.

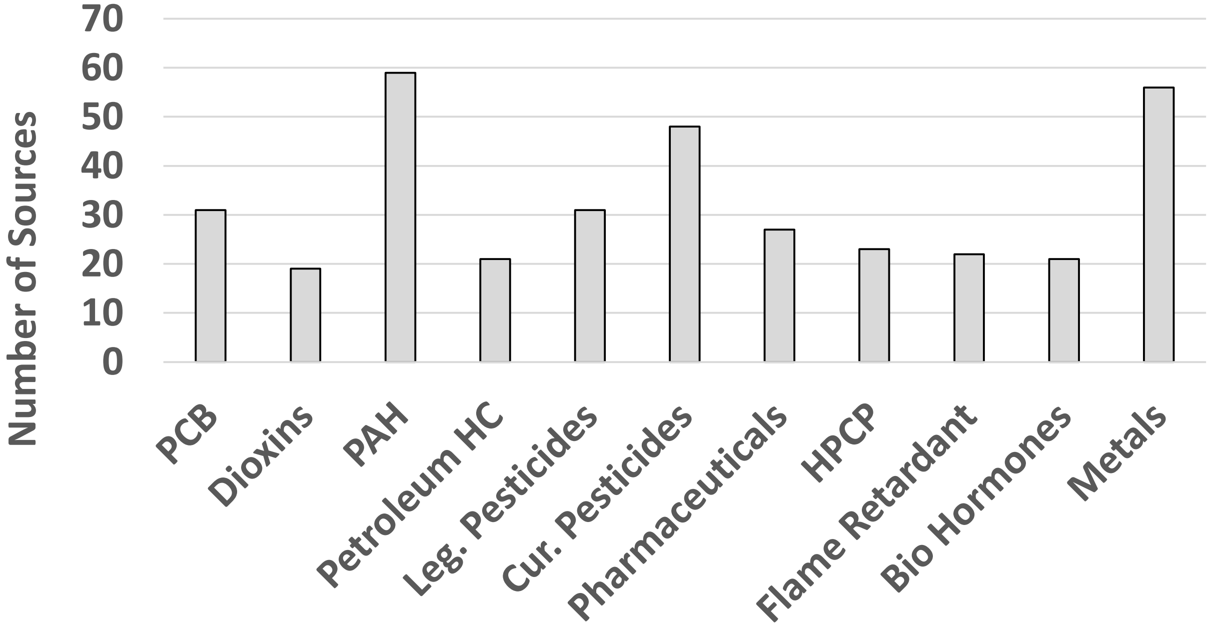


Figure 1: Number of studies evaluated by class of toxic contaminant

Comparative Data Quality for the Different UTCs

One of the primary goals of the review was to evaluate the quality of the available monitoring data for each class of toxic contaminants, with respect to its concentration in stormwater runoff and urban sediments, and its removal and/or retention within urban BMPs.

Table 2 compares the relative quality of available monitoring data for the 12 urban toxic contaminants. As can be seen, data quality ranges from very low to very high, depending on the urban toxic category.

Table 2. Data Quality for Urban Toxic Contaminants

Urban Toxin Category	Runoff EMCs?	Sediment Conc.	Air Deposition	Street Solids	BMP Removal	BMP Sediment
PCBs	VL	M	VL	VL	VL	L
PAH	M	H	L	M	M	M
TPH	M	VL	ND	L	M	L
Hg	H	H	H	VL	L	L
UTM	VH	VH	H	M	VH	H
OTM	H	H	M	L	M	M
PP	M	M	NA	VL	L	L
Legacy OCP	VL	L	NA	ND	ND	L
Legacy OPP	M	L	NA	ND	VL	VL
Plasticizers	VL	L	NA	ND	ND	VL
PBDE	VL	L	VL	ND	VL	VL
Dioxins	VL	VL	VL	ND	ND	ND

UTM: Urban Trace Metals (Cd, Cu, Pb and Zn)
OTM: Other Trace Metals (As, Cr, Fe and Ni)
OCP: Organochlorine Pesticides
OPP: Organophosphate Pesticides
EMC: Event Mean Concentration
ND: No Data

VL = Very Low (<3 studies, none from CB)
L = Low (< 5 studies, some from CB)
M = Moderate (5 to 10 studies)
H = High (10 to 25 studies)
VH = Very High (>25 studies)
NA: Not Applicable

Section 1:
Urban Toxic Contaminants, Sediment and Stormwater BMPs

1.1 Defining an Urban Toxic Contaminant

Broadly speaking, many toxic contaminants had similar sources and characteristics and could be classified as 'urban toxic contaminants'. This class of contaminants shared six common features, as follows:

1. The toxin was primarily associated with the urban land use sector, as indicated by higher measured concentrations or loads.
2. The sources of the toxin were either derived from within the urban sector or by the wash-off of toxins deposited on impervious surface from the atmosphere.
3. The predominant pathway for transporting the toxin in the watershed is via urban stormwater runoff, rather than dry weather flow or groundwater migration.
4. The toxin has "sediment-like characteristics" that may include one or more of the following:
    - A high affinity to bind or adsorb to sediment, street solids, and/or organic carbon particles
    - Found in a particulate form and associated with a larger particle size diameter
    - A low Henry's Law constant (K_H)
    - A high soil organic carbon-water partition coefficient (K_{OC}) (i.e., hydrophobic)
    - Low solubility in water
    - Long half-life for transformation
5. The toxin is generated or produced in an upland landscape position in the watershed where it can be effectively treated by an urban BMP that captures surface runoff.
6. Some physical evidence exists that the toxin is actually captured and/or retained within an urban stormwater BMP.

Based on these criteria, the following 12 toxins were classified as urban toxic contaminants.

1. PCBs
2. PAHs
3. Petroleum Hydrocarbons
4. Mercury

5. Cadmium, Copper, Lead and Zinc
6. Arsenic, Chromium, Iron and Nickel
7. Pyrethroid Pesticides
8. Legacy Organochlorine (OC) Pesticides
9. Legacy Organophosphate (OP) Pesticides
10. Plasticizers (phthalates)
11. Flame retardants
12. Dioxins/Furans

The UTC designation is important as it enables watershed managers to target urban watersheds with effective stormwater BMPs to reduce toxin loads to receiving waters.

It should also be noted that a UTC designation does not imply that other sectors (e.g., wastewater, agriculture or feedlots) do not also contribute to the overall toxin load in the Bay watershed, only that the urban sector produces the largest share.

1.2 Current and Future Urban BMP Coverage in the Watershed

Stormwater BMPs have been required at new development projects in most communities in the Bay watershed for the last three to four decades. Many communities have hundreds or even thousands stormwater BMPs within their jurisdiction. Consequently, a considerable fraction of developed land in the Bay watershed is currently served by urban BMPs.

The best estimate of urban BMP coverage can be gleaned from the current version of the Chesapeake Bay Watershed Model. BMP reporting for the 2014 progress run indicate that urban BMPs serve about a third of the total impervious cover across the watershed (Sweeney, 2015).

This coverage estimate may change in the next generation of the model, as jurisdictions improve their BMP reporting and clean up their historic BMP databases. In addition, BMP coverage will vary regionally and from state to state. It should also be noted that some of the older BMPs may be less effective in removing sediment and other pollutants due to their smaller size, less sophisticated design and/or poor maintenance condition.

Some indication of the future BMP coverage in the watershed can be estimated from the urban sediment reduction targets that the Bay States defined in their Watershed Implementation Plans (WIPs) to meet the Bay TMDL (Table 3). These plans set a target sediment reduction from existing development that ranges from 5 to 50%, with an average of about 30%.

If it is assumed that (a) most communities will use retrofit BMPs to achieve these sediment reductions, and (b) the reductions will be taken from existing acres of urban land, it is conceivable that BMP coverage across the watershed could climb to as much as 50% by 2025.

Of course, this estimate of future BMP coverage should be considered provisional, if not speculative, since most communities are struggling to implement retrofits on existing development.

Table 3. Bay State Urban TSS Reduction Targets by 2025

Bay Jurisdiction	Urban TSS Load Reduction *
Delaware	5%
D.C.	16%
Maryland	29%
New York	10%
Pennsylvania	50%
Virginia	30%
West Virginia	50%

* from existing development. Source: Antos (2013)

1.3 TSS as the Benchmark Removal Rate for UTC Removal

This section outlines the rationale for using TSS removal rates as the initial benchmark for estimating UTC removal rates, when little or no monitoring data are available to derive an estimate.

As noted in the next section, there is extensive monitoring data to establish sediment removal rates for a wide range of urban BMPs (SPS EP, 2013). Sediment removal rates are consistently higher than those reported for nitrogen or phosphorus (SPS EP, 2013). The lower removal capability observed for nutrients reflect the fact that about half of nitrogen and phosphorus are found in soluble form, which is very hard to remove without the aid of algal, plant or microbial removal mechanisms.

By contrast, both suspended sediment and UTCs share many of the same characteristics when it comes to BMP removal. To begin with, most UTCs bind, adsorb or otherwise attach to sediment particles. Unlike nutrients, most UTCs are hydrophobic, have very limited solubility and often have a strong affinity for organic matter.

Both sediments and UTCs are also relatively inert, persistent, and not very biodegradable. In addition, both are often associated with fine and medium-grained particles that are easily entrained in stormwater runoff. Given their particle size, both are subject to high removal rates simply through gravitational settling in the water column and/or filtering through sand, soils, media or vegetation.

Therefore, in the absence of reliable data on UTC removal, it is recommended that the default value be set to the TSS removal rate for the qualified urban BMPs that have been assigned by CBP (i.e., Table 4 rates or values determined from the ST or RR curves).

The basic idea is that the UTC removal rate can be adjusted upwards or downwards from the sediment removal benchmark, depending on the characteristics and properties of the individual toxin.

For example, the UTC removal rate should be adjusted lower than the sediment benchmark if any of the following conditions apply:

- A significant fraction of the UTC is present in soluble form (i.e., 25% or more)
- The UTC is predominantly associated with very fine-grained particles (i.e., silt and clay particles less than 62 microns in diameter)
- Is prone to release after being trapped in BMP sediments (e.g., methylation in hypoxic and organic-rich environment of constructed wetland sediments)

By contrast, the UTC removal rate can be adjusted higher than the sediment removal benchmark when the UTC is:

- Seldom or never found in soluble form
- Predominantly associated with medium or coarse-grained particles that are easier to settle (i.e., more than 250 microns in diameter)
- Documented to persist and accumulate within BMP sediments over time

This benchmark approach can be used to estimate UTC reductions associated with stormwater BMPs for local TMDLs and to estimate the additional toxic removal benefits achieved by the Chesapeake Bay TMDL.

1.4 Sediment Removal Rates for Urban BMPs

This section reviews the sediment removal rates for urban stormwater BMPs in the watershed. The CBP has established an expert panel process to define pollutant removal rates for a wide range of urban BMPs, as shown in Table 4. As can be seen, most urban BMPs are effective at reducing sediment levels in stormwater runoff (ranging from 45 to 90% reduction). This suggests that they will also be effective at removing urban toxic contaminants that behave like sediments.

Most urban BMPs provided in Table 4 are structural practices with a design life measured in decades that will remove sediments year after year. A few BMPs are non-structural practices that must be applied every year to effectively remove sediments (e.g., street cleaning).

In addition, several urban BMPs were excluded from Table 4 since they primarily focus on nutrient removal as opposed to sediment removal. These include urban nutrient management, urban tree planting, floating treatment wetlands, septic system upgrades, and nutrient discharges from grey infrastructure. Also, while urban stream restoration and shoreline management are effective practices for removing sediment, they were excluded from the analysis. It is doubtful whether the sediment reduction achieved by these practices would effectively reduce toxin loads, since they are located so far downstream in the watershed.

Table 4
Approved CBP BMP TSS Removal Efficiency Rates for Different Urban Stormwater Practices [1]

Urban Stormwater Practices		Removal
Stormwater Retrofits [2]		45 to 85%
New Runoff Reduction (RR) Practices [3]		45 to 80%
New Stormwater Treatment (ST) Practices [4]		40 to 75%
Wet Ponds		60
Constructed Wetlands		60
Dry Extended Detention Ponds		60
Infiltration		95
Filtering Practices (Sand Filters)		80
Bioretention	C & D w/UD	55
	A & B w/ UD	80
	A & B w/o UD	90
Permeable Pavement	C & D w/UD	55
	A & B w/ UD	70
	A & B w/o UD	85
Grass Channels	C & D w/o UD	50
	A & B w/o UD	70
Bioswale	aka dry swale	80
Urban Stream Restoration [5]		NA
Street Cleaning [6]		0 to 30
Enhanced Erosion and Sediment Control [7]		NA

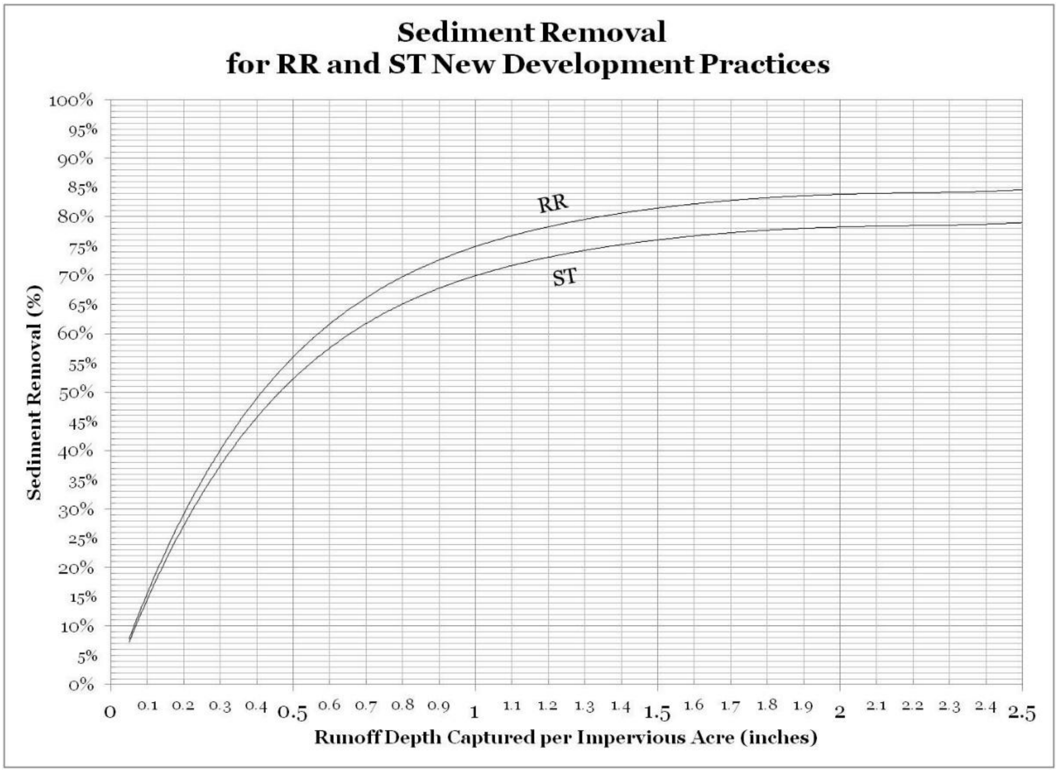
[1] Unless otherwise specified, the TSS removal rates are provided in Appendix B of the State Stormwater Performance Standards Expert Panel Report (SSPS EP, 2013)
[2] Stormwater Retrofit Expert Panel Report (SR EP, 2013)
[3] State Stormwater Performance Standards Expert Panel Report (SSPS EP, 2013)
[4] RR= runoff reduction practices ST= stormwater treatment practices, as defined in SSPS EP (2013)
[5] Urban Stream Restoration Expert Panel Report (USR EP, 2013).
[6] Street and Storm Drain Cleaning Expert Panel Report (SSDC EP, 2015)
[7] Erosion and Sediment Control Expert Panel Report (ESC EP 2014)

Notes: HSG = Hydrologic Soil Group: A, B, C, D soils UD = under drain

In recent years, two expert panels have simplified the approach to calculating sediment removal produced by the entire range of urban BMPs (SSPS EP, 2013 and SR EP, 2013). The new approach relies on a series of curves that express sediment removal as a function of how much runoff volume is captured and whether runoff reduction (RR) or stormwater treatment (ST) practices are employed (Figure 2).

The curves apply to BMPs used for new development projects, as well as stormwater retrofits that serve existing development. Over the next several years, the sediment removal rates for most urban BMPs will be derived from these curves, rather than the fixed removal rates for individual BMPs, as shown in Table 4.

Figure 2: Runoff Reduction (RR) and Stormwater Treatment (ST) Curves to Define Sediment Removal Rates for Urban BMPs and Stormwater Retrofits



1.5 Municipal and Industrial Pollution Prevention Practices

Pollution prevention practices are required at both industrial and municipal operations by NPDES stormwater permits. These practices are primarily intended to prevent rainwater contact with potentially toxic substances utilized at these sites, and prevent them from being carried off-site by stormwater runoff. The basic strategy of "no exposure" is supplemented with operational practices to prevent discharges from:

- Leaking dumpster and compactors
- Uncovered fueling islands
- Loading or unloading docks
- Outdoor wash-water used to clean vehicles, equipment or siding
- Eroded sediments from unpaved areas
- Bulk materials, vehicles and equipment that are stored outside
- Outdoor vehicle maintenance (used fluids, batteries, etc.)

These operational practices, in turn, are supplemented by various "good-housekeeping practices" that keep the surface of the site in good condition, such as:

- Routine cleaning or sweeping of paved areas of the site
- Regular litter and debris control at trash hotspots
- Equipment for rapid spill response and containment
- Frequent inspections of storm drains for illicit discharges
- Green landscaping practices that minimize pesticide use

More than 2,500 industrial sites are subject to NPDES stormwater permits in the Chesapeake Bay watershed (Table 5). While no precise estimate exists as to the number of individual municipal operations that are permitted, more than 1,000 communities have been issued municipal MS4 permits in the Bay watershed to date.

Table 5. Impervious Cover and Industrial Stormwater Permits in the Watershed[1]

Bay State	Permitted Sites in Bay Watershed [2]	Acres of Impervious Cover [3]
Delaware	52	489
DC	42	395
Maryland	886	6853
Pennsylvania	850	7990
Virginia	770	8509
West Virginia	147	1381
Total	**2747**	**25,617**

[1] Data analysis based on state reports

[2] Includes multi-sector general permitted sites (MGSP) --No watershed estimate of permitted sites was available for New York

[3] Actual data for MD and VA, and estimated for all other states using a factor of 9.4 acres per each permitted site.

During our review, we could find no quantifiable data to estimate the potential reduction in UTC inputs to the Bay watershed as a direct result of compliance with existing industrial and municipal stormwater permits. This data gap is not surprising, given that it is much harder to detect the effect of keeping pollutants out of stormwater than to measure the effect of BMP treatment after pollutants get into stormwater. Although no reduction rate can be currently assigned for pollution prevention practices, they should play a considerable role in reducing toxin inputs to both local waterways and the Chesapeake Bay.

Section 2: Polychlorinated Biphenyls (PCBs)

2.1 Overall Findings on Polychlorinated Biphenyls

- While evidence suggests that PCB concentrations are declining in urban estuarine sediments, legacy PCBs are still detected in fish and wildlife tissue nearly four decades after they were banned.

- Based on the review, the overall quality of the available PCB monitoring data is limited. On one hand, there were useful data on PCB sources, generating sectors, and pathways, as well as limited data on PCB concentrations in urban stormwater and sediments. On the other hand, there were only a handful of studies that evaluated how urban stormwater BMPs trap and retain PCBs. In addition, most of the research has occurred outside the Chesapeake Bay watershed.

- Research in other estuaries, such as San Francisco Bay, have also documented a decline in PCB inputs. At the same time, they also forecast that it may take many decades for these persistent chemicals to stop bio-accumulating in the estuarine food chain. The main reason is that PCBs contaminate soils and sediments which slowly move through the watershed in a recurring cycle of mobilization, deposition and re-suspension.

- PCBs have a very strong association with highly urban watersheds, especially older industrial areas where PCBs were once used. While PCB monitoring data is limited, it is clear that it behaves much like a sediment particle, and is primarily conveyed through urban watersheds by stormwater runoff.

- Considering the pervasive impact of PCBs in the urban environment, it is remarkable how little monitoring has been conducted to measure the degree of PCB removal by urban BMPs. Given the characteristics of PCBs and limited settling column experiments, it is estimated that PCB removal rates will be comparable to suspended sediment removal rates for most urban BMPs.

- Much of the PCB load moving through urban watersheds is potentially treatable by stormwater retrofits, and a significant fraction of the existing load may already be trapped within existing stormwater BMPs that serve about a third of existing urban land in the Chesapeake Bay watershed. Targeted street and storm drain cleaning in industrial catchments may also be an effective strategy to control PCB hotspots in the urban landscape.

- The effectiveness of stormwater practices in trapping PCBs poses some risk for contamination of BMP sediments. Elevated PCB levels in BMP sediments, however, may not pose a major environmental risk, given the simplified food chain that exists in most stormwater pond communities. Likewise, the risk to human health is low, given fish consumption is rare in stormwater ponds, and few other modes of direct human exposure are likely..

- The presence of persistent levels of PCBs in BMP sediments may have important implications for stormwater managers regarding how BMP sediments are managed in the long-term. Special emphasis should be placed on testing stormwater sediments from older industrial sites where the risk is presumably the greatest.

- While BMP and retrofits can reduce PCB inputs to the estuary, other PCB management practices will continue to be needed, as well. These include PCB pollution prevention practices, demolition controls during redevelopment projects and continued cleanup of legacy industrial sites and hotspots.

2.2 Background on PCBs

PCBs are a group of synthetic organo-chlorine chemicals widely used as a dielectric and coolant fluid in transformers and capacitors. There are no natural sources of PCBs in the Bay watershed. The U.S. banned the production of PCBs in 1977 out of concern for their persistence in the environment and their bio-accumulation in human, fish and wildlife tissue. PCBs are listed as a probable human carcinogen by the EPA.

Although PCBs have not been produced for more than 35 years, they can still be used in existing transformers and other products so that continues to be potential for accidental release. In addition, erosion of soils that were historically contaminated with PCBs are another key source of PCBs to the urban environment.

2.3 PCBs: Environmental Risk and Trends

All of the jurisdictions within the Chesapeake Bay watershed have identified PCB impaired waters and issued widespread fish consumption advisories. CBP (2012) reviews the aquatic life and human health impacts caused by PCBs, and documents the widespread impairments in waters, sediments and fish tissue across the Chesapeake Bay watershed.

PCB concentrations in many of the fish and wildlife tissues sampled in the Chesapeake Bay have not declined in almost four decades since the production of PCBs was banned (CBT, 2012). Likewise, PCB levels are still a concern in San Francisco Bay as current data show fish, wildlife and humans are all at or near health effects thresholds (Davis et al, 2007).

The trend in PCB inputs appears to be declining slowly. For example, Van Metre and Mahler (2005) studied sediment cores in 38 lakes across the nation and concluded that PCB levels had declined in 25% of the lakes and remained the same in the rest.

Velinski et al (2011) investigated trends in PCB levels over time in dated cores in estuarine sediments of the highly urban Anacostia river. The peak in sediment PCB levels occurred in the 1957 to 1973 time frame (~3,000 ng/g), but gradually declined over the next three decades (~100 to 200 ng/g), presumably as a result of the ban in new PCB production. Velinski et al (2011) also detected a shift in sediment PCB composition over time, with lower molecular weight PCB compounds becoming more prevalent over time.

Davis et al (2007) reviewed efforts to manage PCBs in the San Francisco Bay region. While they concluded that PCBs inputs to the San Francisco Bay had declined substantially over time, they estimated it would take decades to centuries to eliminate them from the environment, due to the "recurring cycle of mobilization, deposition and re-suspension" of particles, soils and sediments that were historically contaminated with PCBs.

2.4 Sources and Pathways for PCBs

Despite the fact that new PCBs have not been manufactured for 35 years, there are some historic sources in the watershed, such as leaking transformers, capacitors in small appliances and fluorescent light ballasts. Significant quantities of PCBs are still used in existing electrical transformers. For example, Davis et al (2007) estimated that there were 420,000 lbs of PCBs still in use in the San Francisco Bay region, as of 2007.

The second main source of PCBs is atmospheric deposition, especially when they fall onto impervious surfaces and are washed off during storm events. Bressy et al (2012) studied a recently developed residential catchment and concluded that atmospheric deposition was the sole source of PCBs in stormwater runoff.

The third PCB source are eroded or re-suspended soil particles that were contaminated by PCBs in the past, and are gradually working their way through the watershed. Several researchers have noted that high PCB levels in hotspots in close proximity to legacy contaminated sites and/or older industrial watersheds where PCBs were used in the past.

In the Chesapeake Bay region, the highest PCB levels in sediments are associated with highly urban watersheds, such as the Patapsco River and Back River estuaries in Baltimore (King et al, 2004) and the Anacostia River in the Washington metropolitan area (Velinsky et al, 2011). King et al (2004) also reported a very strong association between watershed impervious cover and PCB contamination in white perch in small sub-estuaries across the Chesapeake Bay.

Davis et al (2007) also concluded that urban runoff was the major source of PCBs to the San Francisco Bay and loads increased during the more intense storm events. The highest urban PCB loads were delivered from older industrial watersheds.

2.5 PCB Characteristics

Some of the key characteristics of PCBs are profiled in Table 6. PCBs are relatively insoluble in water and are hydrophobic compounds that readily dissolve in organic solvents and lipid tissues. There are 209 PCB "congeners" that reflect the variation of chlorination around biphenyl rings for the compound. PCBs are organic contaminants that are lipophilic, persistent and bio-accumulative in the environment. PCBs are mostly found in a particulate phase, are associated with coarse-grained particles, and behave like sediment particles.

Table 6. Key Chemical Characteristics of PCBs

- Solubility $1x\ 10^3 – 1.6 \times 10^5$ µg/L
- Log K_{OC} 4.6 - 6.9
- K_{Henry} 0.8 – 240 Pa m^3/mol
- Half-life 19.7 years
- High boiling point
- 209 congeners
- Associated with heavier particles
- Found mostly in particulate phase versus dissolved phase
- Able to volatilize, though rate is slow

2.6 PCB Concentrations in Urban Runoff and Urban Sediments

The limited available data on PCB concentrations are summarized in Table A-1, whereas Table A-2 summarizes PCB levels in stormwater runoff and creeks and rivers, which are detected at the parts per trillion level.

The most extensive regional PCB sampling effort has occurred in the San Francisco Bay area over the last decade. Gilbreath et al (2012) concluded that urban stormwater runoff was the most dominant source of PCBs to the estuary. The typical PCB concentration in stormwater runoff in urban watersheds ranged from 4 to 110 ng/l (median EMC of 14.5 ng/L), with the highest concentrations found in older urban areas, especially those with legacy industrial sites. Gilbreath et al (2012) also noted a strong association of high turbidity levels and elevated PCB concentrations.

Ko and Baker (2004) measured PCB concentrations in the Susquehanna River basin, and found that more than 75% of the total PCB load was associated with the particulate phase. Bressy et al (2012) in a study of a French catchment found that 75 to 100% of PCBs were associated with the particulate phase.

2.7 Estimated PCB Removal by Urban BMPs

Remarkably little monitoring has been conducted to assess whether urban stormwater BMPs can remove PCBs. The most comprehensive study was conducted in the San Francisco Bay region. Yee and Mckee (2010) conducted a series of settling column experiments to measure PCB settling rates for stormwater runoff and stream sediments from urban watersheds in the San Francisco Bay area.

They found that 55% of PCB particles in stormwater settled out within 30 minutes, and 30% of re-suspended creek sediments settled out within 2 minutes. Based on these experiments, Yee and McKee (2010) concluded that PCB behaved very much like a sediment particle, and that effective settling of moderate to larger sediment particles was capable of achieving a minimum 50% PCB removal.

A European study found that urban tree pits and their associated bacteria have the capability to degrade PCBs in the soil (Leigh et al, 2006). This finding suggests that practices such as bioretention which have aerobic media conditions may also promote the growth of PCB-reducing bacteria.

Mangarella et al (2012) evaluated a range of stormwater treatment options in the San Francisco Bay area in order to meet a 90% PCB load reduction established in a regional TMDL. They determined that stormwater retrofit practices should be targeted to urban watersheds with current or historic industrial activity. They also concluded that stormwater retrofits and stormwater BMPs installed at redevelopment projects and brown field sites were the most effective strategies to reduce PCBs in urban watersheds.

2.8 PCB Accumulation in BMP Sediments

Only one study has investigated whether PCBs accumulate in BMP sediments. Parker et al (2009) evaluated PCB levels in stormwater pond sediments in Arizona, and concluded many of them exceeded preliminary sediment remediation guidelines, which would require special sediment handling and disposal techniques.

2.9 Other PCB Management Strategies

Other PCB management practices should be coupled with stormwater BMP and retrofits to reduce PCB inputs to the estuary. The first step involves clean-up at legacy industrial sites, whereby PCB contaminated sediments are removed, buried or otherwise sequestered. Mangarella et al (2012) also recommends demolition controls to prevent PCB releases during redevelopment projects.

Another strategy involves frequent street cleaning targeted towards older industrial catchments, using advanced sweeping technology (SSDC EP, 2015). The effectiveness of additional PCB pollution prevention practices are evaluated in Mangarella et al (2012).

Section 3: Polycyclic Aromatic Hydrocarbons (PAH)

3.1 Key Findings on Polycyclic Aromatic Hydrocarbons

- PAHs are ubiquitous in urban sediments across the Chesapeake Bay watershed, and, on a national basis, have been found to contribute more to total sediment toxicity than all other toxin categories combined.

- The quality of existing monitoring data to characterize PAH sources, pathways and loadings in the watershed is classified as moderate, with a few important data gaps in our understanding.

- PAH meet all six criteria to be classified as an urban toxic contaminant -- they are strongly associated with urban land, have unique (and controllable) urban sources, are delivered in urban stormwater, behave in the same manner as sediment, originate in an upland landscape position and are captured and retained by stormwater BMPs.

- Due to the high cost and difficulty of sampling, only a handful of research studies have evaluated whether stormwater BMPs have the capability to remove PAH. Based on this limited monitoring data and given its basic characteristics, PAHs are considered to be highly treatable by most urban stormwater BMPs -- with expected removals slightly greater than those observed for total suspended solids.

- Three recent studies have shown that PAH compounds accumulate and persist in BMP sediments at levels that exceed sediment guidelines, and which may warrant special sediment handling and disposal methods. The risk of sediment PAH contamination is most pronounced within older stormwater ponds, whose hypoxic bottom waters prevent rapid biodegradation of PAH compounds in the sediments. More research is needed to evaluate the comparative risk of PAH contamination in pond sediments, based on the contributing land use, age of the facility or other factors.

- The largest and most controllable source of PAH are the coal tar sealcoats applied to extend the life of asphalt parking lots. Numerous studies have documented that the sealcoats generate a very high PAH load, and several state and local governments in the Chesapeake Bay have banned their use. Imposing a Bay-wide coal tar sealcoat ban would not only be an effective strategy to reduce PAH inputs to the estuary, but would also minimize the risk of PAH sediment contamination in upland stormwater ponds.

- A comprehensive PAH reduction strategy for the Chesapeake Bay might combine the seal coat ban with more widespread installation of stormwater retrofits and more stringent vehicle emission controls.

- Like other urban toxic contaminants, however, there is expected to be a multi-decade lag time before the environmental benefits are fully realized, given how long it will take for past PAH inputs to cycle through the watershed.

3.2 Background on PAHs

PAHs are a class of hundreds of compounds that are composed of carbon and hydrogen in structures of two or more benzene rings. PAH readily adsorb to sediments in water and persist for a long time (half-lives of up to 5 years).

3.3 PAHs: Environmental Risk and Trends

PAHs are detected in stream, river and estuarine sediments across the Chesapeake Bay watershed, with the highest concentrations occurring in highly urban watersheds, such as the Elizabeth and Anacostia rivers and the Baltimore harbor.

Nowell et al (2013) conducted a comprehensive review of PAH levels in stream sediments at 98 urban streams within seven metropolitan areas across the US (none of which were located in the Chesapeake Bay watershed). PAHs were detected in 98% of the urban stream samples, and had the highest concentration of any toxicant monitored during the USGS assessment. More importantly, Nowell et al (2013) found that PAH contributed more to total toxicity than all other contaminants combined (e.g., PCBs, trace elements, organo-chlorine and other pesticides).

Velinsky et al (2011) measured PAH in six sediment cores from the tidal Anacostia river. The dated cores showed the highest PAH levels at depth, and lower levels at the surface. This indicated that PAH loads have declined in the last two decades in this highly urban watershed, which presumably reflects changes in PAH production, use and controls. The surface sediments had a mix of PAHs split between combustion and petrogenic sources.

3.4 Sources and Pathways for PAH

PAH sources include combustion of fossil fuels, fires, driveway and parking lot sealcoats, and creosote treated wood.

Dickhut et al (2000) conducted a comprehensive study of PAH sources in the Chesapeake Bay, and found most of the PAHs measured in air, rain, surface waters and urban estuarine sediments were derived from automotive sources in the watershed, rather than coal burning. By contrast, most of the PAH in the surface sediments of the Chesapeake Bay were predominantly derived the burning of coal.

Hwang and Foster (2006) sampled the inputs of PAH to the tidal Anacostia River at six upstream monitoring stations during storm and dry weather events. The highest PAH concentrations were recorded during storm flow and ranged between 1,500 and 12,500 ng/l. The majority of the PAH measured during storm flow was in the particulate phase (70 to 97%). Hwang and Foster (2006) observed that high molecular weight PAHs predominated during most storms, indicating a higher contribution of automotive-derived PAH compounds (i.e., pyrogenic).

Ko and Baker (2004) sampled PAH inputs in the Susquehanna river basin, and found that about 75% were carried in the particulate phase, particularly by particles enriched with organic matter. The highest loads and concentrations were associated with high river flow, and were linked to episodes of river erosion and sediment re-suspension that frequently occurred in the winter and spring. Given the long history of coal mining in the river basin, it not surprising that coal and coal combustion were a major source of the PAHs in this portion of the Chesapeake Bay watershed. Ko and Baker (2004) also found that PAH levels in Susquehanna river sediments were about twice as high as those found in sediments of the upper Chesapeake Bay.

Brown and Peake (2006) found that street solids were a major source of PAHs in urban stormwater runoff, as measured in two catchments in Dunedin, New Zealand. Stein et al (2006) found that urban runoff was a significant source of PAH in Los Angeles, California. Stein et al (2006) did not detect significant differences in PAH concentrations among different urban land uses, and suggested that this was due to ubiquitous vehicle emissions throughout the region.

A regional monitoring initiative in San Francisco Bay reported that urban stormwater runoff was responsible for 57% of the total PAH regional load, followed by river inflows (28%), wastewater effluent (10%), deposition over open water (8%) and dredging (2% -- Oros et al, 2007).

Bressy et al (2012) monitored the concentration of PAH in rainwater and runoff in a small urban residential catchment in France, and found that atmospheric deposition of PAH could only account for 25% of the observed PAH export from the catchment. They concluded that the majority of the PAH load was generated internally within the catchment by vehicle emissions.

Selbig et al (2013) found very high PAH levels in Wisconsin urban watersheds, as measured in street solids, suspended sediment in stormwater and sediments that had settled on the stream bed. In many cases, the PAH levels were considered to be toxic to aquatic life.

The Key Role of Coal Tar Seal Coats

Mahler et al (2005) was the first study to identify coal tar sealants as a major source of PAH loads in urban watersheds in Austin, TX. This finding has since been corroborated and reinforced by other studies around the country including MN (Crane et al, 2014), seven U.S. metropolitan areas (Nowell et al, 2013), Fort Worth, TX (Yang et al, 2010) and the eastern U.S. (Van Metre and Mahler, 2010). Selbig et al (2013) also implicated coal tar sealants from asphalt driveways as the primary source of PAH in a study of urban watersheds in Wisconsin. Gilbreath et al (2012) also concluded that coal tar sealants and dust were a major source of the PAH load in San Francisco Bay, with automotive emissions a distant second.

3.5 PAH Characteristics

PAHs can either be petrogenic or pyrogenic in nature. The former are derived from petroleum or coal, while the later are derived from fossil fuels or wood. Petrogenic PAHs have lower molecular weights, less than 4 carbon rings, and are more soluble. Pyrogenic PAHs have higher molecular weights, more than 4 carbon rings and tend to be associated with particulates. In general, since PAHs are hydrophobic, they are often found in a particulate phase, and collectively behave as if they were a particle of sediment or organic matter.

Bathyl et al (2012) investigated the particle size distribution of urban creek sediments in Alabama. They found that PAH compounds followed a bi-modal distribution, attaching to both very fine sediment particles and very coarse organic particles. PAHs were more strongly associated with volatile organic carbon than total suspended solids in the stream samples.

3.6 PAH Concentrations in Urban Runoff and Urban Sediments

Appendix A presents a series of tables that compare PAH concentrations in the urban environment, as follows:

- Sediments measured in the Chesapeake Bay watershed (Table A-5)
- Sediments measured outside the Bay watershed (Table A-6)
- Suspended sediments carried in urban stormwater runoff (Table A-7)
- Stormwater runoff from parking lots with various types of seal coats (Table A-8)
- Stormwater pond sediments (Table A-9)
- Stormwater runoff from urban catchments (Table A-10)
- Stormwater runoff data from larger creeks and rivers (Table A-11)

Gilbreath et al (2012) has conducted extensive stormwater sampling of PAH in the San Francisco Bay region. High PAH concentrations were routinely measured in urban stormwater runoff, with an mean flow-weighted concentration of 9,600 ng/l. Gilbreath et al (2012) also reported a modest first flush effect, with concentrations being significantly higher at the beginning of a storm than at the end.

3.7 Estimated PAH Removal in Urban BMPs

Only a handful of research studies evaluated whether stormwater BMPs have the capability to remove PAH, presumably due to the high cost and difficulty of sampling this group of compounds. Based on the existing studies and the basic characteristics of PAH, these pollutants are considered to be highly treatable by most urban stormwater BMPs. PAH removal is expected to be comparable to that observed for total suspended solids, given that both are effectively removed by sedimentation and filtration.
Roinas et al (2014) monitored PAH levels in stormwater runoff as it passed through a series of swales and ponds along a UK motorway. Roinas found that ponds and swales were highly effective at removing the heavier, hydrophobic PAHs (e.g., phenanthrene,

flouranthene, and pyrene). By contrast, the ponds and swales were less effective at removing lighter and more soluble PAHs, such as napthalene.

DiBlasi et al (2009) found that bioretention was highly effective in reducing PAH levels in stormwater runoff, and reported a 87% reduction in the mass of PAH in a field study of a bioretention area in College Park, MD.

LeFevre et al (2014) investigated the primary pollutant removal mechanisms responsible for the high performance of bioretention areas, focusing on experiments with napthalene. Most of the napthalene adsorbed to mulch and media (56 to 73%), about 12 to 18% bio-degraded within the cell, about 10% was taken up by plants, and less than 1% volatilized into the atmosphere.

Bathyl et al (2012) recommended a two stage strategy to remove PAH from urban runoff. The first stage involves pre-treatment to trap, capture and remove the PAH load associated with coarse organic matter, whereas the second stage uses a conventional stormwater "treatment train" to remove the PAH load associated with the more fine-grained sediment particles.

Crabtree et al (2006) evaluated the impact of different BMPs to reduce PAH levels in highway runoff in the UK. Two wet ponds were found to be highly effective at removing PAH, with removal rates of 99% and 57%, respectively. Dry detention ponds were much less effective at remove PAH (22%), and sumps in the storm drain inlet showed no capability to remove PAH.

Sebastian et al (2014) investigated the effectiveness of a dry retention pond in removing PAH levels in an industrial catchment in France. They observed that the pond was more effective at removing PAH with higher molecular weight (HMW-4 to 5 rings), compared to PAHs with lower molecular weight (LMW-2 to 3 rings). Over ten storm events, the pond removed 24 to 67% of the HMW PAHs, but only 4 to 31% of the LMW PAHs. Overall, PAH removal rates were less than TSS removal rates during the entire study.

3.8 PAH Accumulation in BMP Sediments

Table A-9 shows the effectiveness of stormwater BMPs in trapping PAH compounds, as indicated by highly elevated PAH concentrations in pond sediments.

Four recent studies have shown that PAH compounds accumulate and persist in BMP sediments, at levels that exceed sediment guidelines, and which may trigger special sediment handling and disposal methods. The risk of sediment PAH contamination is most pronounced within older stormwater ponds, whose hypoxic bottom waters prevent rapid biodegradation of PAH compounds in the sediments.

Crane (2014) sampled PAH levels in the bottom sediments of 15 stormwater ponds located in the Minneapolis/St Paul area. Based on PAH fingerprinting, Crane (2014) found that coal tar based sealants comprised about 67% of the PAH found in the sediments, whereas vehicle-related emissions were the source of 30%, with the remainder caused by wood burning. The PAH levels in pond sediments were high (see Table A-9), and 20% of the pond sediments exceeded PAH guidelines for aquatic health, and 60% exceeded human health benchmarks.

Gallagher et al (2010) sampled PAH levels in the bottom sediments at 68 stormwater ponds in Baltimore County, MD. Overall, they found a median PAH concentration of 1,052 mg/kg in the stormwater pond sediments, and at least one PAH exceeded the threshold effects concentration (TEC) at 63% of the ponds sampled (see Table 7 for individual PAHs).

Table 7. Percent of MD Stormwater Ponds with Potential PAH Sediment Toxicity

Individual PAH	TEC	PEC
Napthalene	3%	0%
Flourene	12%	1%
Phenanthrene	46%	12%
Anthracene	15%	1%
Flouranthene	34%	13%
Pyrene	34%	15%
Benzo[a]anthracene	24%	7%
Chrysene	34%	10%
Benzo[a]pyrene	38%	7%
Dibenz[a,h]anthracene	44%	NA

Source: Gallagher et al, 2010

Weinstein et al (2010) investigated PAH levels in the bottom sediments of 16 stormwater ponds in coastal South Carolina. The highest PAH levels were recorded in ponds draining commercial areas, whereas PAH levels were much lower at ponds draining residential, golf course or non-urban catchments. The PAH levels in commercial pond were more than 24,000 ng/g, and 42 to 75% of the pond sediments were considered toxic to aquatic organisms.

Kamlakkannan et al (2004) investigated the fate of PAHs that were captured and trapped within wet stormwater ponds. They observed that wet ponds were very effective at trapping PAHs, and reported sediment PAH levels in the range of 38 to 65 mg/kg. The study team noted that ponds were not effective in breaking down PAH within the sediments, primarily because the bottom waters lacked sufficient oxygen.

3.9 Other PAH Management Strategies

The largest and most controllable source of PAH are coal tar sealcoats applied to extend the life of asphalt parking lots. Numerous studies have documented that the sealcoats generate a very high PAH load, and several state and local governments in the Chesapeake Bay have banned their use. Imposing a Bay-wide coal tar sealcoat ban would not only be an effective strategy to reduce PAH inputs to the estuary, but would also minimize the risk of PAH sediment contamination in upland stormwater ponds.

Some indication of the potential value of a Bay-wide coal tar sealcoat ban was provided by Pavelowsky (2012), who developed a regression model to estimate the impact of a ban on future PAH levels in Springfield, MO. He forecast that PAH levels would drop by 80 to 90% after a coal tar sealcoat ban, but estimated that it would take up to 20 years to fully occur.

Section 4: Petroleum Hydrocarbons

4.1 Key Findings for Petroleum Hydrocarbons

- TPH refers to a broader group of petroleum hydrocarbons than PAH. Unlike PAH, there are no numerical aquatic life or human health standards that applies to this class of toxins. Instead, most states usually establish a narrative standard (e.g. no visible sheen) or a maximum concentration of a surrogate parameter (such as oil and grease) in order to regulate fuel spills and other discharges of oil, gas or other hydrocarbons into receiving waters.

- TPH is not as well studied as some other toxins, such as PAH. Overall, the quality of monitoring data to assess TPH sources, pathways and loads in the watershed was classified as low to moderate, with some major data gaps in our understanding.

- The limited TPH data that does exist suggests that it meets all six criteria to be classified as an urban toxic contaminant, and that it can be effectively treated by most urban BMPs that are capable of removing sediment particles in urban stormwater runoff.

- A handful of monitoring studies confirm that the TPH are effectively removed by stormwater BMPs (or surrogate hydrocarbon parameters, such as oil and grease or benzene). TPH removal rates appear to be equal to or greater than total suspended solids rates.

- In addition, recent studies have shown that bioretention and rain gardens are not only effective in trapping TPH, but also in breaking it down via microbial processes in the aerobic soil environment of the media. The reported bioremediation that occurs within bioretention areas is encouraging, as it greatly reduces the potential for TPH accumulation over time (unlike PAH).

- While urban BMPs are effective in removing hydrocarbons, it is important to maintain existing pollution prevention practices to prevent and/or contain spills, leaks and other fuel discharges to the environment.

4.2 Background on Petroleum Hydrocarbons

Petroleum hydrocarbons refers to numerous compounds that are derived from crude oil. There is considerable variation in the chemical properties among this group of compounds, which has a strong influence on their transport and delivery, bioavailability and toxicity. The lower molecular weight hydrocarbons can be partly soluble in water, and may include benzene, toluene, ethylbenzene and xylene. Higher molecular weight hydrocarbons, on the other hand, tend to be less mobile, and more associated with sediment particles or organic matter.

4.3 Petroleum Hydrocarbons: Environmental Risk and Trends

While petroleum hydrocarbons include PAHs, there is no specific aquatic or human health standard that applies to them (unlike PAHs). Instead, states define a narrative standard (e.g., no visible sheen), or define a maximum concentration for a surrogate parameter (e.g., oil and grease).

Consequently, states have reported only limited and localized impairments for petroleum hydrocarbons in surface waters within the Bay watershed, and have not issued any fish consumption advisories for hydrocarbons. Many ecological concerns still exist about the impact of high levels of petroleum hydrocarbons on fish and aquatic life, but PAH tends to be the specific form of hydrocarbons that attracts the most regulation.

4.4 Sources and Pathways of Petroleum Hydrocarbons

The sources of petroleum hydrocarbons are mainly associated with "car habitat" in urban watersheds. Urban land uses with a high degree of traffic have greater potential to produce TPH, such as freeways, urban streets, commercial parking lots and residential streets. The specific pathways include fuel leaks and spills, vehicle emissions and even tire particles.

4.5 Characteristics of Petroleum Hydrocarbons

Petroleum hydrocarbons have a strong affinity for both fine and coarse-grained sediment particles, although minor amounts can sometimes be found in the liquid phase, depending on the molecular weight of the different hydrocarbon compounds. In addition, some hydrocarbons can easily volatilize.

4.6 TPH Concentrations in Urban Runoff and Sediments

Table A-12 summarizes the available data on petroleum hydrocarbon concentrations in urban sediments, whereas Table A-13 compares the measured concentrations of petroleum hydrocarbons in urban stormwater runoff.

James et al (2010) monitored several urban catchments in Tennessee, and found that runoff from impervious cover produced the highest levels of oil/grease (mean 250 mg/l) and TPH (62 mg/l), with much lower concentrations reported from pervious areas. James et al (2010) also found that flouranthene and pyrene were ubiquitous in runoff from impervious areas, as well as the sediments of the creeks and rivers that received the runoff.

4.7 Measured Removal of Petroleum Hydrocarbons in Urban BMPs

Several monitoring studies confirm that TPH (or surrogate hydrocarbon parameters, such as oil and grease or benzene) are effectively removed by stormwater BMPs, often at rates that range from 80 to 90%. These removal rates generally exceed the benchmark removal rate for total suspended solids. The high removal is due to the multiple hydrocarbon removal mechanisms within most BMPs -- not just settling and filtering, but also adsorption, biodegradation and volatilization. Some of the individual studies that monitored hydrocarbon removal by urban BMPs are profiled below.

Hsieh and Davis (2006) found that a bioretention test column removed 99% of the oil and grease it received during 12 synthetic runoff events in a three month period. An Australian study also reported 80 to 90% removal of TPH within a bioretention area (Zhang et al, 2013).

Newman et al (2013, 2014) examined how well an enhanced permeable paving parking lot with underground detention could absorb a fixed quantity of lubricating oil and diesel oil applied to the surface. Despite this simulated spill, the BMP was able to prevent TPH discharges from the facility from exceeding 0.1 mg/l over a five month period.

In a Scottish study, Tang et al (2009) looked at the capability of vertical flow constructed wetlands to reduce benzene. Overall, they reported benzene removal ranging from 73% to 90%, depending on test conditions. The predominant removal mechanisms in the constructed wetland was aerobic biodegradation and volatilization. The wetland plants, by themselves, did not play a significant role in the overall benzene removal.

Roinas et al (2014) monitored TPH levels in stormwater runoff as it passed through a series of swales and ponds along a UK motorway. They found that TPH was attached to particulate matter, and especially organic matter, but the monitoring of TPH through the system was problematic.

Hong et al (2006) conducted a series of bench scale column tests to evaluate the effect of bioretention in reducing levels of oil and grease in stormwater runoff. Oil and grease removal rates of 80 to 95% were observed, with most of the removal due to sorption and filtration, much of which occurred on the surface mulch layer. In addition, Hong et al (2006) observed that 90% of the hydrocarbons trapped in the bioretention area were effectively bio-degraded within several days after each simulated runoff event.

4.8 Petroleum Hydrocarbon Accumulation in BMP Sediments

LeFevre et al (2012) collected 75 sediment samples from 58 rain gardens and 4 upland control sites in Minneapolis, MN. The samples were monitored for TPH concentration, as well as the presence of genes from micro-organisms that break down petroleum hydrocarbons.

LeFevre et al (2012) measured low levels of TPH from all rain garden soils (maximum of 3 ug/kg), but these were still higher than the upland control soils (but several orders of magnitude below benchmark levels for sediment quality). Microorganisms that break down TPH were found at all rain garden sites, especially those that were planted with deep-rooted prairie plant species (as opposed to just mulch or turf cover). LeFevre et al

(2012) concluded that rain gardens and bioretention were an ideal practice to both remove and break down urban hydrocarbons.

4.9 Other Petroleum Hydrocarbon Management Strategies

While urban BMPs are effective in removing hydrocarbons, it is important to maintain existing pollution prevention practices that prevent and/or contain spills, leaks, other fuel discharges to the environment. The stormwater benchmarking tool can help identify effective pollution prevention practices for industrial and municipal sites (CSN, 2009).

Section 5: Mercury (Hg)

5.1 Key Findings for Mercury

- Mercury accumulation in fish tissue is a major cause of widespread water quality impairment in rivers, impoundments and estuaries across the Chesapeake Bay watershed.

- Overall, the quality of monitoring data to evaluate mercury sources, pathways and loads in the watershed is considered high, although there is much less monitoring data available on mercury removal by stormwater BMPs or its presence in BMP sediments.

- Although mercury is a global pollutant that is deposited across the entire watershed and over the open waters of the Bay, it still meets the six criteria to qualify as an urban toxic contaminant.

- Mercury loading rates are highest in urban watersheds, due to the wash-off of mercury deposited on impervious surfaces into the storm drain network.

- Although mercury exists in several forms, it is strongly associated with sediment particles and primarily moves through the watershed during high urban stormwater flows.

- The encouraging trend over time is toward lower levels of mercury in lake and estuarine sediments, and lower levels within the Chesapeake Bay bald eagle population.

- Despite these positive trends, mercury levels will continue to be a problem for many decades, given the considerable lag time between when mercury is deposited on watershed soils, and when the contaminated soils move through the stream network in the watershed to reach the Chesapeake Bay.

- Further complicating the issue is the methylation process. Under certain environmental conditions, mercury is transformed in methyl-mercury, which rapidly accumulates in fish tissue, and magnifies up the food chain to cause toxicity to fish, birds, mammals and humans.

- The treatability of mercury inputs is not as great as other UTCs for several reasons. The first is that significant mercury inputs bypass the stream network and are directly deposited on the open waters of the Bay. The second relates to the methylation process that is enhanced in anoxic and organic-rich sediments of natural wetlands and estuaries. Some researchers estimate that more than half the methyl-mercury is produced within the open waters and wetlands of the Chesapeake Bay, which sharply limits any impact from upland stormwater treatment.

- Given the amount of water quality impairment that mercury causes, it is surprising how few monitoring studies have been undertaken to determine if urban BMPs can effectively remove mercury. Based on the limited data available, it appears that mercury does behave very much like a sediment particle, and should be removed by urban BMPs that can effectively settle out or filter sediment particles.

- One monitoring study showed that constructed stormwater wetlands were very effective at removing mercury from urban runoff, and that mercury was retained in the bottom sediments. The researchers cautioned that that the hypoxic and organic rich conditions that occurred within the constructed wetland also increased the rate of methyl-mercury conversion to that observed in natural wetlands.

- Two pollution prevention practices could also help reduce mercury loads -- recycling of thermostats and fluorescent bulbs. In addition, targeted street cleaning efforts may also have a moderate ability to reduce mercury levels contained in street dirt.

5.2 Background on Mercury

Mercury is truly a global pollutant, as it is generated by power generation facilities and deposited across watersheds of all kinds. While urban watersheds have higher mercury loading rates, mercury bio-accumulation problems are encountered in all watersheds, including undeveloped ones.

Under certain environmental conditions, mercury can be transformed in methyl-mercury, which rapidly accumulates in fish tissue, and magnifies up the food chain to cause toxicity to fish, birds, mammals and humans (Wentz et al, 2014). The conversion occurs in a process known as methylation which occurs in anoxic and organic rich sediments in natural wetlands and estuaries.

5.3 Mercury: Environmental Risk and Trends

Mercury accumulation in fish tissue is a significant cause of water quality impairment in the Chesapeake Bay watershed, with more than 600 river miles and 20,000 acres of impoundments listed (CBP, 2012). As many as ten fish and shellfish species are subject to fish consumption advisories, and they are widely distributed across the tidal and non-tidal portions of the Chesapeake Bay watershed.

Nationally, mercury is responsible for more river miles and lake acres being under fish consumption advisories than all other contaminants combined (Wentz et 2014). Mercury is also the leading cause of water quality impairment nationally, and is the subject of more than 20,000 TMDLs (see Table 8).

Table 8. Toxics and TMDLs in the United States

Rank	Pollutant	# of TMDLs in US
1	Mercury	21,545
2	Pathogens	13,016
3	Metals (excluding Hg)	9,828
4	Nutrients	6,034
5	Sediment	3,922
11	Pesticides	1,233
13	PCBs	698
17	PAH and Toxic Organics	158

Source: EPA OWOW Website, Accessed July 2015

Despite the relatively high loadings, the impact of mercury on avifauna in the Chesapeake Bay appears to be low. For example, Cristol et al (2014) discovered low mercury in the molted feathers of a large sample of bald eagles in the Chesapeake Bay watershed. The mercury levels in the population of Chesapeake Bay eagles was the lowest observed in North America.

Velinski et al (2011) has observed a gradual decline in mercury levels in estuarine sediments in the Anacostia River, and concluded they may return to pre-industrial levels within a few decades. Peak mercury levels were reported for sediment layers that were dated to the 1950's and early 1970's.

Lawson and Mason (2001) noted the long lag time between when mercury is deposited from the atmosphere onto watershed soils, and it slowly moves through the stream network to reach the Chesapeake Bay -- suggesting that even mercury inputs were to cease today, it would take multiple decades for those inputs to cycle through the watershed and reach the Bay. Mason et al (1999) estimated that only 12% of the mercury deposited from the atmosphere to the watershed ends up reaching the Chesapeake Bay estuary.

Analysis of lake sediment core data has shown a decreasing trend in mercury deposition in recent decades, particularly in highly urban areas (Wentz et al, 2014). This is reinforced by a declining trend in the concentration of mercury in rainfall (NADP, 2013).

Mason et al (2006) note that methyl-mercury accumulation in fish tissue tends to be lower in urban estuaries and non-urban estuaries, compared to fish in reservoir and impoundments in the Chesapeake Bay watershed.

5.4 Sources and Pathways of Mercury

Numerous researchers have documented that atmospheric deposition is the predominant mercury loading source at the watershed level, and that the highest rates of deposition occur in urban watersheds (Van Metre and Mahler, 2003, Van Metre, 2012, Mangarella et al 2012, Wentz, 2014).

5.5 Characteristics of Mercury

Mercury (Hg) occurs in the environment as a variety of organic and inorganic compounds, in solid and/or dissolved state, as well as in liquid and gas phases. Despite this variability, mercury behaves much like a sediment particle as it is transported through the watershed.

Monitoring studies have shown that mercury levels in storm flow are strongly correlated with turbidity (Gilbreath et al, 2012) and suspended particulate matter and particulate organic matter (Mason et al, 1999). David et al (2009) reported a strong correlation between mercury concentrations and suspended sediment in urban and agricultural rivers in California.

In a study of 39 Wisconsin rivers, Hurley et al (1995) found the highest rates of mercury methylation in watersheds with a high percentage of wetland surface area. Methyl-mercury levels, however, were not correlated with turbidity or storm flow (Gilbreath et al 2012), and actually tended to comprise a higher fraction of dry weather flows (Mason et al, 1999).

Surprisingly, methyl-mercury concentrations are actually higher for predator fish in un-developed watersheds that are dominated by forest and/or wetland cover than for urbanized watersheds -- which have much higher mercury inputs (Wentz et al, 2014). It is speculated that urban watersheds had fewer wetlands, simpler food webs and much lower aquatic diversity -- and consequently fewer predatory fish to accumulate mercury.

5.6 Mercury Concentrations in Urban Runoff and Sediments

The measured concentrations of mercury in urban stormwater runoff are compared in Table A-14, whereas the available data on mercury concentrations in urban sediments is summarized in Table A-15.

Gilbreath et al (2012) documented that mercury loads were high in storm flows for urban watersheds compared to rural ones in California. In the same region, Mangarella et al (2012) established that the highest unit area mercury loads in runoff were produced from industrial and commercial land uses, as compared to residential and open space. In a national review, Wentz et al (2014) concluded that urban watersheds had the highest mercury concentrations, compared to all other non-urban land uses.

Mason et al (1999) reported higher mercury levels in estuarine sediments in urban areas of the Chesapeake Bay, compared to rural ones. Mason et al (1999) also reported that urban watersheds produced the highest mercury inputs in the Bay watershed, presumably due to the wash-off of mercury deposited on their impervious surfaces.

Lawson et al (2001) reported that the highly urban Anacostia River had a very high mercury load, also thought to be due to wash-off of mercury from impervious surfaces. They estimated that nearly 85% of the mercury inputs from atmospheric deposition in the watershed reached the estuary. Other urban areas with high mercury inputs and sediment enrichment include the Baltimore Harbor (Mason and Lawrence, 1999) and the Elizabeth River (CBP, 2013).

5.7 Estimated Removal of Mercury by Urban BMPs

Given the amount of water quality impairment it causes, it is remarkable how little monitoring has been performed to determine if urban BMPs can remove mercury from stormwater runoff.

Yee and Mckee (2010) conducted a series of settling column experiments using stormwater runoff and sediment samples collected from urban watersheds in the San Francisco Bay area. They found that 10 to 30% of mercury entrained in stormwater settled out within 20 minutes, and 90% of mercury re-suspended from creek sediments settled out within 10 minutes. Based on these experiments, Yee and McKee concluded that mercury behaved very much like a sediment particle, and that any urban BMP that promoted settling of fine sediment particles or captured fine-grained street solids (e.g., street cleaning) should be effective at reducing mercury loads in urban watersheds.

Monson (2007) monitored the effect of 10 constructed wetlands in Minnesota to remove mercury in urban stormwater runoff and found that they were extremely effective in trapping and retaining mercury inputs (e.g., 80 to 90% removal, primarily due to particle sedimentation).

Mason et al (1999) noted that there are limits to extent of BMP treatment for mercury in the Chesapeake Bay watershed, for the simple reasons that about half of the total mercury load in the watershed is directly deposited to the open waters of the Bay. In addition, Mason et al (2006) also note that historical mercury inputs to Chesapeake Bay wetlands and estuarine sediment are prone to methylation -- as much as 60% of the methyl-mercury which bio-accumulates in the estuarine food chain is produced in-situ within the Bay.

5.8 Mercury Accumulation and Methylation in BMP Sediments

Monson (2007) also measured the amount of methylation that occurred in ten constructed wetlands in Minnesota, and found they produced methyl-mercury which bio-accumulates in organisms. About 1.4% of the mercury trapped within the constructed wetlands was converted to methyl-mercury, primarily during periods of wetland anoxia and sediment phosphorus release.

Wentz et al (2014) observed that the methylation process is exacerbated in recently flooded terrestrial soils, and in wetland sediments subject to repeated wetting and drying cycles, both of which are common in constructed wetlands in agricultural and urban settings.

5.9 Other Mercury Management Strategies

Mangarella et al (2012) performed an extensive analysis of mercury reduction strategies for urban watersheds in the San Francisco Bay area. They concluded that two pollution prevention practices -- recycling of thermostats and fluorescent bulbs -- could help reduce urban mercury loads.

Mangarella et al (2012) also determined that stormwater retrofits and street cleaning efforts targeted at commercial, industrial and redevelopment sites also showed a moderate capability to reduce urban mercury loads.

Section 6: Urban Trace Metals (Cd, Cu, Pb and Zn)

6.1 Key Findings for Urban Trace Metals

- Four trace metals, cadmium, copper, lead and zinc, are detected in virtually every sample of urban stormwater runoff, and are measured at concentrations that are consistently higher than any other watershed land use. Consequently, they are referred to as "urban trace metals" or UTMs.

- The data quality for the four UTMs was rated as moderate to very high and ranked the highest of any of the UTCs reviewed in this study. In particular, more than 50 studies are available that evaluate how urban BMPs remove urban trace metals from stormwater.

- UTMs qualify as an urban toxic contaminant as they meet at least five of the qualifying criteria. They have unique urban sources including roofing materials, brake pads, tires, vehicle emissions and atmospheric deposition.

- The only criteria that UTMs do not fully meet is behaving like a sediment particle. Depending on the metal, as much as 10 to 60% of UTMs are found in soluble form which also exerts the greatest toxic impact to aquatic life.

- In terms of environmental impact, the concentrations of soluble Cd, Cu and Zn exceed acute toxicity standards for aquatic life in about 50% of urban stormwater runoff samples collected across the nation.

- Lead levels in urban runoff have declined sharply in the last three decades, due to the introduction of unleaded gasoline. Consequently, lead levels in runoff no longer exceed aquatic life or human health standards. No long term trend data are available for cadmium, copper or zinc.

- UTMs are highly treatable and their BMP removal rates tend to be slightly lower than total suspended solids. Individual trace metal removal rates range from moderate to very high, depending on the type of stormwater practice employed. In general, the highest overall UTM removal rates were reported for bioretention, wet ponds and sand filters.

- Several studies have looked at UTM accumulation in BMP sediments or media, and the potential for breakout and release of soluble metals over time. The studies have generally found that metal binding sites are finite in number, but several decades would be needed to fully exhaust them. Periodic removal of the top few inches of sediment or media should prevent any soluble metal loss over time.

- While stormwater BMPs are an effective strategy to reduce urban trace metals to receiving waters, they need to be augmented by other management strategies to comprehensively reduce trace metal loads. These include stormwater benchmarking and pollution prevention at industrial sites, as well as product substitution to reduce metals delivered from brake pads, rotors, tires and roofing material.

6.2 Background on Urban Trace Metals

Cadmium, copper, lead and zinc are considered urban trace metals since they are detected in virtually every urban stormwater sample and are measured at concentrations that are greater than any other watershed land use. The toxicity and transport of trace metals depends on their bioavailability, which is influenced by variations in metal speciation, pH, redox potential, particle size distribution, organic matter and temperature.

6.3 Urban Trace Metals: Environmental Risks and Trends

LeFevre et al (2014) notes that the greatest toxicity risk is associated with dissolved forms of trace metals, which are more bioavailable to aquatic life. Based on a national review of stormwater data, LeFevre et al (2014) concluded that 50% of the dissolved cadmium and copper concentrations measured during storm events exceeded the ambient acute water quality criteria. They reported that 45% of the dissolved zinc samples, and 18% of the dissolved lead samples also exceeded the acute criteria.

Lead levels in urban runoff have declined sharply in the last three decades, due to the introduction of unleaded gasoline. Consequently, lead levels in runoff seldom exceed aquatic life or human health standards anymore.

6.4 Sources and Pathways for Urban Trace Metals

Sabin et al (2005) measured the load of trace metals deposited from the atmosphere and compared it to the stormwater runoff load in Los Angeles, CA. They determined that atmospheric deposition comprised 74%, 108% and 57% of the copper, lead and zinc load measured in stormwater runoff, respectively. Sabin et al (2005) also noted that the metals had a much higher urban signature, compared to other non-urban areas sampled elsewhere in the nation.

McKenzie et al (2009) provided solid evidence linking automotive products as a source of urban trace metals. Tires were found to a significant source of copper, lead, and zinc, whereas brake pads and rotors were a major source of cadmium, copper and possibly zinc.

Van Metre and Mahler (2003) examined the sources of trace metals in Austin, TX, and discovered that metal roofing was a source of both cadmium and zinc, whereas asphalt roof shingles were a source of lead. Overall, Van Metre and Mahler (2003) estimated that roofing generated about 20% of the total stormwater load of lead and zinc, as measured at the subwatershed scale.

Reddy et al (2013) noted that painted structures were a key source of lead in urban runoff, and that building siding and downspouts also contributed cadmium, copper, lead and zinc. LeFevre et al (2014) also reported that metal-coated roofs were a major source of dissolved copper, lead and zinc in urban runoff.

6.5 Urban Trace Metal Concentrations in Runoff and Sediments

The median concentration of urban trace metals in stormwater runoff and sediments are shown in Table 9.

Table 9. Comparison of Urban Trace Metals

Factor	**Cadmium**	**Copper**	**Lead**	**Zinc**
Runoff EMC[1] (ug/l)	1	16 ug/l	17 ug/l	115 ug/l
Solubility[2] (%)	45%	60%	10%	50%
Acute Toxicity[3] (%)	50%	50%	18%	45%
Sediment Level[4] (ug/g)	0.2 - 0.5	40 - 150	20 - 200	200 - 500
Removal Rates[5] (%)	40 - 70%	40 - 60	50 - 90	55 - 75
Sediment Risk[6]	Low	Moderate	Low	Moderate

[1] Median value from National Stormwater Quality Dataset. More detailed information on runoff concentrations for Cd, Cu, Pb and Zn can be found in Tables A-18, A-20, A-25 and A-29, respectively.

[2] Median inflow concentrations from ISBD (2014)

[3] From LeFevre et al, 2014

[4] More detailed information on sediment concentrations for Cd, Cu, Pb and Zn can be found in Tables A-19, A-21, A-26 and A-30, respectively.

[5] More detailed information on BMP removal rates for Cd, Cu, Pb and Zn can be found in Tables B-2, B-5, B-6 and B-8, respectively.

[6] Sediment risk refers to either the possibility of metal accumulation exceeding a TEC or potential breakout or release of the metal

Tiefenthaler et al (2008) conducted extensive sampling of trace metal EMCs during storm events for urban and non-urban land uses in Southern California. The highest copper, lead and zinc concentrations were recorded in industrial catchments with more than 70% impervious cover. Most of the metals exhibited a "first flush" phenomena with higher concentrations occurring at the beginning of storm events (although this is not uncommon for runoff occurring in semi-arid climates). Tiefenthaler et al (2008) consistently found that trace metal EMCs from developed catchments were one to two orders of magnitude higher than non-urban catchments.

The SSDC EP (2015) examined the trace metal content in street solids, based on a national data review, and the results are presented in Table 10. In general, street solids had roughly twice the trace metal content of urban soils, which are also enriched due to past metal deposition of metals from dry and wetfall.

Lau and Stenstrom (2005) measured trace metal levels and particle sizes for street solids in Los Angeles, CA. They found that particles in the 100 to 250 micron size range comprised the greatest fraction of the total metal load for street solids, which suggested that advanced street cleaning and BMPs that emphasize settling and or filtering would promote high levels of trace metal removal.

Table 10. Comparison of Trace Metal Content of Street Sweeper Waste and Urban Soils (mg/kg)				
Study	STATE	**Copper**	**Lead**	**Zinc**
Sorenson, 2013	MA	72	62	146
Sorenson, 2013	MA	47	111	169
SPU, 2009	WA	49	103	189
CSD, 2011a	CA	92	23	136
CSD, 2011b	CA	157	204	210
Walch, 2006	DE	64	81	208
MEAN		**80**	**97**	**176**
Urban Soils (MD, Pouyat et al, 2007)		35	89	91
Source: Street and Storm Drain Cleaning Expert Panel Report (SSDC EPR, 2015, including reference in table).				

6.6 Measured Removal of Urban Trace Metals Urban BMPs

Several recent studies reinforce the notion that urban BMPs are generally effective at removing trace metals from stormwater runoff. Table 11 summarizes the capability of different stormwater BMPs to remove urban trace metals.

Table 11. Comparative Ability of Stormwater BMPs to Remove Selected Trace Metals				
Stormwater BMP	Urban Trace Metals			
	Cadmium	Copper	Lead	Zinc
Bioretention	H	VH	VH	VH
Wet Pond	M	H	H	H
Wetland	M	H	M	M
Sand Filter	H	M	VH	H
Permeable Pavement	L	M	VH	VH
Dry Swale	L	H	--	VH
Grass Channel	M	L	L	M
Grass Filter	L	M	L	M
Dry Pond	L	L	M	M
VH: Very High Removal (76% to 100%) H: High Removal (50% to 75%)	M: Moderate Removal (26% to 50%) L: Low Removal (0% to 25%)			
Sources: Appendix B with an emphasis on Leisenring (2014) and Winer (2000)				

Bioretention appears to be the most effective stormwater practice to removal all four urban trace metals. This finding is strongly reinforced by a recent review of 12 field and laboratory studies that evaluated how well bioretention areas removed the four trace metals (see Table 12).

Table 12.
Summary of Trace Metal Removal in 12 Bioretention Studies

Trace Metal	Research Studies (N)	Removal Rate (%)
Cadmium	2	66-90
Copper	10	43-98
Lead	12	75-98
Zinc	11	62-99

Source: LeFevre et al (2014)

The new studies shed light on the processes that maximize urban trace metal removal in bioretention areas. For example, Li and Davis (2008) examined the fate and movement of trace metals within a test bioretention column. They found that most of the trace metals are captured on the surface mulch layer or the top few inches of the bioretention media. Li and Davis (2008) concluded 12 to 18 inches of bioretention media were sufficient to maximize trace metal removal. Jang et al (2005) also reported that trace metals tended to sorb onto the shredded hardwood mulch layer, which is a common component of rain gardens and bioretention areas. Jang et al (2005) found the greatest mulch sorption for lead, followed by copper and then zinc.

Hunt et al (2012) provides a good synthesis of how to maximize trace metal removal in bioretention areas. They concluded that even shallow media depths can produce high rates of trace metal removal, but these rates could be enhanced if more organic matter were added to the bioretention media recipe to increase metal binding sites. They also noted that prior research had not shown a strong phytoremediation or uptake effect by the plants within the bioretention area. LeFevre et al (2014) also observed that bioretention plants were not very effective at taking up trace metals, although they also noted that plant species were not selected based on their phytoremediation capability.

Design and Media Enhancements for Bioretention

Several recent studies investigated various design and media enhancements to improve trace metal removal in bioretention areas and sand filters. For example, Reddy et al (2013) evaluated the impact of adding calcite, zeolite and/or iron filings to a sand filter media, and found that they sharply increased cadmium, copper, lead and zinc removal rates, compared to the conventional sand media (which is used in both sand filters and bioretention areas). Reddy et al (2014) also looked at the potential of biochar to enhance metal removal in settling column tests. With the exception of copper, however, the addition of biochar to a sand filter did not greatly enhance removal of other trace metals.

Blecken et al (2009) conducted an experiment in a bioretention mesocosm to determine if adding organic carbon and a submerged gravel zone to the bottom of the filter could enhance the removal of trace metals. Adding carbon to an anoxic submerged zone has been shown to increase denitrification and overall nitrogen removal in other bioretention areas (Hunt et al, 2012). Blecken et al (2009) reported up to 95% removal of copper, lead and zinc within the bioretention mesocosm, and noted that it did not appear to negatively interfere with the denitrification process.

6.7 Urban Trace Metal Retention in BMP Sediments

Crawford et al (2010) investigated trace metal levels in bottom sediments of 18 stormwater wet ponds located in coastal South Carolina. They found that wet ponds draining commercial land use had elevated levels of lead and zinc in bottom sediments, compared to ponds that drained other land uses. Cadmium, copper and sometimes zinc exceeded sediment screening criteria to protect aquatic life, but were consistently below human health benchmarks. Lead levels were generally below the sediment guidelines for both aquatic life and human health.

Casey et al (2006) investigated copper, lead and zinc levels in the bottom sediments of 20 stormwater ponds located in central MD over a ten year period. Casey et al (2006) reported that metal levels in pond sediments did not increase over the ten year period, with the exception of zinc levels in several ponds receiving highway runoff. Overall, the levels of copper and lead fell below the sediment threshold effects concentration (TEC) limit, although 30% of the zinc sediment samples did exceed the TEC. Casey et al (2006) concluded that sediment and invertebrate trace metals levels were at a steady state in stormwater ponds, and the risk of metal exposure to pond organisms did not vary as a function of pond age.

Gallagher et al (2011) measured the trace metal content of bottom sediments of 68 stormwater ponds in Baltimore County, and found that 96% of them exceeded the threshold effects concentration (TEC) for at least one trace metal. Copper exceeded the TEC for 78% of the pond sediment samples, followed by zinc (60%), lead (18%) and cadmium (3%).

Li and Davis (2008) noted that there was a small risk for lead accumulation in bioretention areas, but observed that the lead is very tightly bound to sediment particles, and thus unlikely to create much risk for human exposure. They also noted that copper removal might be limited in bioretention areas under certain conditions that may cause copper release and breakthrough. Lefevre et al (2014) also noted that dissolved copper can sorb to organic matter, but under certain conditions may leach out with dissolved organic matter.

Jones and Davis (2013) reported on a field monitoring study of a four year old bioretention area in Maryland. They reported that cadmium, copper, lead and zinc all accumulated near the surface of the bioretention cell (i.e., within the top 5 inches), but none of the metals exceeded regulatory sediment thresholds. Jones and Davis (2013) concluded that the greatest trace metal accumulation would occur in the surface media in closest proximity to the stormwater inlet, and recommended that these surface sediments be removed every 10 to 15 years, and disposed properly.

LeFevre et al (2014) estimated the approximate number of years that it would take a bioretention area to utilize all of the metal binding sites within the media, under typical stormwater inflows. Their best estimate was 90 years for cadmium, 21 years for copper and 36 years for zinc, all of which are greater than the typical design life for a bioretention area.

Hatt et al (2011) conducted lab experiments to ascertain whether trace metals would eventually breakout from the filter medium of bioretention cells in the most bioavailable (and toxic) dissolved form. The Australian study subjected to filter columns to accelerated dosing of metals during simulated runoff events. Based on the results, Hatt et al (2011) concluded that there was some risk of dissolved zinc breakout within ten years, but cadmium, copper and lead showed very little breakout potential.

Consequently, Hatt et al (2011) recommended that surface layers in bioretention areas should be replaced every 10 to 15 years to prevent the risk of metals breakout. They also noted that adding more organic matter or compost to the media could increase metal retention within bioretention areas, albeit at the risk of causing greater nutrient leaching when the organic matter decomposes.

6.8 Other Urban Trace Metal Management Strategies

While stormwater BMPs are an effective strategy to reduce urban trace metals to receiving waters, they need to be augmented by other management strategies to comprehensively reduce trace metal loads. These include stormwater benchmarking and pollution prevention at industrial sites, as well as product substitution to reduce metals delivered from brake pads, rotors, tires and roofing surfaces.

Section 7: Other Trace Metals (As/Cr/Fe/Ni)

7.1 Key Findings for Other Trace Metals

- The quality of monitoring data to assess the sources and pathways of arsenic, chromium, iron and nickel was rated as moderate to high, although BMP removal data was somewhat limited. Most of the monitoring data has occurred outside the Chesapeake Bay, and much of our understanding about this group of metals has come from the urban watersheds of San Francisco Bay.

- Arsenic, chromium, iron and nickel are all frequently detected at high levels in urban sediments, stormwater runoff and during high river flow conditions in the Chesapeake Bay watershed.

- The main environmental risk associated with this group of trace metals is potential drinking water contamination, although the metal concentrations during most storm events fall well below most primary and secondary drinking water standards. Violations of acute freshwater toxicity standards are also generally uncommon. There is insufficient trend data to determine if the concentrations of the four metals are increasing, decreasing or remaining the same.

- Although these metals can be naturally produced through geological weathering and soil erosion, their concentrations tend to be much higher in urban watersheds, especially those with extensive industrial operations. The metals are exposed on many surfaces in the urban landscape where they "weather" or corrode in response to acid rain, and become entrained in stormwater runoff.

- All four of the trace metals --arsenic, chromium, iron and nickel--meet the six criteria to qualify as an urban toxic contaminant. Higher concentrations are found in urban watersheds, due to unique urban sources and emissions. They are primarily delivered in the watershed by urban stormwater.

- Higher concentrations of all four metals are strongly correlated with high flow, sediment and/or turbidity levels. The four metals are also strongly associated with sediment and organic matter, and behave like a sediment particle when it comes to stormwater treatment.

- Given their upland position, the four metals are treatable with stormwater BMPs, and there is abundant evidence that most BMPs are moderately effective in trapping the metals and retaining them in their sediment.

- The four trace metals are highly treatable with new or existing stormwater practices in urban watersheds. The highest removal rates (50 to 80%) are reported for iron, which is not surprising given its very limited solubility. By contrast, BMP removal rates for arsenic, chromium and nickel are more modest, ranging from 15 to 65%.

- The type of stormwater practice also has a strong influence on metal removal, with wet ponds, infiltration, sand filters and grass channels recording the highest overall removal rates.

- There was insufficient data to assess the risk that any of the four metals might breakout or be otherwise released from BMP sediments over time. Stronger evidence was found that trace metals can accumulate in the bottom sediments of stormwater ponds at levels that may sometimes exceed sediment toxicity guidelines.

7.2 Background on Other Trace Metals

The quality of monitoring data to assess the sources and pathways of arsenic, chromium, iron and nickel was rated as moderate to high, although the BMP removal data was somewhat limited. Most of the monitoring data has occurred outside the Chesapeake Bay, and much of our understanding about this group of metals has come from the urban watersheds of San Francisco Bay (Gilbreath et al, 2012).

All four of the trace metals --arsenic, chromium, iron and nickel--meet the six criteria to qualify as an urban toxic contaminant. Higher concentrations are found in urban watersheds, due to unique urban sources and emissions. They are primarily delivered by urban stormwater and behave much like sediment particles. Given their upland position, the four metals are treatable with stormwater BMPs, and there is abundant evidence that most BMPs are moderately effective in trapping the metals and retaining them in their sediment.

This group of trace metals is also frequently detected at high levels in urban sediments, stormwater runoff and during high river flow conditions in the Chesapeake Bay. While the highest concentrations of arsenic, chromium, iron and nickel are measured in urban watersheds during storm events, they can also be locally high in other non-urban watersheds (due to natural weathering of the metals from the geology and/or soils of certain physiographic regions of the Bay watershed).

7.3 Other Trace Metals: Environmental Risk and Trends

The main environmental risk associated with this group of trace metals is potential drinking water contamination, although the metal concentrations during most storm events fall well below most primary and secondary drinking water standards. Violations of acute freshwater toxicity standards are also generally uncommon. Recent research suggests that nickel and chromium levels in stormwater pond sediments may exceed sediment contamination guidelines.

There is insufficient trend data to determine if the concentrations of the four metals are increasing, decreasing or remaining the same.

7.4 Sources and Pathways of Other Trace Metals

Although some of these metals are naturally produced through geological weathering and soil erosion, their concentrations tend to be much higher in urban watersheds, especially those with extensive industrial operations. This due to the fact that these metals are exposed on many surfaces in the urban landscape where they can be "weathered" or corroded, often enhanced by the acid rain which falls on urban watersheds.

Some of the many different sources of these metals in the urban landscape are outlined in Table 13.

Table 13. Anthropogenic Sources of Other Trace Metals in the Urban Landscape

Trace Metal	Urban Sources
Arsenic	Wood preservatives, pesticide formulations, paints, dyes, semi-conductors and incinerator fly ash
Chromium	Stainless steel, chrome-plating, paint and some wood preservatives
Iron	Rust and corrosion of pipes, metal roofs and other iron surfaces
Nickel	Automotive batteries, household and industrial appliances, fabricated metals, fuel and lubricating oil

Source: Gilbreath et al (2012) and other sources

7.5 Concentrations of Other Trace Metals in Urban Runoff and Sediments

The typical concentrations of the four metals in urban stormwater runoff and urban sediments are compared in Table 14.

Table 14: Summary Comparison of Other Trace Metals

Factor	Arsenic	Chromium	Iron	Nickel
Runoff EMC[1] (ug/l)	3	7	700	3 - 8
Solubility[2] (%)	48	35	15	45
Sediment Level[3] (ug/g)	4	42	ND	37
Removal Rates[4] (%)	15 to 30	35 to 65	50 to 80	40 to 60
Sediment Risk[5]	?	Moderate	Moderate	Moderate

[1] Median value from National Stormwater Quality Dataset. More detailed information on runoff concentrations for As, Cr, Fe and Ni can be found in Tables A-17, A-23, and A-28, respectively.
[2] Median inflow concentrations from ISBD (2014)
[3] More detailed information on sediment concentrations for As, Cr, Fe and Ni can be found in Tables A-19, A-21, A-26 and A-30, respectively.
[4] More detailed information on BMP removal rates for As, Cr, Fe, and Ni can be found in Tables B-1, B-3, B-4 and B-7, respectively
[5] Sediment risk refers to either the possibility of metal accumulation exceeding a TEC or potential breakout or release of the metal
ND= No data

Higher concentrations of all four metals are strongly correlated with high flow, sediment and/or turbidity levels in urban streams and rivers. Most of the metals are strongly associated with sediment and organic matter, and behave like a sediment particle when it comes to stormwater treatment. It should be noted that the solubility of the four metals ranges from 15 to 48% (see Table 14), and it can vary over time due the influence of metal speciation, pH, redox potential, organic matter content and temperature.

7.6 Measured Removal of Other Trace Metals by Urban BMPs

The four trace metals are highly treatable with new or existing stormwater practices in urban watersheds. The highest removal rates (50 to 80%) are reported for iron, which is not surprising given its very limited solubility. On the other hand, removal of arsenic, chromium and nickel by stormwater BMPs ranges from 15 to 65% (See Table 15).

The comparative ability of different types of urban BMPs to remove the four metals is shown in Table 15. As can be seen, the type of stormwater practice has a strong influence on metal removal rates, with wet ponds, infiltration, sand filters and grass channels recording the highest removal rates. Surprisingly, bioretention areas, which were highly effective in removing Cd, Cu, Pb and Zn, were ineffective at removing nickel and iron, with several negative removal rates reported. On the other hand, bioretention was highly effective at removing chromium.

Table 15: Comparative Ability of Stormwater BMPs to Remove Selected Trace Metals

Stormwater BMP	Other Trace Metals			
	Arsenic	Chromium	Iron	Nickel
Bioretention	--	H	L	L
Wet Pond	M	H	H	H
Wetland	--	--	H	
Sand Filter	L	M	H	M
Permeable Pavement	--	L	--	H
Infiltration	--	H-	--	--
Grass Channel	M	M	L	H
Grass Filter	L	M	L	M
Dry Pond	L	M	--	L

VH: Very High Removal (76% to 100%)
H: High Removal (50% to 75%)
M: Moderate Removal (26% to 50%)
L: Low Removal (0% to 25%)

Sources: Appendix B with an emphasis on Leisenring (2014) and Winer (2000)

7.7 Retention of Other Trace Metals in BMP Sediments

There was not enough data to assess the risk that either arsenic, chromium, iron or nickel might breakout or be released from BMP sediments or media, although the fact several studies reported negative removal efficiency for some bioretention areas implies that the possibility does exist.

Stronger evidence exists for trace metal accumulation in the bottom sediments of stormwater ponds. Gallagher et al (2010) sampled the bottom sediments for trace metal levels at 68 stormwater ponds located in Baltimore County, MD. They found that nickel and chromium levels in stormwater pond sediments exceeded sediment contamination guidelines.

For nickel, the threshold effect concentration level was exceeded in 82% of the stormwater ponds, and the probable effects concentration was exceeded at 35% of the ponds. In the case of chromium, the threshold effect concentration level was exceeded in 49% of the ponds, whereas the probable effects concentration was exceeded at 4% of the ponds.

More research is needed to assess the risk of pond sediment contamination for these two metals, but it clearly shows the need to exercise care when handling and disposing of sediments during stormwater pond cleanouts.

Table 16. Evolution of Insecticides Over the Last 75 Years

To fully understand the next three sections of the memo, it is important to understand the evolution of insecticides over the last seven decades. Pesticides have been used to control insect problems in residential areas, crop lands (especially for specialty crops and orchards) and to manage mosquitoes since the 1940's.

Over this time span, the specific types of insecticides applied have changed frequently in response to both environmental concerns and market forces. Three broad trends in insecticide use are evident over the years. First, insecticide formulations have become much less persistent in the environment over time -- insecticide half lives that were once measured in years are now measured in months or even weeks. Second, the market has shifted away from formulations that can bio-accumulate in the tissues of vertebrates such as fish, birds and humans. Both of these trends are obviously a good thing.

The third and more disturbing trend is that insecticides are extremely lethal to target and non-target invertebrates at extremely low concentrations, and remain a major threat to aquatic life in streams, lakes and rivers.

Evolution in Use of Urban and Agricultural Insecticides Over Time

Era	Insecticide Category	Types	Notes
1940 to 1970	Organochlorines (OC)	DDT	Banned in the 1970s
		DDD/DDE	DDT degradation products
		Dieldrin	Banned in 1985
1960 to 2000	Organophosphates (OP)	Chlordane	Banned in 1978
		Chlorpyrifos	Restricted in 2002
		Diazinon	Restricted
		Dichlorvos	Increased use after 2002
2000 to present	Pyrethroids	Bifenthrin	Replacements for OCP and OPP
		Permethrin	Less toxic than bifenthrin
2005 to present	Fipronil [1]	Fipronil	Most aquatic life toxicity in recent surveys
	Neonictinoids [2]	Imdiacloprid	Emerging concerns about aquatic toxicity (Morrissey et al, 2015)

[1] Stone et al (2014) reported that fipronil had emerged as the urban insecticide exerting the greatest toxicity in urban streams in over the last decade. In a national assessment, fipronil violated the benchmark for aquatic life protection in more than 70% of urban streams sampled.

[2] Scientists have recently expressed major concerns about the toxic impact of neonictinoid pesticides that have recently entered the insect control market. This class of insecticides has been found to be extremely toxic to aquatic invertebrates and non-target species such as honeybees at extremely low (ng/l) concentrations (Morrissey et al, 2015).

Section 8: Pyrethroid Pesticides

(bifenthrin, permethrin and others)

8.1 Overall Findings

- Pyrethroid pesticides are a new class of insecticides that have entered the insect control market in the last decade. As a group, pyrethroids are relatively non-persistent in the environment and are unlikely to bio-accumulate in vertebrates. Nonetheless, pyrethroids are extremely lethal at very low concentrations to aquatic invertebrates in urban streams.

- Pyrethroids meet most of the basic criteria to qualify as an UTC, although there is no monitoring data to confirm whether they are trapped in BMP sediments and persist over time. In addition, the majority of the research has been conducted in unique climate and landscape conditions of California, which may limit its transferability to the Chesapeake Bay watershed.

- Pyrethroids have a strong affinity for sediment and organic matter and they are routinely detected in urban creek sediments where they exert their toxic effects. Consequently, pyrethroid removal rates in urban BMPs should be broadly comparable to those observed for suspended sediment, although more monitoring data is needed to confirm this. More research is also needed to assess the risk of that pyrethroids will persist and exert toxicity in pond sediments.

- Given how much aquatic toxicity has been linked to pyrethroids and other insecticides, it may be wise to expand public outreach and social marketing efforts to educate homeowners, landscape contractors, applicators and others about the proper methods and timing to apply insecticides.

8.2 Background and Characteristics of Pyrethroids

Pyrethroids are a group of insecticides used for structural pest control, landscape maintenance and home and garden use, and numerous formulations are sold at the retail level. To date, most of the concerns has been associated with improper homeowner applications to control insects, especially when granular forms are applied on or adjacent to impervious surfaces.

As a group, pyrethroids are highly toxic to fish and aquatic invertebrates in freshwater, estuarine and marine environments at extremely low concentrations. Among the pyrethroids, bifenthrin is both the most persistent and most toxic to aquatic life, whereas permethrin is the least toxic.

Pyrethroids are hydrophobic, preferentially adsorb to sediment particles, and are often found in urban stream sediments. Pyrethroids are strongly associated with sediment particles, especially clays, particles enriched with organic carbon or decomposing organic matter (Budd et al, 2011).

8.3 Sources and Pathways of Pyrethroids

Gilbreath et al (2012) monitored pyrethroids in runoff from urban watersheds in the San Francisco Bay region, and reported that stormwater delivered approximately 99% of the total pyrethroid load to the estuary. Bifenthrin and permethrin were detected in 100% of the stormwater samples. Sampling also revealed a strong correlation between higher pyrethroid concentrations and higher turbidity levels, as well as higher flow events.

Weston and Lydy (2002) measured bifenthrin levels in stormwater in urban creeks and the American river (CA). Urban stormwater was found to be the main source of bifenthrin, which was detected at levels ranging from 10 to 30 ng/l, which was well above the toxicity threshold for aquatic life. Extremely high bifenthrin concentrations were also detected downstream of large commercial nurseries in San Diego (Budd et al, 2007).

Hanzas et al (2010) investigated whether over-irrigation of residential lawns could be a pathway for pyrethroid loss in surface runoff. A test plot and sprinkler system were used to simulate three over-irrigation "events" and a winter storm event in California. Hanzas et al (2010) concluded that lawn runoff was not a major source of pyrethroid loss, as the maximum loss amounted to less than 1% of what had been applied.

In a companion study, Jiang et al (2012) investigated the significance of direct application of pyrethroids to concrete driveways in California, which is a common homeowner approach to control insects. Jiang et al (2012) found that bifenthrin and permethrin loss from runoff was very high one day after application, and that pyrethroid residues remained available for wash-off 3 to 7 months after they were initially applied to the driveways.

8.4 Pyrethroid Concentrations in Urban Runoff and Sediments

Lao et al (2010) monitored estuarine sediments for the presence of pyrethroid pesticides in Ballona Creek, which is a highly urban watershed in coastal CA. Lao et al (2010) found high concentrations of permethrin, bifenthrin and cypermethrin in the sediments that were toxic to an estuarine amphipod in bioassay tests. The highest pyrethroid concentrations were associated with fine-grained sediment particles that were enriched with organic matter.

Ensminger et al (2013) monitored pesticide levels in urban streams from residential watersheds in three metropolitan areas in California. They reported that at least one pesticide was detected in 90% of the stream samples. Bifenthrin was the most frequently detected pesticide (56 to 80%, depending on the metropolitan area).

Holmes et al (2008) sampled sediments in 30 urban creeks in California for the presence of pyrethroids, and found them in every sample. Bifenthrin was the pyrethroid found at the highest concentration and with the greatest toxicity. Bioassays confirmed that all 30 of the creek sediments were toxic to test organisms.

Ding et al (2010) investigated pesticide levels in creek sediments for urban and agricultural streams in Illinois. Many pyrethroids were detected in the urban streams, with bifenthrin found in 80% of the streams sampled. Pyrethroids were found to contribute the most to total toxicity in the urban stream sediments. By contrast, agricultural streams had much lower pyrethroid concentrations, and much less toxicity.

Kuvila et al (2012) conducted research on pyrethroid levels in urban stream sediments at seven metropolitan areas across the nation. The USGS research team detected one or more pyrethroids in the sediments in 45% of the 98 streams they sampled. Bifenthrin was the most frequently detected pyrethroid, followed by cyhalothrin and permethrin. The research team found regional differences among the metropolitan areas, which may have reflected differences in target insects (e.g., fire ants).

Overall, Kuvila et al (2012) reported somewhat lower concentrations of pyrethroids than earlier studies (most of which occurred in California) but many of the sediment samples were still toxic to the amphipod (*Hyella azteca*).

Amweg et al (2006) also measured pyrethroids in the sediments of 27 creeks located in both California and Tennessee. Pyrethroids were frequently detected in the CA streams and caused toxicity, most notably by bifenthrin. By contrast, pyrethroids were detected less frequently in Tennessee creek sediments, and produced much less toxicity. The reinforces the findings of Kuvial et al (2012) and Sprague and Nowell (2008) that there are major differences in pyrethroid loadings and impacts across the nation. In particular, the California pyrethroid data may not be transferable to the rest of the country, given its semi-arid climate, seasonal rainfall distribution, intensive landscape irrigation practices and unique insect control problems.

8.5 Estimated Removal by Urban BMPs

The monitoring data is rather sparse on whether BMPs can effectively remove pyrethroid pesticides, and consists of three studies that investigated constructed wetlands and swales that were treating agricultural runoff. For example, Moore et al (2009) investigated the capability of a constructed wetland to remove pyrethroid pesticides in agricultural runoff in Mississippi. They determined that the wetland did trap the pesticides effectively, with most of them sorbing either to wetland sediments or vegetation.

Budd et al (2011) monitored the impact of a constructed wetland in reducing pyrethroid pesticides generated from agricultural irrigation return flows in the Central Valley of California. The constructed wetland was found to be very effective at trapping pyrethroids in its bottom sediments. Budd et al (2011) found low to moderate rates of microbial biodegradation of pyrethroid pesticides within the constructed wetland. The pesticides had measured half lives of several months to a year in the wetland sediment. Given the low rate of biodegradation, Budd et al (2011) expressed some concern that pyrethroids could persist and possibly accumulate in the sediments of the constructed wetland, increasing the potential toxicity risk for the fish and wildlife that utilize these habitats.

Werner et al (2010) evaluated the capability of a 400 meter vegetated swale to reduce toxicity from alfalfa and tomato fields that were treated with permethrin. Based on their tests, they concluded that the swale had very little capability to reduce permethrin toxicity.

Delorenzo et al (2012) monitored for the presence of urban pesticides in the water column of stormwater ponds that drained residential catchments in coastal South Carolina. Pyrethroids were detected in 10% of the ponds sampled, and occasionally exceeded benchmarks to protect aquatic life.

8.6 Other Pyrethroid Management Strategies

The quest continues to develop an environmentally safe insecticide that kills target insect species but is not toxic to aquatic life. At this point, it is difficult to fully assess the environmental risk of pyrethroid pesticides in urban watersheds, or for that matter, whether any new insecticides on the market would be a less harmful substitute. For example, neonictinoids and fipronil were recently introduced but recent research indicates that they are frequently detected in urban stormwater and sediments and are also known to be toxic to aquatic life at very low concentrations. More research is needed about their watershed sources, pathways or dynamics or whether they can be effectively treated by urban BMPs.

The other key pyrethroid management strategy is to expand public outreach and social marketing efforts to educate homeowners, landscape contractors, applicators and others about proper insecticide use and applications. It would also be helpful to sample insecticide levels in streams in urban neighborhoods that were specifically targeted for outreach and compare them to control streams that were not targeted.

Section 9: Legacy Organochlorine (OC) Pesticides

(DDT, DDE, Chlordane)

9.1 Overall Findings

- Organochlorine (OC) pesticides include insecticides such as DDT, DDE and dieldrin that have been banned for decades but still persist in the environment.
- These legacy pesticides meet all the criteria to classified as an urban toxic contaminant (UTC), despite the fact that were also historically applied to crops, orchards and wetlands (i.e., mosquito control). The main reason is that urban soils that were contaminated by OC pesticides in the past tend to be much more mobile in urban watersheds.
- The encouraging news is that nearly all monitoring studies have shown sharply declining trends in OC pesticides in urban stormwater runoff and creek sediments, which appears to have greatly reduced their bioaccumulation and toxicity in vertebrates, such as fish, eagles and marine mammals.
- Very limited monitoring data has been conducted to date on whether OCP pesticides are trapped with urban BMPs, but given their affinity for sediment particles, it is very likely that they would be present within BMP sediments and persist over time. The greatest risk would presumably be for older stormwater ponds that have trapped sediments laden with OC pesticides for many decades.
- OC pesticides are a classic example of how highly persistent and lipophilic insecticides can have an enduring environmental impact nearly a half century after their use was banned. The long watershed lag time associated with OC pesticides in the Chesapeake Bay watershed suggest that continued tracking may be warranted for another decade or two.

9.2 Background and Characteristics of Legacy OC Pesticides

OC pesticides meet the six criteria to be classified as a UTC since (a) they were historically applied in urban areas (b) their highest concentrations in water and sediments occur in urban watersheds (c) urban stormwater runoff is their primary pathway in the watershed (d) they tightly bind to sediment particles and essentially behave like them (e) contaminated soils are still located in upland areas (although they may be gradually moving down in the watershed over time) and (f) limited evidence suggests that OC pesticide may be found in the bottom sediments of older stormwater ponds.

This is not meant to imply that OC pesticides are not also generated by soil erosion that occurs on crops and other land uses in the Chesapeake Bay watershed. The main reason is soils that were contaminated by OC pesticides tend to be much more mobile in urban watersheds than agricultural ones.

OC pesticides are highly persistent in the environment, with half-lives typically measured at a thousand days or more. They have a high affinity for soil organic matter and tightly bind to soil and sediment particles. Connor et al (2007) reported a very strong relationship between high OC pesticide concentrations and high suspended sediment levels in urban storm water runoff.

9.3 Pathways and Trends with Legacy OC Pesticides

Connor et al (2007) investigated DDT and dieldrin sources in the San Francisco Bay area more than thirty years after their use was banned. The research team concluded that sediments carried in urban stormwater runoff were the greatest source of DDT and dieldrin in the region, far exceeding the inputs from agricultural runoff and irrigation return flows from the Central Valley of California. Stormwater runoff also dominated all other sources of DDT and dieldrin in the San Francisco Bay region, such as atmospheric deposition and discharges from municipal or industrial wastewater treatment plants.

Gilbreath et al (2012) also monitored legacy pesticides in urban runoff from the San Francisco Bay region, and found that DDT and dieldrin were routinely detected during storm events. The EMC for DDT ranged from 5.1 to 59 ng/l (median: 15 ng/l), and was positively correlated with elevated turbidity and flow levels.

The persistence of OC pesticides in soils and estuarine sediments after so many decades is a sobering example of the lag time between management action and realization of environmental benefits. Connor et al (2007) estimated that it may take several more decades for DDT and other OC pesticides to diminish to the point that they would no longer bio-accumulate in the tissue of fish, birds or marine mammals of the San Francisco Bay.

Gilliom et al (2006) summarized the findings of a national assessment of DDT levels in streambed sediments and fish tissue within agricultural and urban streams during the period 1992 to 2001. At that point in time, DDT was detected in 80% of streambed sediments and 94% of fish tissues sampled in urban streams, compared to a DDT detection rate in agricultural streams of 57% for streambed sediments and 92% of fish tissues, respectively.

Gilliom et al (2006) concluded that OC pesticides were still the leading cause of stream impairment for sediments and fish tissue across the country for any pesticide during the 1992 to 2001 era. In general, Gilliom et al (2006) found that DDT levels were about twice as high in urban streams compared to agricultural streams, which was attributed to the greater mobility of eroded soils that were historically contaminated by OC pesticides.

Van Metre and Mahler (2005) examined trends in OC pesticides in sediment cores for urban lakes across the country. They reported sharp drops in DDT levels in recent lake sediments (circa 2005) and predicted a further 50% drop in DDT levels in lake sediments by the year 2015. This finding is reinforced by more recent monitoring studies. For example, Ding et al (2010) investigated legacy pesticide levels in urban creek sediments in Illinois (e.g., DDT, DDE). While they found at least one legacy pesticide in 95% of the urban sediment samples, the concentrations of the legacy pesticides were low, and did not induce toxicity.

More recent USGS sampling indicates OC pesticides are now being detected at even lower concentrations in both urban and agricultural streams (Stone, 2014). Gilliom et al (2006) also documents the impressive decline in DDT levels in fish tissue in the three decades from 1970 to 2000. Both authors caution, however, that OC pesticides can degrade into multiple congeners that can persist in the environment and whose potential environmental impact is still very poorly defined.

9.4 Estimated Legacy OC Pesticide Removal by Urban BMPs

It is somewhat surprising how little monitoring has been conducted to define how well urban BMPs remove legacy pesticides over the last 30 years. This appears to reflect an assumption that banning legacy pesticides would effectively keep them out of the environment and further urban BMP monitoring would have little management value. At any rate, no monitoring studies could be found that investigated OC pesticide removal rates for urban BMPs.

Despite the lack of monitoring data, it can be safely assumed that OC pesticides behave in the same manner as a sediment particle, and should have a similar BMP removal rate to the suspended sediment benchmark. This conclusion is also supported by the chemical characteristics of legacy OC pesticides (see Table A-34).

The monitoring evidence for OC pesticides being trapped in BMP sediments is also rather sparse. Parker et al (2000) measured OC pesticides in stormwater pond sediments in the arid Arizona environment. They discovered that OC pesticides were routinely detected in nearly every stormwater pond they sampled (DDT degradates, DDE and dieldrin). Overall, Parker et al (2000) noted that OC pesticide levels were all found at fairly low levels when the data was collected some 20 years ago.

9.5 Legacy OC Pesticide Management Strategies

OC pesticides appear to be a classic example of how to manage pollutants with a long watershed lag time. The Bay watershed is currently experiencing a stage of relatively low exposure to legacy OC pesticides (i.e., only localized toxicity or bioaccumulation issues), and they may well decline to acceptable levels in the environment within a decade or two.

Section 10: Legacy Organophosphate (OP) Pesticides

10.1 Overall Findings

- Organophosphate (OP) pesticides refers to a group of insecticides that include chlordane, chlorpyrifos, diazinon and dichlorovos that were introduced toward the end of the last century to replace the more persistent OC pesticides.
- Research emerged in the late 1990s that confirmed that that these relatively non-persistent insecticides were highly toxic to aquatic invertebrates in urban streams at extremely low concentrations.
- Consequently, the use of most OP pesticides has been banned or highly restricted (chlordane in 1978, chlorpyrifos and diazinon in 2000-2002). The use of dichlorvos is still allowed, although it is more restricted than in the past.
- This class of legacy pesticides meets most (but not all) of the criteria to be classified as an urban toxic contaminant. They are predominately found in urban watersheds, are highly mobile, are carried by urban stormwater runoff and generally behave like a sediment particle.
- No research could be found on how effective urban BMPs were in reducing OP pesticides, nor was there any research available on their presence and persistence within BMP sediments. Given their chemical characteristics, however, it is not very likely that OP pesticides would persist long in pond sediments.
- The encouraging news is that levels of the two most common OP pesticides -- diazinon and chlorpyrofos -- have declined sharply in stormwater runoff and urban creek sediments since they were banned some 15 years ago. This represents a real success story about how quickly less persistent pesticides can be eliminated from the environment due to short watershed lag times.

10.2 Background on Legacy OP Pesticides

In the context of this report, organophosphate (OP) pesticides refers to a group of insecticides including chlordane, chlorpyrifos, diazinon and dichlorovos.

These formulations were introduced toward the end of the last century to replace the more persistent and harmful OC pesticides, which, in turn, are now being replaced by even newer insecticides, such as pyrethroids, neonictinoids and fiprinol.

Chlordane was banned in 1978, followed by chlorpyrifos and diazinon in 2000-2002. The use of dichlorvos is still allowed, although it is more restricted than in the past

Diazinon and chlorpyrifos have a half life of about 40 days in both soil and water, and are both very soluble and highly mobile (Schueler, 1999). Several research studies profiled in Schueler (1999) indicate that diazinon and chlorpyrifos are both highly toxic at extremely low concentrations.

Gilliom et al (2006) found that diazinon and chloropyrifos were more frequently detected in urban streams compared to agricultural streams, despite the fact that the greatest use of these pesticides was on crop lands (corn, alfalfa, wheat) and orchards. Gilliom et al (2006) did not detect diazinon and choropyrifos in groundwater samples during a national assessment, which reflects their relatively low persistence in the environment.

Researchers in the San Francisco Bay region found that diazinon and chlorpyrifos were highly mobile in stormwater in urban creeks. Both insecticides appear are generated in upland landscape positions with residential subwatersheds, and more than 90% of their load is delivered via urban stormwater runoff.

The use of diazinon and chlorpyrifos were largely eliminated around 2002, and their concentrations in urban creeks and sediments has rapidly declined. When combined with their (relatively) short persistence in soils and water, these insecticides no longer exerting toxic impacts to the aquatic environment.

As noted earlier, the application of dichlorvos is still allowed and it continues to have an increasing impact on urban streams. Stone et al (2014) reported that dichlorvos was the second most frequently detected insecticide in urban streams across the nation from 2002 to 2011. More than 45% of urban streams across the nation exceeded their aquatic life benchmark for dichlorvos (fipronil exceeded benchmarks at more than 70% of urban streams).

10.3 Trends in Legacy OP Pesticides Over Time

Stone et al (2014) reported that diazinon and chlorpyrifos were among the most frequently detected and most consistently toxic insecticides measured in urban streams in the 1990's and the first few years of the new century. Subsequent national monitoring from 2002 to 2011, however, confirms that chlorpyrifos and diazinon are rarely detected in either urban or agricultural streams (Stone et al, 2014). The number of urban streams that exceeded aquatic life benchmarks also dropped sharply for both insecticides. Both trends were directly attributed to the stringent restrictions imposed on their use at the turn of the century.

Gilliom et al (2006) reported on long term trends in diazonin levels in an urban creek before and after the use of OP pesticides was heavily restricted. They reported that diazinon levels dropped sharply in Accontink Creek, VA within a few years after the restrictions were imposed. The rapid response observed after diazinon was restricted reflects its relatively short persistence in the environment (e.g., soil half lives measured at 30 to 40 days).

10.4 Estimated Legacy OP Pesticide Removal by Urban BMPs

No monitoring data were available to assess the ability of urban BMPs to remove OP pesticides. Two studies examined the capability of agricultural BMPs to remove OP pesticides. Budd et al (2010) found that a constructed wetland was effective in removing chlorpyrifos in California, whereas Werner et al (2010) reported that a grass swale had little value in reducing toxicity from irrigation return flows containing high levels of chlorpyrifos.

Despite the lack of monitoring data, it can be safely assumed that OP pesticides behave much like a sediment particle, and should have a similar removal rate to the suspended sediment benchmark within an urban BMP. This conclusion is also supported by the chemical characteristics of legacy OP pesticides (see Table A-34).

In addition, no monitoring data was available to determine the presence and persistence of OP pesticides in urban or agricultural BMP sediments over time.

10.5 Legacy OP Pesticide Management Strategies

Banning OP pesticides represents an encouraging success story about how quickly less persistent pesticides can be eliminated from the environment due to short watershed lag times. Due to their high mobility, diazinon and chlorpyrifos largely disappeared from urban streams within 5 years after they were effectively banned. This suggests that it may wise to either ban or further restrict the last OP pesticide still in use (dichlorvos) should an environmentally safer replacement exist.

The other key management issue is how to improve outreach to landscape contractors and individual homeowners that are applying retail insecticides to control insects on their properties. Early research profiled by Schueler (1999) indicates that improper insecticide applications at a handful of homes in an residential watersheds can create toxic conditions in urban streams.

Section 11: Plasticizers (Phthalates)

11.1 Overall Findings

- Phthalates are a type of plasticizer that is emitted from a diverse array of flexible PVC products and coatings. These plasticizers are increasing detected in urban stormwater and sediments, and are suspected to be an endocrine disrupting compound (Clara et al , 2010).

- Less than a dozen monitoring studies have been conducted on phthalates around the globe, all of which were located in Europe or the west coast. While phthalates appear to meet many of the criteria to be classified as a UTC, major data gaps on their sources and pathways in the Bay watershed prevent us from conclusively assigning them to the UTC category.

- For example, several European studies suggest that wastewater treatment discharges and/or land application of municipal biosolids are responsible for significant phthalate loads. At this time, it cannot be determined whether wastewater phthalate loads are greater than urban stormwater loads at the watershed level.

- A handful of monitoring and modeling efforts have sought to estimate the phthalate removal rate that occurs in urban BMPs. Based on this limited data, it is estimated that phthalate removal will be comparable to the TSS removal benchmark.

- Other management strategies to reduce phthalates could involve wastewater treatment plant upgrades and deeper incorporation of municipal biosolids.

11.2 Background and Characteristics of Phthalates

Phthalates are a plasticizer, especially for flexible polyvinyl chloride (PVC). The impact of phthalates on human health is still be assessed, and no benchmarks have been established for acceptable human exposure. The limited available research indicates that phthalates are ubiquitous in the urban environment, and are detected in urban rain water, surface water, wastewater, stormwater and sediments. Once released into the environment, phthalates tend sorb to sediment particles and can then be mobilized in urban stormwater. Clara et al (2010) found that phthalates were strongly associated with sediment particles and that stormwater runoff was a major source of the phthalate load in a watershed.

11.3 Sources and Pathways for Phthalates

In a European study, Bjorkland (2010) reported that phthalates were frequently detected in both stormwater runoff and urban stream sediments. Bjorkland used a detailed emission inventory and determined that 95% of the total phthalate emissions were derived from flexible PVC products, such as roof coating, cable coating, garden hoses and vehicle under-coating, with remainder coming from paint and sealants. Bjorkland (2010) also concluded that atmospheric deposition was not a major source of phthalates in urban watersheds.

The other significant source of phthalates in the watershed are WWTPs. Research indicates that while WWTPs can sharply reduce incoming phthalate concentrations, they can still discharge in them in treated effluent. For example, Rule et al (2006) found that residential wastewater influent had very high phthalate concentrations which were effectively removed (~95%) during the wastewater treatment process. Phthalates that are removed do persist in sludge and can potentially leach out when municipal biosolids are applied to croplands.

11.4 Estimated Phthalate Removal by Urban BMPs

Only one study has monitored phthalate removal in an urban BMP. Zhang et al (2014) reported >80% removal of phthalates in an Australian biofilter. A number of European researchers have tried to model how urban BMPs remove phthalates (and other micro-pollutants), but data limitations prevented them from providing reliable estimates of removal efficiency (Bjorkland, 2011, DeKeyser et al 2010, Vezzaro et al 2010 and 2011).

Ren and Smith (2012) investigated whether synthetic liners used in trenchless repair of leaking water and sewer pipes could be a source of phthalates, and concluded that they were not a significant source.

11.5 Other Phthalate Management Strategies

Research is the main phthalate management strategy to pursue at this time, as we lack basic data on their loads, sources and pathways in urban watersheds, and do not fully understand their potential impacts to human health.

Section 12: Flame Retardants
(PBDE, polybrominated diphenyl ether)

12.1 Overall Findings

- Only a handful of research studies have assessed this potential UTC, most of which were located in Europe or California. As with phthalates, flame retardants appear to meet many of the criteria to be classified as a UTC, but major data gaps for the Bay watershed makes it difficult to conclusively assign them to this category.

- For example, some studies indicate that wastewater discharges and/or land application of municipal biosolids could also be a significant potential source of flame retardants at the watershed scale.

- PBDE is a flame retardant that includes numerous compounds that are persistent and tend to bio-accumulate in fish and wildlife tissues in much the same way as PCBs or dioxins.

- PBDE is emitted from several sources, including atmospheric deposition over urban watersheds. Like many other UTCs, PBDE strongly sorbs to soil, sediments and organic matter, and moves through the watershed when these particles are mobilized by stormwater runoff.

- Given the characteristics as PBDE, it is expected that it can be effectively reduced by urban BMPs that are able to trap or filter out sediment particles. The limited European research on PBDE removal by biofilters and retention ponds suggests that it falls within the same general range as suspended sediment removal.

12.2 Background and Characteristics of PBDE

PBDE is a flame retardant that includes chemical compounds that are persistent, lipophilic, hydrophobic and tend to bio-accumulate in the tissues of fish and wildlife (Kupper et al, 2006). In many ways, PBDEs have many of the same characteristics as PCBs and dioxins, which influences their treatability as an urban toxic contaminant.

Flame retardants are commonly used in household items such as computer and television wiring, plastics, foam cushions and insulation foams. This contaminant has recently been studied in the United States and Europe due to concern over its use (Focazio et al, 2008, Gilbreath et al, 2012, Muresan et al, 2010, Oros et al, 2005, Rule et al, 2006, Teil et al, 2014). Due to its widespread use in the urban sector, flame retardants have become ubiquitous in the urban environment.

Over 200 congeners of flame retardants exist, though only a few have been of major concern. Polybrominated diphenyl ethers (PBDE) are the most commonly studied flame retardants and contain the most harmful and frequently used congeners. PBDEs are lipophilic, have low solubility and increased resistance to degradation in water which allows for strong persistence in the environment (Morace, 2012). PBDEs are similar to PCBs and dioxins in that they are hydrophobic and bio-accumulate in tissue.

Gorgy et al (2011) reports that PBDE and its congeners break down slowly over time, and have a half life that is measured in years. For example, Palm (2002) reported that BDE-209 has a half life of 319 days in air, 150 days in water and 600 days in sediments.

12.3 PBDE Sources and Pathways

Most of the research done on flame retardants in the environment has occurred in Europe or in the San Francisco Bay region. Oros et al (2005) reported that PBDE levels in San Francisco Bay are among the highest recorded in the world. This finding prompted a series of monitoring investigations to track the sources and pathways of PBDE.

PBDE is volatile and is capable of long range transport in the atmosphere. Kupper et al (2008) and Oram et al (2008) concluded that atmospheric deposition is an important source of PBDE, especially when it occurs on impervious surfaces where it can be washed away in stormwater runoff. PBDE compounds strongly sorbs to soils, sediments and organic matter. PBDE moves through the watershed when these suspended sediment particles are mobilized by stormwater runoff (Muresan, et al 2010).

Gilbreath et al (2012) measured a median EMC of 50 ng/l for PBDE in urban storm flows, and also reported that PBDE was strongly correlated with turbidity. Nearly all the mass of PBDE was transported during storm flow in the urban creeks, rather than base flow. On a regional level, Gilbreath et al (2012) estimated that urban stormwater contributes 71 to 77% of the total PBDE load delivered to the San Francisco Bay.

There are also indoor emissions that occur when PBDE-treated fabrics are laundered and end up in wastewater. Most wastewater treatment plants are effective in removing PBDE, although it does concentrate within municipal biosolids (Rule et al, 2006). When these biosolids are applied to croplands, there is some risk that they can leach into streams and rivers.

Gorgy et al (2011) conducted a series of soil column studies to determine whether PBDE could leach from municipal biosolids applied to croplands, and concluded that it could be a significant pathway to reach the aquatic environment. Rief et al (2012) also reports that land application of biosolids is a meaningful pathway for PBDE to enter the environment

12.4 Estimated PBDE Removal by Urban BMPs

Because PBDEs have similar characteristics as PCBs and dioxins, it is anticipated that PBDE runoff can be effectively reduced by urban BMPs that are able to trap or filter out sediment particles. The limited research seems to support this contention

Three European studies investigated whether urban BMPs can effectively remove flame retardants (Table 17). Sebastian (2014) reported that a retention pond removed 20 to 66% of PBDE over the one year monitoring period. They noted that while PBDE was clearly associated with sediment particles, it did not always behave like them when it came to settling out in the pond. Biofilters had high to very high PBDE removal efficiencies (Table 17), depending on the type of compound.

Table 17. Summary of PBDE Removal Efficiencies

Author	Year	Location	Removal	Contaminant	BMP
Bester	2009	Germany	96-99 %	Lipophilic compounds	Biofilter peat and sand
			81-98 %	Hydrophilic compounds	Biofilter peat and sand
Gilbert	2012	Paris	44-87 %	Light congeners	Biofiltration
			75 %	Heavy congeners	Biofiltration
Sebastian	2014	France	20-66 %	PBDEs	Dry retention pond

Section 13: Dioxins and Furans

13.1 Key Findings on Dioxins/Furans

- Dioxins have a localized impact on three areas in the Chesapeake Bay, where they have been found to contaminate fish tissue, due historic industrial discharges.
- Dioxins and related compounds are also found at lower, but detectable levels in many urban watersheds. The environmental risks posed by these low concentrations, however, are not well understood.
- The primary sources of dioxins in urban watersheds are air deposition onto impervious surfaces (and subsequent wash-off), as well as erosion or wash-off of older contaminated soils.
- In general, dioxins and furans have the least certainty and most data gaps of any class of urban toxic contaminants reviewed in this study.
- It appears that dioxins/furans meet most, if not all, of the six criteria to qualify as an urban toxic contaminant, although this conclusion is based on very limited monitoring data. As such, it is likely that dioxins and furans will be trapped by existing or future urban stormwater BMPs.
- There is insufficient monitoring data, however, to derive a credible estimate of the background dioxin load from urban areas in the watershed, what fraction of that load may be effectively removed by stormwater BMPs, and how much the load reduction might diminish the environmental impact of dioxins.
- Much more research is needed on this toxin category, especially to determine whether dioxins are accumulating in stormwater BMP sediments, and whether they are toxic or not.

13.2 Background on Dioxins

Dioxins and furans are generic terms for a group of toxins that contain chlorine and carbon atoms associated with dibenzodioxin and dibenzofurans. They have no known use and are not intentionally produced. They are inadvertently produced by combustion processes that involve chlorine in uncontrolled reactions.

Dioxins and furans are released by the combustion of fossil fuels and wood, and during municipal, medical and hazardous waste incineration. Dioxins are persistent and lipophilic compounds that accumulate in the environment (Horstmannn and McLachan, 1995).

13.3 Dioxins: Environmental Risk and Trends

Dioxins and furans are likely human carcinogens and cause other human health impacts. The impact of dioxins and furans is localized within the Chesapeake Bay watershed -- three sites in the watershed which are associated with industrial discharges (the Elizabeth River and two sites in the North Branch of the Potomac River). CBP (2012) documents limited contamination of fish and shellfish in these areas of the Chesapeake Bay watershed.

13.4 Sources and Pathways of Dioxins

The sources of dioxins include incomplete waste combustion, legacy chemical manufacturing sites, paper mills and atmospheric deposition. Some are even produced by night-time fireworks and others are found as impurities in certain organo-chlorine pesticides (e.g., 2-4-D). Schmid et al (2014) looked at the role of evening fireworks in producing dioxins in Switzerland, and concluded that these pyrotechnics could produce from 2 to 14% of the dioxin load in that country (although waste incineration was still by far the major source).

Urban stormwater runoff is the greatest pathway of dioxins to the San Francisco Bay, as a result of wash off of air emissions and soil particles deposited on impervious surfaces (Gilbreath et al 2012). Fisher et al (1999) came to the same conclusion in Santa Monica Bay after a extensive sampling effort. Their monitoring revealed that dioxins were found in urban rainfall, as well as the wash-off of particles from impervious surfaces and the erosion of older contaminated urban soils (Fisher et al, 1999). Dioxin levels reported for these urban watersheds were higher than those reported for Houston, TX (Suarez et al, 2006).

Horstmann and McLachan (1995) sampled dioxins and furans concentrations in street and roof runoff. They also measured high concentrations of both within household wastewater (i.e., laundry and shower water), and concluded that these two sources were responsible for the presence of these compounds in sewage sludge.

Rifai et al (2013) sampled the origins of dioxin levels in rainwater and sediments near the port of Houston. The research team concluded that the erosion of legacy sediments from the urban watersheds were responsible for 70 to 95% of the total annual dioxin loads, as compared to 5% of the load generated by wastewater discharges or atmospheric deposition.

13.5 Dioxin Characteristics

Researchers generally report that dioxins are mostly found in a particulate phase, and effectively act as a sediment particle when deposited on impervious surfaces or urban soils. Suarez et al (2006) sampled dioxin levels in stormwater runoff in the Houston, Texas area, and found higher concentrations associated with the particulate phase than the dissolved phase. Dixons tend to be lipophilic, which increases the potential bio-accumulation in fish tissue.

13.6 Dioxin Concentrations in Urban Runoff and Urban Sediments

Table A-3 shows the limited monitoring data available on dioxin concentrations in urban runoff, whereas Table A.4 summarizes the dioxin concentrations found in urban sediments. Dioxin concentrations were higher in urban stormwater runoff as compared to urban dry weather flows, and the highest overall levels were correlated with commercial and industrial urban land uses (Fisher et al, 1999).

Suarez et al (2006) reported that the primary source of dioxins were air deposition, but the runoff concentrations of dioxins were the same or even lower than those observed in the receiving water body (i.e., Houston Ship Channel).

13.7 Estimated Dioxin Removal by Urban BMPs

No monitoring data was discovered to determine whether urban BMPs can remove dioxins in urban runoff. The lack of data is due to the difficulty and expense to obtain reliable dioxin samples in the field during storm conditions.

13.8 Dioxin Accumulation in BMP Sediments

No data was available to document whether dioxins are trapped in BMP sediments or media, and whether they have the potential to accumulate and persist over time.

13.9 Other Dioxin Management Strategies

Other dioxin management practices include reducing waste incineration and removing contaminated sediments from industrial hotspots.

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Appendix A
Supplementary Data Tables

1. Polychlorinated Biphenyls
PCBs

Table A-1 PCB Concentration in Sediments

Author	**Year**	**Location**	**Concentration ng/g**	**Sample Type**
Hartwell	2007	Chesapeake	ND – 122	Sediment
Jartun	2008	Norway	29	Sediment traps
Ko & Baker	2004	Susquehanna	165	Sediment
		N. Chesapeake Bay	80	Sediment
Nowell	2013	Atlanta	ND-500	Sediment
		Boston	100	Sediment
		Dallas	ND-10	Sediment
		Denver	ND-50	Sediment
		Milwaukee	10	Sediment
		Seattle	ND-50	Sediment
		Salt Lake City	10	Sediment
Parker	2000	Arizona	7	Soil
Teil	2014	France	12-199	Sediment
Velinsky	2011	Anacostia	288	Surface sediments
Yee	2010	San Francisco	ND-7650	Sediment
ND: Not detected				

Table A-2: PCB Concentration in Surface and Storm Water

Author	Year	Location	Concentration ng/L	Sample Type
Bressy	2012	Paris	5.2	Surface water
Gilbreath	2012	San Francisco	24	Storm flow
Ko & Baker	2004	Susquehanna	1.7	Surface water
Teil	2014	France	2.6-8.7	Surface water

2. Dioxins

A-3 Dioxin Concentrations in Stormwater				
Author	**Year**	**Location**	**Median Concentration** ng/L	**Sample Type**
Fisher et al	1999	Santa Monica Bay	0.035	Stormwater
Gilbreath et al	2012	San Francisco Bay	3.7	Stormwater
Horstmann	1995	Germany	0.0044	Stormwater
Suarez et al	2006	Houston	0.12	Stormwater

A-4 Dioxin Concentrations in Urban Sediments				
Author	**Year**	**Location**	**Median Concentration** ng/g	**Sample Type**
Horstmann	1995	Germany	0.012	Catch basin sediment
Israelsson et al	2014	New Jersey	10	Urban sediment

3. Polycyclic Aromatic Hydrocarbons (PAHs)

A-5: PAH Concentration Chesapeake Bay Sediments				
Author	**Year**	**Location**	**Concentration** mg/kg	**Sample Type**
Hartwell	2007	Chesapeake Bay	0.004 – 22	Surficial sediments
Hwang	2006	Anacostia	32.4-103	River sediments
Ko & Baker	2004	Susquehanna	6	Sediments
		N. Ches Bay	3	Sediments
Velinsky	2011	Anacostia	0.01-32	River sediments

A-6: PAH Concentration in Non-Chesapeake Bay Sediments				
Author	**Year**	**Location**	**Concentration mg/kg**	**Sample Type**
Bathi	2011	Alabama	92-2637	River sediments
DeBruyn	2009	Lake Erie	0.42	Lake sediments
Echols	2008	Missouri River	0.150-3.97	River sediments
Nowell	2013	7 US Urban Cities	0.1-10	Stream sediments
Sanders	2002	Georgia	0.97	River sediments
Selbig	2013	Wisconsin	22.5	Stream Sediments
Van Metre	2004	Boston	30	Lake sediment
Yang	2010	Texas	11	Stream sediments
			5.7-10	Lake Sediments

A-7: PAH Concentration in Stormwater Sediments				
Author	**Year**	**Location**	**Concentration** mg/kg	**Sample Type**
Brown	2006	New Zealand	136	Catch basin sediment
Selbig	2013	Wisconsin	72.85	Suspended Sed.
Selbig	2013	Wisconsin	47.5	Stormwater sed.
Yang	2010	Texas	43	Suspended sed.

A-8 PAH Concentration From Parking Lots w/ Seal Coat				
Author	**Year**	**Location**	**Concentration** mg/kg	**Sample Type**
Crane	2010	Minnesota	54	Unsealed parking lot
			620	Asphalt sealcoat
			3500	Coal Tar sealcoat
Mahler	2010	Texas	4760	Coal Tar sealcoat
			9	Non-coal tar sealcoat

A-9 PAH Concentrations in Sediments of Stormwater Ponds				
Author	**Year**	**Location**	**Concentration** mg/kg	**Sample Type**
Gallagher	2011	Baltimore Co.	1.05	Stormwater pond sediments
Jartun	2008	Norway	3.4	Stormwater sediment traps
Weinstein	2010	South Carolina	0.186-159.04	Stormwater Pond sediments

A-10 PAH Concentrations in Urban Stormwater				
Author	**Year**	**Location**	**Concentration** µg/L	**Sample Type**
Diblasi	2009	UMD	2.08	Bioretention inflow
Gobel	2007	Germany	2.61	Stormwater
			0.44	Roof runoff
Rule	2006	England	0.7-1.6	Stormwater
Stein	206	Los Angeles	2.18	Stormwater
Steuer	1996	Wisconsin	ND – 4.8	Stormwater

A-11 PAH Concentrations in River Samples				
Author	**Year**	**Location**	**Concentration** µg/L	**Sample Type**
Alvarez	2008	Shenandoah/James	ND -0.017	Surface water
Gilbreath	2012	San Francisco	9.6	Storm flow
Hwang	2006	Anacostia	1.51-12.5	Storm flow
Ko & Baker	2004	Susquehanna	0.067	Surface water
Oros	2007	San Francisco	0.005-0.147	Surface water

4. Total Petroleum Hydrocarbons

A-12 Petroleum Hydrocarbon Concentrations in Sediment				
Author	**Year**	**Location**	**Concentration** µg/kg	**Sample Type**
LeFevre	2012	Minnesota	ND-3.0	

A-13 Petroleum Hydrocarbon Concentrations in Stormwater				
Author	**Year**	**Location**	**Concentration** µg/L	**Sample Type**
Gobel	2007	Germany	510-6,500	Trafficked area runoff
James	2010	Tennessee	22,000	Impervious Runoff
			5,000	Pervious runoff
Kayhanian	2007	California	1,400	Highway runoff
Ronias	2014	England	100-300	Detention pond runoff

5. Mercury

A-14 Mercury Concentration in Surface and Storm Water					
Author	**Year**	**Location**	**Median Concentration** ng/L	**Sample Type**	**% methylated**
Clark	2004	NSQD	200	Stormwater	
Clary	2011	ISQD	200	Urban stormwater	
Gilbreath	2012	San Francisco	29*	Stormflow	
Hurley	1995	Wisconsin	4.5	Stormflow	
Kannan	1998	Florida	4.1	Surface water	
Lawson	2001	Patuxent	1.3*	River Stormflow	
		Susquehanna	7.7*	River Stormflow	2
		Patapsco	5.4*	River Stormflow	8
		Potomac	18.7*	River Stormflow	0.6
		Rappahannock	5.0*	River Stormflow	3
		Choptank	3.2*	River Stormflow	8
		Herring Run	12.6*	Creek Stormflow	1
Marvin-Dipasquale	2009	OR, WI, FL	1.3	Pore water	9
Scudder	2009	National	2.1	Stream samples	
*Mean value					

A-15 Mercury Concentrations in Sediments

Author	Year	Location	Median Concentration ng/g	Sample Type
David	2009	San Francisco	200	Suspended sediments
Jartun	2008	Norway	60	Urban runoff sediments
Kannan	1998	Florida	15	Creek sediments
Marvin Dipasquale	2009	OR, WI, FL	23.1	Urban stream sediment
Mason	1999	Baltimore Harbor	341	Surficial sediments
		Upper Potomac	162	Surficial sediments
		Boston Harbor	453	Surficial sediments
		Lower Hudson	699	Surficial sediments
Meador	2005	Alaska	50	Sediment cores
		California	115	Sediment cores
Scudder	2009	National	31.8	Stream bed sediments
Velinsky	2011	Anacostia	440*	Sediment core
Yee &McKee	2010	San Francisco	170*	Sediments and soil
* Mean value				

6. Arsenic

A-16 Arsenic Concentrations in Stormwater

Author	Year	Location	Median Concentration µg/L	Sample Type
Clark	2004	NSQD	3	Stormwater
Clary	2011	ISQD	3.3	Stormwater
Gilbreath	2012	San Francisco	1.1	Stormflow mean
Kayhanian	2007	California	1.1	Highway runoff
Leisenring	2014	ISBD	0.58 (dissolved)	BMP inflow
			1.30	BMP inflow

A-17 Arsenic Concentration in Sediments

Author	Year	Location	Median Concentration µg/g	Sample Type
Jartun	2008	Norway	3.7	Catch basin sediments

7. Cadmium

A-18 Cadmium Concentrations in Surface and Stormwater

Author	Year	Location	Median Concentration µg/L	Sample Type
Al-Anbari	2008	Australia	2.2	Parking lot runoff
Clark	2004	NSQD	1	Stormwater
Clary	2011	ISBD	1	Urban stormwater
Crabtree	2006	England	0.49	Urban stormwater
Gilbreath	2012	San Francisco	0.27 *	Stormflow mean
Kayhanian	2007	California	0.44	Highway runoff
Lawson	2001	Susquehanna	0.35 *	River Stormflow
		Potomac	0.34 *	River Stormflow
		Rappahannock	0.38 *	River Stormflow
		Choptank	0.34 *	River Stormflow
		Patapsco	0.42 *	River Stormflow
		Herring Run	0.43 *	Creek Stormflow
Leisenring	2014	ISBD	0.12 dissolved	BMP inflow
			0.3 total	BMP inflow
*mean value				

A-19 Cadmium Concentrations in Sediments

Author	Year	Location	Median Concentration µg/g	Sample Type
Echols	2008	Missouri River	0.43	River sediments
Gallagher	2011	Baltimore Co.	0.19	Stormwater sediments
Jartun	2008	Norway	0.42	Urban runoff sediments
Meador	2005	Alaska	0.39	Sediment Core
		California	0.18	Sediment core

8. Copper

A-20 Copper Concentrations in Stormwater and Surface Water				
Author	**Year**	**Location**	**Median Concentration** µg/L	**Sample Type**
Al-Anbari	2008	Australia	36	Parking lot Runoff
Borne	2013	New Zealand	9.2	Retention pond influent
Bressy	2012	France	17	Urban Runoff
Clark	2004	NSQD	16	Stormwater
		Alabama	57	Parking lot runoff
Clary	2011	ISWD	16	Urban stormwater
Crabtree	2006	England	41	Runoff EMC
Kayhanian	2007	California	21.1	Highway runoff
Lawson	2001	Susquehanna	2.7*	River Stormflow
		Potomac	3.8*	River Stormflow
		Rappahannock	3.8*	River Stormflow
		Choptank	3.1*	River Stormflow
		Patapsco	11.1*	River Stormflow
		Herring Run	6.8 *	Stormflow mean
Leisenring	2014	ISBD	4.9 dissolved	BMP inflow
			9.35 total	BMP inflow
Tiefenthaler	2008	S. California	20	Runoff EMC
Sabin	2005	California	27	Runoff EMC)
* mean value				

A-21 Copper Concentrations in Sediments				
Author	**Year**	**Location**	**Median Concentration** µg/g	**Sample Type**
Brown	2006	New Zealand	146	Stormwater sediment
			142	Street solids
Gallagher	2011	Baltimore Co.	42	Stormwater sediments
Jartun	2008	Norway	97	Urban runoff sediments
Stewart	2014	New Zealand	42	Surficial Sediment

9. Chromium

A-22 Chromium Concentrations in Surface and Stormwater

Author	Year	Location	Median Concentration µg/L	Sample Type
Al-Anbari	2008	Australia	1.2	Parking lot Runoff
Clark	2004	NSQD	7	Stormwater
Clark	2004	Alabama	12	Parking lot runoff
Clary	2011	ISWD	7	Urban stormwater
Crabtree	2006	England	6.0	Urban stormwater
Kayhanian	2007	California	5.8	Highway runoff
Lawson	2001	Susquehanna	2.7*	River storm flow
		Potomac	2.7*	River storm flow
		Rappahannock	2.2*	River storm flow
		Choptank	1.7*	River storm flow
		Patapsco	5.7*	River storm flow
		Herring Run	2.7*	Creek storm flow
Leisenring	2014	ISBD	1.2 dissolved	BMP inflow
			4.0 total	BMP inflow
Sabin	2005	California	3.1	Stormwater runoff
* mean value				

A-23 Chromium Concentrations in Sediment

Author	Year	Location	Median Concentration µg/g	Sample Type
Gallagher	2011	Baltimore Co.	42.2	Stormwater sediments
Jartun	2008	Norway	25	Urban runoff sediments
Van Metre & Mahler	2004	Texas	68.5	Suspended sediment
		Boston	275	Suspended sediment

10. Iron

A-24 Iron Concentrations in Stormwater

Author	Year	Location	Median Concentration µg/L	Sample Type
Al-Anbari	2008	Australia	13	Parking lot Runoff
Clark	2004	Alabama	5,170	Parking lot runoff
Gilbreath	2012	San Francisco	5,000	Stormflow mean
Kayhanian	2007	California	12,600	Highway runoff
Leisenring	2014	ISBD	53 dissolved	BMP inflow
			686 total	BMP inflow

11. Lead

A-25 Lead Concentrations in Stormwater and Surface Water				
Author	**Year**	**Location**	**Median Concentration** µg/L	**Sample Type**
Al-Anbari	2008	Australia	168	Parking lot Runoff
Bressy	2012	France	13	Urban runoff
Clark	2004	NSQD	17	Stormwater
Clark	2004	Alabama	19	Parking lot runoff
Clary	2011	ISWD	15.9	Urban stormwater
Gilbreath	2012	San Francisco	12	Stormflow mean
Kayhanian	2007	California	12.7	Highway runoff
Lawson	2001	Susquehanna	2.1 *	River Stormflow
		Potomac	4.2 *	River Stormflow
		Rappahannock	4.0 *	River Stormflow
		Choptank	3.7 *	River Stormflow
		Patapsco	11.5*	River Stormflow
		Herring Run	6.2*	Creek Stormflow
Leisenring	2014	ISBD	0.80 dissolved	BMP inflow
			7.1 total	BMP inflow
Tiefenthaler	2008	S. California	9	Stormwater runoff
Sabin	2005	California	12	Stormwater runoff
* mean value				

A-26 Lead Concentrations in Sediments				
Author	**Year**	**Location**	**Median Concentration** µg/g	**Sample Type**
Brown	2006	New Zealand	208	Stormwater sediment
			262	Street solids
Gallagher	2011	Baltimore Co.	21.9	Stormwater sediments
Jartun	2008	Norway	61	Urban runoff sediments
Meador	2005	Alaska	16.6	Sediment core
		California	21.2	Sediment core
Stewart	2014	New Zealand	53	Surficial Sediment
Van Metre & Mahler	2004	Texas	130	Suspended sediment
		Boston	328	Suspended sediment

12. Nickel

A-27 Nickel Concentrations in Stormwater and Surface Water

Author	Year	Location	Median Concentration µg/L	Sample Type
Al-Anbari	2008	Australia	2.3	Parking lot Runoff
Clark	2004	NSQD	8	Stormwater
Clary	2011	ISQD	3	Urban stormwater
Crabtree	2006	England	5.31	Urban stormwater
Gilbreath	2012	San Francisco	12*	Stormflow mean
Kayhanian	2007	California	7.7	Highway runoff
Lawson	2001	Susquehanna	8.1*	River Stormflow
		Potomac	11.5*	River Stormflow
		Rappahannock	15.9*	River Stormflow
		Choptank	12*	River Stormflow
		Patapsco	31.2*	River Stormflow
		Herring Run	14.1*	Creek Stormflow
Leisenring	2014	ISBD	1.8 dissolved	BMP inflow
			4.6 total	BMP inflow
Sabin	2005	California	6.6	Urban stormwater
* mean value reported				

A-28 Nickel Concentrations in Sediments

Author	Year	Location	Median Concentration µg/g	Sample Type
Echols	2008	Missouri River	9.5	River Sediments
Gallagher	2011	Baltimore Co.	37.2	Stormwater sediment
Jartun	2008	Norway	24	Urban runoff sediments

12. Zinc

A-29 Zinc Concentrations in Stormwater and River Flow

Author	Year	Location	Median Concentration µg/L	Sample Type
Al-Anbari	2008	Australia	998	Parking lot runoff
Borne	2013	New Zealand	35	Retention pond influent
Bressy	2012	France	600	Urban runoff
Clark	2004	NSQD	117	Urban stormwater
Clark	2004	Alabama	167	Parking lot runoff
Clary	2011	ISWD	112	Urban stormwater
Crabtree	2006	England	140	Urban stormwater
Gilbreath	2012	San Francisco	120 *	Urban storm flow
Kayhanian	2007	California	111	Highway runoff
Lawson	2001	Susquehanna	7.0*	River Stormflow
		Potomac	9.8*	River Stormflow
		Rappahannock	7.9*	River Stormflow
		Choptank	20*	River Stormflow
		Patapsco	42.8*	River Stormflow
		Herring Run	10.9*	Creek Stormflow
Leisenring	2014	ISBD	22.6 dissolved	BMP inflow
			49.1	BMP inflow
Tiefenthaler	2008	S. California	151	Urban runoff
Sabin	2005	California	160	Urban runoff)

A-30 Zinc Concentration in Sediments

Author	Year	Location	Median Concentration µg/g	Sample Type
Brown	2006	New Zealand	1079	Stormwater sediments
			528	Street solids
Gallagher	2011	Baltimore Co.	144.5	Stormwater sediments
Jartun	2008	Norway	403	Urban runoff sediments
Stewart	2014	New Zealand	210	Surficial Sediment
Van Metre & Mahler	2004	Texas	487.5	Suspended sediments
		Boston	360	Suspended sediments

A-31 Pyrethroid Levels in Sediments

Author	Year	Location	Concentration ng/g	Pesticide	Sample Type
Budd	2007	California	ND-142	Pyrethroids	Runoff sediments
Holmes	2008	California	2.19-291	Bifenthrin	Stormflow sediment
Lao	2010	California	43	Pyrethroids	creek sed.

A-32 Pyrethroid Levels in Surface Water

Author	Year	Location	Median Concentration µg/L	Pesticide	Sample Type
Budd	2011	California	ND-0.0048	Bifenthrin	Agricultural runoff
			ND-0.035	Permethrin	Agricultural runoff
Weston	2012	California	0.01985	Bifenthrin	Surface water
			0.0064	Cyfluthrin	surface water
			0.0107	Permethrin	surface water

A-33 Legacy Pesticides in Surface Water and Stormwater Runoff

Author	Year	Location	Median Concentration µg/L	Pesticide	Sample Type
Baris	2010	US/Canada	0.059[2]	DDT	surface water
			1.4[2]	Diuron	surface water
			0.4[2]	Chlorpyrifos	surface water
Connor	2007	San Francisco	0.000327	DDT	Water column
			0.000077	Chlordane	Water column
			0.000047	Dieldrin	Water column
Dileanis	2002	Sacramento	0.044	Diazinon	Stormflow
Ensminger	2013	California	0.02	Diuron	storm flow (36%)
Gilbreath	2012	San Fran	0.021	DDT	Stormflow
			0.0094*	Chlordane	storm flow
			0.0018*	Dieldrin	storm flow
Kolpin	2002	US streams	0.04	Carbaryl	surface water (16.5%)
			0.02	Chlordane	surface water (4.7%)
			0.06	Chlorpyrifos	surface water (15.3%)
			0.07	Diazinon	surface water (25.9%)
			0.18	Dieldrin	surface water (4.7%)
			0.02	Lindane	surface water (5.9%)

*Mean value
2. Maximum value
(%) = percent of samples in which it was detected

Table A-34 Flame Retardants Concentrations in Various Flows

Author	Year	Location	Median Concentration µg/L	Flame retardant	Sample Type
Gilbreath	2012	San Francisco	0.05*	PBDE	Flow weighted mean
Muresan	2010	France	0.006*	PBDE	Runoff
Oros	2005	San Francisco	0.0000839	PBDE	Surface water
Rule	2006	England	0.8*	PBDE	Wastewater
Teil	2014	France	0.0013	BDE209	Surface water
*Mean value 2. Maximum value					

Table A-35 Some Key Characteristics of Legacy Pesticides

Legacy Pesticide	$\log K_H$	$\log K_{OC}$	Solubility mg/L
DDT	0.9	5.2	0.04
Chlordane	-0.47	5.5	0.056
Dieldrin	0.05	4.08	0.17
Diazinon	-1.39	2.76	60
Chlorpyrifos	0.037	3.78	0.73

Table A-36 PBDE Concentrations in Sediments and Sludge

Author	Year	Location	Median Concentration ng/g	Flame Retardant	Sample Type
Kolpin	2013	Potomac	0.02-0.32	PBDE	Sediments
Kupper	2008	Switzerland	0.1-282	PBDE	Sewage sludge
Muresan	2010	France	2.24	PBDE	Urban sediment
Oros	2005	San Fran	9.63	PBDE	Sediment
Stewart	2014	New Zealand	10.3	PBDE	Sediment intertidal
			9.6	BDE-209	Sediment intertidal
Teil	2014	France	72.65	BDE-209	Sediment

Table A-37 Phthalate Levels in Sediments

Author	Year	Location	Median Concentration ng/g	HPCP	Sample Type
Teil	2014	France	1230	Phthalates	River sediments
Velinsky	2011	Anacostia	50-7500	Phthalates	Sediment cores

Appendix B
Urban BMP Removal Efficiencies for Selected Trace Metals

B-1: Median Removal Rates for Total Arsenic					
Author	Year	Location	% Removal	N	BMP
Leisenring	2014	ISBD	8	149	Grass strip
			30	37	Grass swale
			14	72	Detention pond
			19	100	Media Filter(mostly sand)
			31	23	Retention Pond
ISBD = International Stormwater BMP Database N = number of paired storm events					

B-2: Median Removal Rates for Total Cadmium					
Author	Year	Location	% Removal	n	BMP
Ladislas	2013	France	38 - 82	L	Floating treatment wetland
Leisenring	2014	ISBD	63	149	Grass strip
			48	124	Grass swale
			-(13)	87	Composite
			20	168	Detention basin
			69	194	Media filter (mostly sand)
			-(4)	130	Porous pavement
			49	366	Retention pond
			35	125	Wetland basin
			44	491	Wetland basin/retention pond
			6	54	Wetland channel
Reddy	2014	Illinois	20 - 100	L	Media (calcite, zeolite, iron)
			3.5 - 8	L	Sand filter
Reddy	2014b	Illinois	18	L	Media filter (biochar)
Stagge	2012	MD	55*	18	Grass swale
Sun	2007	MD	97*	L	Bioretention
ISBD = International Stormwater BMP Database N = number of paired storm events, L= laboratory study *Mean value. Negative(-) represents an increase in concentration					

B-3: Median Removal Rates for Total Chromium					
Author	Year	Location	% Removal	N	BMP
Cedervist	2013	Germany	68-83	L	Infiltration swale normal rain
			26-34	L	infiltration swale extreme rain
Leisenring	2014	ISBD	49	152	Grass strip
			49	29	Grass swale
			87	63	Bioretention
			36	62	Detention basin
			47	109	Media filter (mostly sand)
			-(9)	130	Porous pavement
			66	153	Retention pond
			19	55	Wetland channel
Reddy	2014	Illinois	3-16	L	Media (calcite, zeolite, iron)
			0.4-9	L	Sand Filter
Reddy	2014b	Illinois	19	L	Media filter (biochar)
Roinas	2014a	England	54	4	Vegetated pond

ISBD = International Stormwater BMP Database
N = number of paired storm events, L= laboratory study
*Mean value. Negative(-) represents an increase in concentration

B-4: Median Removal Rates for Total Iron					
Author	Year	Location	% Removal	n	BMP
Leisenring	2014	ISBD	22	53	Grass strip
			7	49	Grass swale
			(-50)	42	Bioretention
			83	56	Composite
			65	132	Media filter (mostly sand)
			74	312	Retention pond
			60	399	Pond/wetland system

ISBD = International Stormwater BMP Database, N = number of paired storm events

B-5: Median Removal Rates for Total Copper					
Author	Year	Location	% Removal	N	BMP
Blecken	2009	Australia	87	L	Vertical flow biofilter
Borne	2013	New Zealand	39	17	Floating treatment wetland
Headley	2007	New Zealand	50-65	L	Floating treatment wetland
Leisenring	2014	ISBD	69	163	Grass strip
			14	256	Grass swale
			39	259	Bioretention
			42	98	Composite
			44	249	Detention basin
			43	330	Media filter (mostly sand)
			35	246	Porous pavement
			51	715	Retention pond
			53	238	Wetland basin
			51	955	Pond Wetland system
Reddy	2014	Illinois	96-100	L	Media(calcite, zeolite, iron)
			5-43	L	Sand filter
Reddy	2014	Illinois	65	L	Media filter (biochar)
Ronias	2014	England	68	4	Vegetated wet pond
Stagge	2012	MD	61*	18	Grass swale
Sun	2007	MD	91*	L	Bioretention
Winer	2000	CBPRD	26	MS	Stormwater dry ponds
			57	MS	Stormwater wet ponds
			40	MS	Stormwater wetland
			97	MS	Bioretention
			42	MS	Grass channel
			70	MS	Dry swale
			11	MS	Wet swale
ISBD = International Stormwater BMP Database CBPRD = Center for Watershed Protection BMP Pollutant Removal Database MS = median of multiple performance studies N = number of paired storm events, L= laboratory study, *Mean value.					

B-6: Median Removal Rates for Lead					
Author	Year	Location	% Removal	n	BMP
Blecken	2009	Australia	99	L	Vertical flow biofilter
Jang	2005	Ohio	>90	L	Hardwood mulch in biofilter
Leisenring	2014	ISBD	76	163	Grass strip
			18	230	Grass swale
			91	121	Bioretention
			76	135	Composite
			48	214	Detention basin
			84	289	Media filter (mostly sand)
			82	183	Porous pavement
			68	618	Retention pond
			45	141	Wetland basin
			67	763	Wetland basin/retention pond
			31	78	Wetland channel
Li and Davis	2008	Anacostia	77	7	Bioretention
Reddy	2014	Illinois	95-100	L	Media (calcite, zeolite, iron)
			1 - 17	L	Sand Filter
Reddy	2014b	Illinois	75	L	Media Filter (biochar)
Stagge	2012	MD	46*	18	Grass swale
Sun	2007	MD	96*	L	Bioretention
ISBD = International Stormwater BMP Database N = number of paired storm events, L= laboratory study, *Mean value.					

B-7: Median Removal Rates for Nickel					
Author	Year	Location	% Removal	n	BMP
Ladislas	2013	France	60-86	L	Floating treatment wetland
Leisenring	2014	ISBD	42	149	Grass strip
			66	23	Grass swale
			-(27)	56	Bioretention
			46	70	Detention basin
			38	109	Media filter (mostly sand)
			51	130	Porous pavement
			51	109	Retention pond
			22	53	Wetland channel
Reddy	2014	Illinois	3-90	L	Media (calcite, zeolite, iron)
			1.2 - 10		Sand filter
Reddy	2014b	Illinois	17	L	Media Filter (biochar)
Ronias	2014	England	40	4	Vegetated Pond

B-8: Median Removal Rates for Zinc					
Author	Year	Location	% Removal	N	BMP
Blecken	2009	Australia	98	L	Vertical flow biofilter
Borne	2013	New Zealand	40	17	Floating treatment wetland
Ladislas	2013	France	72 - 82	L	Floating treatment wetland
Leisenring	2014	ISBD	75	163	Grass strip
			26	286	Grass swale
			75	293	Bioretention
			64	131	Composite
			57	249	Detention basin
			77	358	Media filter (mostly sand)
			79	256	Porous pavement
			56	760	Retention pond
			59	266	Wetland basin
			57	1026	Wetland basin/retention pond
			32	86	Wetland Channel
Li & Davis	2008	Anacostia	83	7	Bioretention
Reddy	2014	Illinois	65 - 99	L	Media (calcite, zeolite, iron)
			43 - 58	L	Sand Filter
Reddy	2014	Illinois	24	L	Media Filter (biochar)
Ronias	2014	England	68	4	Vegetated pond
Stagge	2012	UMD	63*	18	Grass swale
Sun	2007	UMD lab	94*	L	Bioretention
Winer	2000	CBPRD	26	MS	Stormwater dry ponds
			66	MS	Stormwater wet ponds
			44	MS	Stormwater wetlands
			95	MS	Bioretention
			99	MS	Porous pavement
			45	MS	Grass channel
			86	MS	Dry swale
			33	MS	Wet swale

ISBD = International Stormwater BMP Database
CBPRD = Center for Watershed Protection BMP Pollutant Removal Database
MS = median of multiple performance studies
N = number of paired storm events, L= laboratory study, *Mean value.